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Bat Habitat Use and Activity in Forests of Central Louisiana

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BAT HABITAT USE AND ACTIVITY IN FORESTS OF CENTRAL LOUISIANA

A Thesis

Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
in partial fulfillment of the
requirements for the degree of
Master of Science

in

The School of Renewable Natural Resources

by
Jane Marie Kunberger
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May 2022

To mom, dad, and Cori, thank you for always being there for me. And to Luis, who has supported me and stuck with me through thick and thin. I couldn’t have done this without you.

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Abstract

In the southeastern U.S., where forests are the primary land cover type and trees are often harvested for production purposes, understanding how forestry practices influence bat distributions is critical for bat conservation and management. It is also important for researchers to quantify and report variation in the performance of automated recordings units (ARUs) used to survey for bats because several key features of ARUs (e.g., microphone sensitivity, triggering thresholds) can influence an ARUs ability to detect bat calls. My goals were (1) to examine the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana, and (2) to compare the number of bat call files, echolocation pulses, and species recorded by two ultrasonic ARUs (i.e., AudioMoths and Song Meter SM4BAT-FS monitors) and identified using automated classification software (i.e., SonoBat). For (1), I deployed ARUs at sites representing five pine management treatments and bottomland hardwood forests to record bat calls. I also collected environmental data at the landscape and local scales. I detected *Eptesicus fuscus*, *Lasiurus borealis*/*L. seminolus*, *Myotis* species, *Perimyotis subflavus*, *Tadarida brasiliensis*, and *Aeorestes cinereus* during both seasons, and additionally detected *Nycticeius humeralis* during the breeding season. I found that activity was higher at group selection harvest, red-cockaded woodpecker, and clearcut treatments and that habitat use was different between periods for some species. I used ARU data that I collected at the study sites described above and at an urban greenspace in Baton Rouge to address (2). I found that SonoBat identified more call files to species, call files with high-frequency bat calls, echolocation pulses, and species from SM4BAT recordings compared to AudioMoth recordings, but that SonoBat identified a similar number of call files with low-frequency bat calls between monitors. My research identifies forest management practices and habitat characteristics that promote bat species diversity and activity. In addition, my research demonstrates that SM4BATs provide more comprehensive data that can be used with automated classification software than the version of AudioMoths I used, which has implications for survey results and comparability across studies.

Chapter 1. Introduction

Bats are experiencing significant population declines in the U.S. due to disease (e.g., white nose syndrome caused by *Pseudogymnoascus destructans*), collisions with human infrastructure, and habitat loss (Hammerson et al. 2017). Bats in the U.S. are especially affected by the loss of forested habitat because all species depend on trees for foraging, navigation, or roosting during at least some portion of the year (Altringham 2011). Fortunately, commercial forests in the southeastern U.S. provide important roosting and foraging habitat for bats. Management practices used in commercial forests across the southeastern U.S., such as thinning and group selection harvest, (LaFayette et al. 2012), can increase edge habitat where bats forage and decrease understory vegetation structure that can interfere with a bat's ability to navigate the landscape (Loeb 2020). In contrast, clearcutting can decrease the amount of roosting habitat that is available for bats (Loeb 2020). Though we have information on the influence of forest management practices in other areas of the southeast, we have limited data to understand how bats use forested landscapes in Louisiana, which are primarily composed of loblolly pine (*Pinus taeda*) and cover approximately 40% of land cover in the state (Oswalt et al. 2014). Such information could help maintain bat populations in Louisiana and inform management practices that promote bat abundance and diversity in commercial forests throughout Louisiana and across the southeast. In Chapter 2 of my thesis, I examine the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana.

Researchers use both direct (e.g., mist netting, emergence counts) and indirect (e.g., passive acoustic monitoring) methods to evaluate bat-habitat relationships. Passive acoustic monitoring, whereby autonomous recording units (ARUs) record the echolocation calls of bats, is an indirect method that provides information on bat species presence and relative activity

(Gibb et al. 2019). However, ARUs vary in their sensitivities (i.e., the distance at which they record sound), directionalities, storage capacities, and recording capabilities, among others (Limpens and McCracken 2004, Adams et al. 2012, Britzke et al. 2013). Understanding the differences among ARUs is important when designing studies for bats, as units that record fewer usable files could affect survey results. A newly developed and inexpensive ARU, the AudioMoth, can record sound in the audible and ultrasonic spectrums (Hill et al. 2019) and, to date, researchers and hobbyists have purchased >20,000 of the units (GroupGets, LLC 2021). However, we lack data to quantify how this unit compares to other commercially available ARUs that record ultrasound. For Chapter 3, I used the automated classification software SonoBat to compare bat call files, echolocation pulses, and species recorded by AudioMoths and a commonly used full-spectrum monitor, the Song Meter SM4BAT. Overall, my thesis research provides information on the occurrence and habitat use of bats in the commercial forests of Louisiana and emphasizes that researchers should consider all factors (e.g., cost, recording quality, microphone sensitivity) when selecting ARUs for bat surveys.

Chapter 2. The Influence of Forest Management Practices on Seasonal Bat Species Occurrence and Activity in the Southeastern United States

2.1. Introduction

Of more than 1,300 bat species worldwide, 47 occur in the United States, each with unique adaptations to a variety of roosting and foraging conditions (Kalko et al. 2008, Altringham 2011). Such biodiversity is the foundation for ecosystem processes and associated ecosystem services (i.e., human benefits derived from natural processes; Mace et al. 2012), which, for bats, includes nutrient relocation (Jones et al. 2009, Kunz et al. 2011), pollination and seed dispersal (Kunz et al. 2011), and opportunities for ecotourism (Kunz et al. 2011). Insectivorous bats consume insect pests (McCracken et al. 2012, Maas et al. 2013, Russo et al. 2018), an ecological service that helps reduce zoonotic disease transmission to humans and livestock (Schneeberger and Voigt 2016) and contributes \$3.7–54.0 billion per year toward protection of economically important agricultural crops (Boyles et al. 2011).

Nearly one third of bat species in the United States are experiencing population declines associated with habitat loss and degradation (Hammerson et al. 2017), disease (e.g., white-nose syndrome caused by the fungus *Pseudogymnoascus destructans*; Blehert et al. 2009), and collisions with human infrastructure (e.g., wind turbines; Hammerson et al. 2017). Loss of forested habitat has been particularly problematic, as all bats that occur in the United States use forests during some portion of their life cycle (e.g., roosting, foraging; Altringham 2011). Given these declines, several bat species that use forested landscapes are now listed as Threatened (e.g., northern long-eared bat [*Myotis septentrionalis*]) or Endangered (e.g., Indiana bat [*M. sodalis*]), or are currently under review for listing protections by the United States Fish and Wildlife Service (USFWS) (e.g., tricolored bat [*Perimyotis subflavus*]). As such, we require more information on the life histories and ecologies of bat species across the United States to make informed conservation and management decisions in light of current and emerging threats.

We particularly lack information on bat-habitat relationships and bat population dynamics in the southeastern United States, where the estimated area of forest is approximately half of what existed prior to European settlement and timber production is the primary land use (Oswalt et al. 2014). In the southeastern United States, forest management practices are influenced by a variety of factors (e.g., tree species composition, geographic location, local economies; LaFayette et al. 2012), but typically include (1) clearcuts (i.e., all trees and vegetation cleared from a stand); (2) thinning (i.e., proportion of trees harvested from a stand to encourage growth of remaining trees); and (3) group selection harvest (i.e., small plots clearcut throughout the stand to open up room for a new crop while the rest of the stand is allowed to grow) (LaFayette et al. 2012). Forest heterogeneity created by multi-aged stand management may promote bat species diversity and provide increased opportunities for foraging on aerial insects along edge habitat (Grindal and Brigham 1999, Menzel et al. 2002, Tibbels and Kurta 2003). Alternatively, commercially managed forests may provide less roosting habitat because trees are harvested before they are old enough to generate cavities, peeling bark, and snags (Drake et al. 2020).

Forest management practices and the physiognomy of resulting stands are not the only factors that influence bat occurrence, abundance, and foraging activity. Landscape-level environmental factors such as distances to water sources (Krusic et al. 1996, Ford et al. 2005), urban centers (Kurta and Teramino 1992), and mature forests (Grindal and Brigham 1999, Mering and Chambers 2012), as well as the amount of edge habitat (Bender et al. 2015) and nearby forest composition (Bender et al. 2015), can influence bat occurrence and foraging activity. In addition, annual movement patterns could influence trends in bat occurrence, abundance, and foraging activities, as some tree-dwelling bat species are year-round residents of the southeastern United States (e.g., northern yellow bats; *Dasypterus intermedius*) and others are migratory (e.g., hoary bats; *Aeroestes cinereus*) (Cryan 2003, Fraser et al. 2012). Further, distribution, abundance, and diversity of food resources for insectivorous bats can vary widely in response to seasonal patterns in temperature and precipitation (Wolda 1988). To date, most information about bat-habitat relationships and foraging activity in forests of the southeast is derived from data collected during the summer months to align with increased activity of adults and newly volant young (i.e., June–August, hereafter referred to as the breeding period; Harvey et al. 2011). Information collected across multiple seasons could provide important information on the phenology of bat species in this region.

My objective was to examine the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana, a region where commercial forests cover over 40% of the land area (Oswalt et al. 2014). My treatments included loblolly pine (*Pinus taeda*) managed with thinning, group selection harvest, or clearcutting; loblolly pine controls; bottomland hardwoods; and red-cockaded woodpecker habitat predominated by loblolly or longleaf pine (*P. palustris*). I also measured environmental characteristics at local and landscape scales to help explain and predict variation in bat occurrence and activity across my treatments. In addition, I surveyed for bat presence at potential tree roosts and manmade structures in the vicinity of my sampling points to investigate differences in roosting ecology across the treatments.

I expected that I would find more open space aerial foragers (i.e., big brown bats [*Eptesicus fuscus*], hoary bats, and Mexican free-tailed bats [*Tadarida brasiliensis*]) in clearcuts during both the breeding and non-breeding periods because these species have more successful foraging bouts in areas that are open and free from an increased understory vegetation structure (Table 2.1.; Sherwin et al. 2000, Owen et al. 2004, Morris et al. 2010). I expected that species richness would be greatest in forest stands with group selection harvest during both the breeding and non-breeding periods because group selection harvests provide edge habitat, which often equates to greater insect prey diversity and abundance for bats (Table 2.1.; Menzel et al. 2002, Morris et al. 2010), and vegetative conditions preferred by both large and small bat species (i.e., open and intermediate vegetation structure; Table 2.1.; Menzel et al 2002, Morris et al. 2010, Titchenell et al. 2011). However, I expected that the overall species assemblage would change between periods as species that roost in cavities may move into culverts and other human-built structures during the non-breeding period, which could decrease species richness in the forest (Table 2.1.; Stevens et al. 2017). Conversely, bat species that migrate south for the non-breeding period may roost in forested habitats, which could increase species richness (Table 2.1.; Cryan 2003). Finally, I expected that bat activity and foraging activity would be greatest in forest stands with group selection harvest because there is a mosaic of available foraging habitat (Grindal and Brigham 1998, Menzel et al. 2002, Loeb and Waldrop 2008), and that bat activity and foraging activity would be highest during the breeding period for all treatments (Table 2.1.; Kurta et al. 1989, Kunz and Stern 1995).

Table 2.1. Expectations for differences in each response variable across treatments for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (NB; January–February 2020 and December 2020–February 2021) and breeding (B; June–August 2020 and 2021) periods. OS = Open Space Aerial Foragers, ES = Edge Space Aerial Foragers, G = Narrow Space Passive Gleaning Foragers (Denzinger and Schnitzler 2013). My expectations for species richness and foraging activity across forest management treatments (low, medium, and high) are relative and I compared these responses across periods and years.

	Species composition		Species richness		Activity		Foraging activity	
Treatment	B	NB	B	NB	B	NB	B	NB
Clearcut	OS	OS	Medium	Medium	Medium	Low	Medium	Low
Group Selection Harvest	OS, ES	OS, ES	High	High	High	Medium	High	Medium
Thinned	ES	ES	Medium	Low	Medium	Low	Medium	Low
Red-cockaded woodpecker	OS, ES	OS, ES	High	Medium	High	Medium	High	Medium
Control	ES, G	ES, G	Low	Low	Low	Low	Low	Low
Bottomland hardwood	ES, G	ES, G	Low	Low	Low	Low	Low	Low

2.2. Methods

2.2.1. Study Area and Sampling Periods

I conducted my research at the Catahoula and Winn Ranger Districts of the United States Department of Agriculture–Forest Service's Kisatchie National Forest in central Louisiana, USA (Fig. 2.1.). The Kisatchie National Forest encompasses approximately 243,000 ha and includes rolling upland hills, high ridges, and sandy creek bottoms, consisting primarily of pine forests. Commercial pine forests within the region are composed of loblolly or longleaf stands. My study area is located in a subtropical climate, with an average annual rainfall of 114 cm and daily temperatures ranging from around 5˚C in January to around 35˚C in July (National Weather Service 2021). I conducted sampling during the following sampling periods: January–February 2020 (2020 non-breeding period), June–August 2020 (2020 breeding period), December 2020–February 2021 (2021 non-breeding period), and June–August 2021 (2021 breeding period). My sampling periods represent periods associated with torpor (i.e., November–April) or periods associated with increased foraging activity by adults and newly volant young (i.e., July–August) in bats throughout the southeastern United States (Harvey et al. 2011).

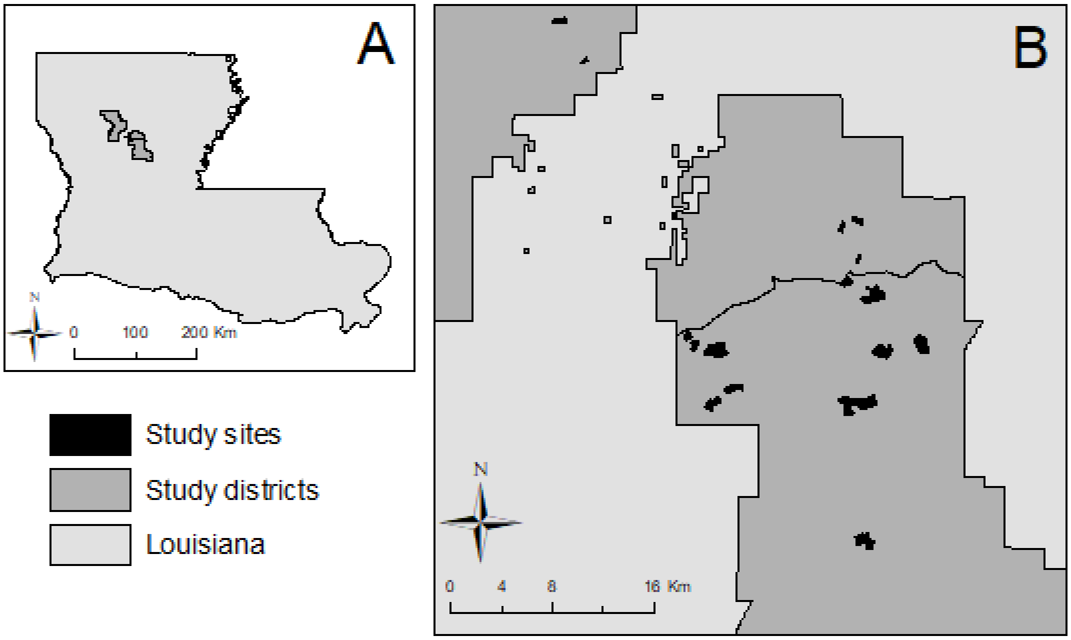

Figure 2.1. Study districts (A) and study sites (B) in the Kisatchie National Forest, Louisiana that I used to examine the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

2.2.2. Study Design

I collected data at 18 study sites (20–140 ha with one site that was 10 ha; Fig. 2.1.) representing six treatments. Of these study sites, three sites were clearcut within five years from the start of my project (designated as clearcuts), three were loblolly pine treated with group selection harvests (group selection harvests), and three were thinned loblolly pine (basal area ~15 m^2/ha; thinned). In addition, three sites were treated to provide habitat for red-cockaded woodpeckers before the start of my project (i.e., thinned old growth pine; hereafter red-cockaded woodpecker habitat) and were composed of either loblolly or longleaf pine. Three sites were composed of mature pine ≥40 years old that were not thinned, burned, or treated with herbicides for ≥5 years (controls). Last, three sites were bottomland hardwood forests ≥40 years old, characterized by intermittent flooding, and predominated by oaks (*Quercus* spp.), gums (*Nyssa* spp.), and bald cypress (*Taxodium distichum*) (hereafter bottomland hardwoods). I sampled 15 sites in 2020 and added two clearcut sites and one red-cockaded woodpecker site in 2021.

Within each study site, I deployed acoustic monitors at two acoustic sampling points located >100 m from the edge (excluding group selection harvests, where I located acoustic sampling points at the edge of a cut, and one loblolly red-cockaded woodpecker site, where I located acoustic sampling points ~80 m from an edge due to area constraints; Owen et al. 2004, Morris et al. 2010) and >200 m from other acoustic sampling points (Hyzy et al. 2020). My study design resulted in sampling at a total of 30 points with acoustic monitors during the non-breeding and breeding periods of 2020 and 36 points with acoustic monitors during the non-breeding and breeding periods of 2021. I resampled the same points during each sampling period.

2.2.3. Ultrasonic Monitoring

I deployed full-spectrum ultrasonic monitors (hereafter monitors; Song Meter SM4BAT-FS Ultrasonic Recorder, Wildlife Acoustics, Inc., Maynard, Massachusetts, U.S.) at acoustic sampling points to record bat echolocation calls during each sampling period. During the 2020 non-breeding period, I deployed monitors for a total of 2 to 4 nights due to time constraints. In subsequent sampling periods, I deployed monitors for a total of 6 to 13 nights. For each deployment, I secured a microphone (SMM-U2 microphone, Wildlife Acoustics, Inc., Maynard, Massachusetts, U.S.) on top of a PVC pole 2 m above the ground, with the microphone sitting flat on top of the pole to direct the microphone upwards (Wildlife Acoustics, Inc. 2018). I placed my acoustic sampling points in areas with minimal foliage to decrease the possibility of missed echolocation calls (Britzke et al. 2013). I programmed monitors to record from 30 min before dusk to 30 min after dawn (Bender et al. 2015) with the default settings, which included triggering at a minimum detected frequency of 16 kHz for a minimum recording length of 1.5 ms and recording at a 256 kHz sample rate with a 12 dB gain.

2.2.4. Live Capture

To verify species presence in my study area, I set up single-high mist nets (~3 m high, 6 m across) and triple-high mist nets (~9 m high, 6 m across) across trails and at edges to catch bats during the 2021 breeding period. I sampled for three hours after sunset and two hours before sunrise each night at three study sites: one group selection harvest site, one red-cockaded woodpecker site, and one control site. I checked nets once every five minutes so that I could remove bats from the nets soon after capture (Kunz and Kurta 1988). Once in hand, I identified each bat to species (Morgan et al. 2019), determined the sex and relative age (i.e., juvenile or adult; Brunet-Rossinni and Wilkinson 2009), and noted whether the individual had wing damage (Reichard and Kunz 2009). I also measured forearm length and weight (Haarsma 2008), as well as ear length, tragus length, body length, tail length, and hindfoot length. My advisor, Dr. Ashley M. Long, obtained permits for these activities from the Louisiana Department of Wildlife and Fisheries (permit number WDP-19-082) and the Louisiana State University Agricultural Center IACUC (protocol number A2019-28), and obtained a permit to handle and process Threatened northern long-eared bats according to the above procedure from the United States Fish and Wildlife Service (permit number 59340D-0).

2.2.5. Environmental Characteristics

I measured environmental characteristics to quantify differences in both landscape and local characteristics across sites, treatments, and seasons, and to determine which factors have the strongest influence on bat species occurrence and activity in my study area. At the landscape scale, I used ArcMap (V. 10.7; ESRI, Redlands, California, USA) to create buffer zones of 50, 100, 450, and 1000 m (1 km) around each acoustic sampling point, which reflected values used in other studies based on the average distance that bats move per night (Bender et al. 2015, Kirkpatrick et al. 2017). I used ArcMap to measure the Euclidean distance (m) from each sampling point to the nearest perennial water source (Bender et al. 2015, Hyzy et al. 2020) with the National Hydrography Dataset (USGS 2020*b*, *c*, 2021) and to measure the Euclidean distance (m) from each sampling point to the nearest incorporated place (Shute et al. 2021) with the National Boundary Dataset (USGS 2020*a*). Within each buffer zone, I calculated the proportion of area attributed to each dominant tree species and the proportion of area attributed to stand age categories (i.e., stand ages 0 to 9 years, 10 to 19 years, 20 to 29 years, and >30 years; Bender et al. 2015) with U.S. Forest Service data for the Kisatchie National Forest (USDA Forest Service n.d.). With the proportion of dominant tree species and stand age data, I calculated forest and age diversity (Bender et al. 2015) using the Shannon Diversity Index (Shannon 1948). I also quantified structural heterogeneity within each buffer by calculating the variation in normalized difference vegetation index (NDVI) with LANDSAT-8 imagery (LANDSAT-8 images courtesy of the U.S. Geological Survey) for each sampling period. Last, I used FRAGSTATS (V 4.2.1; McGarigal and Marks 1995) to calculate total edge (Bender et al. 2015) and proportion of area attributed to wetlands and open water using 2016 National Land Cover Database (NLCD) data (NLCD data courtesy of the U.S. Geological Survey) for each buffer. Before I calculated these metrics, I manually edited the NLCD layer to include clearcuts that occurred after 2016.

At the local scale, I measured vegetation characteristics at the acoustic sampling point and at 25 and 50 m from the acoustic sampling point in each cardinal direction. At each vegetation sampling point, I used a wedge 10-factor prism (Bender et al. 2015) to estimate basal area (m^2/ha). In addition, I visually estimated percent ground cover (i.e., shrubs, herbaceous, or bare; Tibbels and Kurta 2003), measured the DBH of trees >10 cm (Titchenell et al. 2011), and counted the number of saplings (trees <10 cm DBH) within a 5-m radius circle of the vegetation sampling point. Last, I counted the number of snags (dead trees >10 cm diameter at breast height [DBH]; Mattson et al. 1996, Klug et al. 2012, Lucas et al. 2015) and their decomposition state (Thomas et al. 1979) within a 12-m radius circle of the vegetation sampling point (Kirkpatrick et al. 2017).

2.2.6. Roost Surveys

To investigate the influence of forest management practices on bat roosting ecology, I conducted tree roost surveys at my acoustic sampling points and at 25 and 50 m from the acoustic sampling points in each cardinal direction (i.e., the vegetation sampling points described above). I searched for potential roosts within a 12-m radius circle at each sampling point during the breeding period of 2020 and defined potential roosts as trees with woodpecker or decay cavities >3 cm in diameter, large sections of shedding/loose bark, and/or tree decay that created a crevice >3 cm in width (Swystun et al. 2007). Each characterization is based on the common roosting habitat for tree roosting bats (Barclay and Kurta 2007). After I located a potential roost, I surveyed the roost for bat presence using a combination of methods, including shining a spotlight into crevices and cavities and under bark, threading a borescope into crevices and cavities, and carefully moving a wireless cavity inspection camera (available for purchase from ibwo.org) that was attached to the top of a fiberglass telescoping pole into cavities. As such, I only surveyed for potential roosts up to the height of my camera (5.5 m). At each potential roost, I documented the tree species, decay stage, roost type (i.e., woodpecker hole, decay cavity, bark, or crevice), aspect, and roost height (from the bottom of the roost to the ground) (Alston et al.

2019). If I detected a bat, I recorded the bat species and number of individuals present. I revisited potential roosts at least once during each sampling period to survey for bat presence.

In addition, I surveyed infrastructure within 2 km of each acoustic point to identify potential roosting habitat surrounding my study sites. Within my search radius, I surveyed culverts and bridges >2 m in height for bat presence with binoculars, headlamps, and spotlights (Stevens et al. 2017). I documented the structure type (i.e., box culvert, circular culvert, double-T bridge, or other; Fig. A.1.) and, if I detected a bat, the bat species present and number of individuals of each species. I conducted infrastructure surveys at least once during each sampling period to survey for bat presence. I did not handle any bats during tree roost surveys or my surveys of manmade structures.

2.2.7. Echolocation Call Analyses

To evaluate my bat call data, I used the echolocation analysis software SonoBat (V. 4.4.1; SonoBat, Arcata, California, U.S.), which automatically identifies bats to species using call characteristics such as minimum and maximum frequencies, duration, and slope (Fenton and Bell 1981). First, I used the SonoBat batch file scrubber with the settings for medium quality tolerance and ≥20 kHz call search to exclude files that only contained background noise. Then, I ran the remaining files through the SonoBat Southeastern southeast classifier with the default values (acceptable call quality of 0.60 and decision threshold of 0.90) to automatically identify calls to species. Though the automatic classification process can both misidentify and miss calls (Rydell et al. 2017, Nocera et al. 2019), this process provides more reproducible methods compared to manual identification (Nocera et al. 2019).

Because of their similar call structures, SonoBat does not distinguish between eastern red bats [*L. borealis*] and Seminole bats [*L. seminolus*] and refers to all identifications as eastern red bats. Therefore, I combined these species into the eastern red/Seminole bat group for my work (Titchenell et al. 2011). Similarly, of the two *Myotis* species known to occur in Louisiana, SonoBat only identifies southeastern bats [*M. austroriparius*] with the southeastern southeast classifier. As such, I consider SonoBat's identifications of southeastern bats as a combination of southeastern and northern long-eared bats due to their similar call structures and refer to these identifications as the *Myotis* species group (Avila-Flores and Fenton 2005, Titchenell et al. 2011). The probability of acoustic detection should be equal for all species in my study area except Rafinesque's big-eared bat [*Corynorhinus rafinesquii*], which calls at a low amplitude and frequency and is difficult to detect from acoustic surveys; thus, Rafinesque's big-eared bats were likely under-detected in my study (Sherwin et al. 2000).

Following acoustic identification, I calculated species richness and mean activity (mean call files/night) for each species at acoustic sampling points during each sampling period. I also determined mean activity for each period (breeding or non-breeding) by averaging mean activity for each point between sampling periods. I then subset my data to nights with call files $\geq$1 for each species and considered species activity as the number of identified call files at acoustic points during individual nights for later analyses. Then, I used the maximum likelihood estimates from SonoBat output to determine detection or non-detection of a species at each point for each sampling period, considering a likelihood of presence value of $\geq$0.50 as a detection and <0.50 as a non-detection. Last, I defined bat foraging activity as the number of feeding buzzes (i.e., sequences characterized by rapid (~1 ms pauses) calls; Fig. 2.2.) during each night at acoustic sampling points for each sampling period. Due to the large number of files that I collected during the breeding periods, I randomly subsampled 25% of scrubbed files from each point for manual feeding buzz identification during these sampling periods.

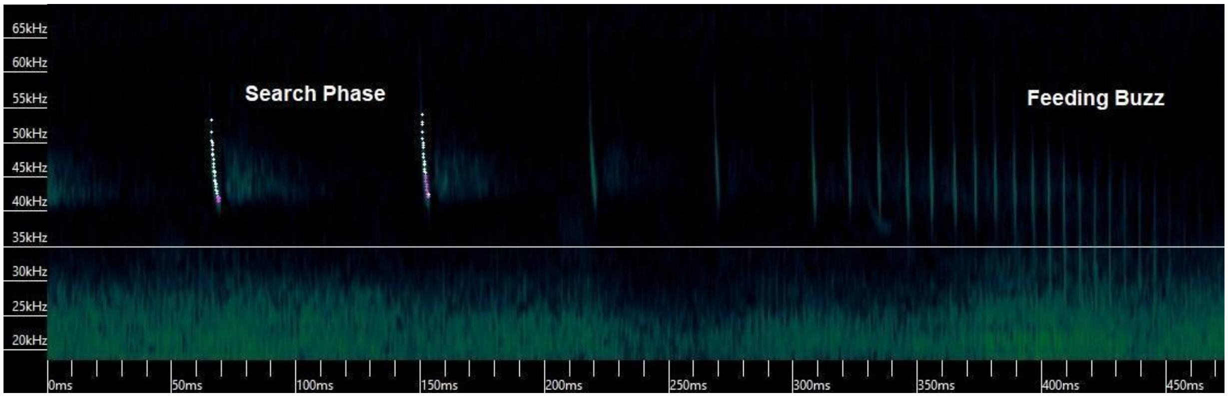

Figure 2.2. An example echolocation sequence from an eastern red/Seminole bat (*Lasiurus borealis/L. seminolus*) where the search phase and feeding buzz are clearly distinguishable, which I quantified to examine the influence of forest management practices on seasonal bat foraging activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. I defined foraging activity as the number of feeding buzzes during each night for acoustic sampling points.

2.2.8. Statistical Analyses

I conducted my statistical analyses in Program R (V. 4.1.1; R Core Team 2021). I used the detection data to determine species occurrence for each period and to investigate differences in bat species composition between the non-breeding and breeding periods. I examined the differences in species richness among treatments for each sampling period using Kruskal-Wallis tests and determined which groups were different using pairwise Wilcoxon rank sum tests (α = 0.05; Kruskal and Wallis 1952, Wilcoxon 1945). I only investigated the differences in species richness among the bottomland hardwood, control, group selection harvest, and thinned treatments during the 2020 non-breeding and breeding periods due to fewer replicates in the red cockaded woodpecker and clearcut treatments during these sampling periods. However, in subsequent sampling periods, I compared species richness across all treatments. I examined the difference in median species richness between the breeding and non-breeding period with a Wilcoxon rank sum test (Wilcoxon 1945) using the coin package in R (v. 1.4-1; Hothorn et al. 2008) and use α = 0.05 to indicate statistical significance. Last, I used median species richness to estimate the magnitude of difference in species richness among treatments and between periods.

I determined foraging group presence at each acoustic point (Denzinger and Schnitzler 2013) to address my expectations (Table 2.1.), considering the foraging group present if at least one species from that foraging group was detected at a point. Of the foraging groups that occur in the Kisatchie National Forest (Stevens et al. 2020), open space aerial foragers include hoary bats, Mexican free-tailed bats, and big brown bats; edge space aerial foragers include eastern red/Seminole bats, evening bats (*Nycticeius humeralis*), and tricolored bats; and narrow space passive gleaning foragers include *Myotis* species and Rafinesque's big-eared bats (Denzinger and Schnitzler 2013).

I examined the differences in the median of each species' mean activity (hereafter relative activity) among treatments for each sampling period using Kruskal-Wallis tests and determined which groups were different using pairwise Wilcoxon rank sum tests ($\alpha = 0.05$). I also examined the difference in relative activity between the breeding and non-breeding period with Wilcoxon rank sum tests and determined the significance of these differences at $\alpha = 0.05$. As with species richness, I only investigated the differences in relative activity among the bottomland hardwood, control, group selection harvest, and thinned treatments during the 2020 non-breeding and breeding periods but compared relative activity among all treatments during the 2021 non-breeding and breeding periods. Last, I used median values of species' relative activity to estimate the magnitude of difference in species' relative activity among treatments.

I chose environmental characteristics for my activity and occupancy analyses by conducting a correlation analysis on all potential variables and retained variables that had low to no correlation with another variable ($r < |0.50|$; Asuero et al. 2006). If two variables had a moderate to very high correlation ($r \geq |0.50|$; Asuero et al. 2006), I retained the most relevant environmental characteristic based on previous research. Relevant variables included distance to water (m), distance to an incorporated area (m), variance in NDVI in a 50 m radius, variance in NDVI in a 1 km radius, total edge in a 1 km radius (m), forest diversity in a 1 km radius (Shannon diversity value), proportion of forest aged >30 years in a 1 km radius, percent shrub cover, basal area (m^2/ha), DBH (cm), and snag density (snags of decay categories 3–6/ha).

To investigate the influence of treatment on species' activity, I created proportional odds models (McCullagh 1980) with a logit link function for each period using the MASS package in R (V. 7.3-54; Venables and Ripley 2002). I categorized species' activity during each night as being "low" (1–5 call files), "moderate" (6–15 call files), or "high" (>15 call files) based on natural breaks in my data and used these grouped categories my response variable. I used treatment as the explanatory variable to look at how activity differed between treatments for each species. In addition, I used pairwise comparisons to compare the odds of high activity among all treatments. I found that activity was too low to model during the non-breeding period for all species except for the eastern red/Seminole bat group, so I only included this species group in the non-breeding period analysis. I exponentiated estimates, standard errors, and 95% confidence intervals to examine the odds ratios of high activity at each treatment and only considered treatments with 95% confidence intervals that did not include 1 as informative (McCullagh 1980). As such, I interpreted odds ratios that were below 1 as having lower odds of high activity at the treatment compared to the reference treatment, whereas I interpreted odds ratios above 1 as having higher odds of high activity at the treatment compared to the reference treatment. Similarly, if the 95% confidence intervals included 1, I concluded that the odds of high activity at the treatment were not different from 1 when compared to the reference treatment. Last, I followed the same proportional odds modeling protocol to analyze my foraging activity data.

To investigate the associations in activity among environmental characteristics, treatments, and species, I conducted non-metric multidimensional scaling (NMDS) using the vegan package (V. 2.5-7; Oksanen et al. 2020). I first subset my data to exclude excess sampling nights from the 2020 breeding period, 2021 non-breeding period, and 2021 breeding period. To do this, I subset the first three nights from the first sampling occasion and the first three nights from the second sampling occasion for each point within each sampling period, resulting in six nights of data for each point during each sampling period. Then, I calculated species' proportional activity within each sampling period. To do this, I divided the total call files for a species at each point by the total call files identified as the species across all points (Nelson and Gillam 2020). I sampled <6 nights during the 2020 non-breeding period; therefore, I retained all nights for this sampling period. Additionally, I excluded species with activity at <6 points and points with no activity for any species.

To create the ordination space for each sampling period, I determined Bray-Curtis dissimilarity distances using the metaMDS function with a maximum of 100 tries to find the best or convergent solution using my proportional activity data (McCune and Grace 2002). I chose the minimum number of dimensions that yielded a result with stress <0.20 (McCune and Grace 2002). I created separate ordinations for each sampling period because I added acoustic points during the 2021 sampling periods, sampled fewer nights during the 2020 non-breeding period, and had different variation in NDVI values for each sampling period. Last, I used the uncorrelated environmental variables above with the envfit function and 999 iterations to investigate the associations of proportional activity with environmental characteristics. I plotted environmental variables that were significantly correlated with the ordination ($\alpha < 0.10$) and examined the amount of variation that they described (r^2). I then summarized the associations of species, points, and environmental variables in each NMDS by their positions in the ordination space (Nelson and Gillam 2020).

I investigated the predicted probability of occupancy for each bat species to identify which environmental characteristics best predict species' occupancy in managed forests (Burnham and Anderson 2002). Because bats are highly mobile and violate the assumption of a closed population (MacKenzie and Royle 2005), my analyses are representative of habitat use rather than occupancy; however, I will use the word "occupancy" to describe these analyses for this project. I only created models for species with detections or non-detections at >6 points within a period (breeding or non-breeding) to avoid overdispersion of data. Last, I ran the breeding and non-breeding period data separately because I assumed that occupancy state changed between periods due to differences in foraging and roosting habitat, as well as possible migration of certain species, between the breeding and non-breeding periods (Cryan 2003, Stevens et al. 2017, Stevens et al. 2020).

First, I scaled uncorrelated environmental characteristic data to ensure that variables were comparable. From the resulting variables, I created models representing predictions about bat foraging group and species occupancy based on previous research, using my detection data as the response variable (Table 2.2.). I created my models using generalized linear modeling with a binomial distribution and logit link function with the lme4 package in R (V. 1.1-27.1; Bates et al. 2015). I included year as a parameter in my models to capture between-year differences in site occupancy.

I examined over-dispersion by calculating $\hat{c}$ values for the most parameterized model for each species, which included all potential parameters (MacKenzie and Bailey 2004). I considered data over-dispersed if $\hat{c} \geq 2$ and used QAICc model selection procedures in subsequent analyses

(MacKenzie and Bailey 2004). If $\hat{c}$ <2, I used AICc model selection procedures to examine model fit (Anderson 2008). I considered the top model set for each species as models with ΔAICc or ΔQAICc <2, which indicated that the models had sufficient evidence to describe occupancy compared to the other models in the candidate set (Anderson 2008). I used the AICcmodavg package in R (V. 2.3-1; Mazerolle 2020) to perform model selection procedures. I reported parameter estimates and their 95% confidence intervals for the models in the top model set to assess the magnitude and direction of effect of each environmental characteristic on the predicted probability of occupancy (Austin et al. 2020). If a 95% confidence interval included 0 for a parameter, I considered the parameter uninformative (Austin et al. 2020). I graphed plausible models using the ggplot2 package in R (V. 3.3.5; Wickham 2016) to visualize the effect of parameters on the predicted probability of occupancy.

2.2.9. Roosting Habitat Analyses

I investigated the influence of forest management practices on available roosting habitat and the roosting ecology of bats by identifying potential roosts in and around my study sites. I examined the differences in available roosting trees among treatments and described the characteristics of potential tree roosts. In addition, I summarized results from infrastructure surveys across sampling periods, highlighting bat species presence and structural characteristics that promoted bat presence.

Table 2.2. The models that I used to investigate the predicted probability of occupancy of bat species during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana. I created each model based on research conducted outside of the state of Louisiana that I considered plausible for my study area.

Model	Description	Citations
Constant (null)	Occupancy is constant across acoustic points, meaning that no explanatory variables influence occupancy.	
Year	Occupancy is influenced by year only. All subsequent models include year as a parameter.	
Treatment	Treatment could influence bats due to the different ways areas are managed and the habitats that result from these practices.	Loeb 2020
Distance to water	The distance to a perennial water source (m). Some species are more likely to occur in areas near permanent water sources because water attracts insects and is crucial for hydration.	Bender et al. 2015, Rojas et al. 2019, Hyzy et al. 2020, Gallagher et al. 2021
Distance to developed	The distance to an incorporated place (m). Developed areas can provide manmade roosting habitat, aggregations of insects at lights, and more open areas for navigating/foraging; however, developed areas can remove natural roosting habitat (e.g., snags) and decrease the amount of potential forest foraging habitat.	Rojas et al. 2019, Cable et al. 2021
Variance in NDVI (50 m)	The variation in NDVI in a 50 m radius. This measure can indicate differences in local habitat structure based on reflectance, helping identify areas with open spaces and identify edge habitat.	Cable et al. 2021
Variance in NDVI (1 km)	The variation in NDVI in a 1 km radius. This measure can indicate different habitats based on their structural characteristics at a landscape scale.	Cable et al. 2021

Table cont'd.

Model	Description	Citations
Total edge (1 km)	The total amount of edge in a 1 km radius (m). Edges provide foraging habitat for multiple species, including edge space aerial foragers.	O'Keefe et al. 2009, Bender et al. 2015
Forest diversity (1 km)	The diversity of forest habitats at a 1 km radius (Shannon diversity value), indicating a mosaic of forest habitats at a landscape scale. High values could indicate a mosaic of foraging, roosting, and navigating habitat for bats.	Bender et al. 2015, Cable et al. 2021
Proportion forest aged >30 years (1 km)	The proportion of forest area in a 1 km radius that is aged >30 years. This measure indicates older growth in the surrounding landscape that could provide opportunities for roosting.	Bender et al. 2015
Percent shrub cover	Percent shrub cover can represent canopy cover and the distance between trees. In addition, shrubby areas may provide habitat for different insect communities compared to herbaceous cover.	Summerville and Crist 2002, Ulyshen et al. 2005, Dodd et al. 2012
Basal area	Basal area (m^2/ha) is an estimate of the area that trees occupy. High basal area could negatively influence open space aerial forager presence because of increased understory vegetation structure, whereas narrow space passive gleaning foragers may be present in areas with high basal area because they can navigate through understory vegetation and glean off surfaces.	Yates and Muzika 2006, Bender et al. 2015, Gallagher et al. 2021
Snag density	Estimated density of snags (decay categories 3–6) per hectare. This is an estimate of the available snag roosting habitat, as snags can provide cavities and peeling bark that bats can use.	Loeb and Waldrop 2008, Lucas et al. 2015, Ketzler et al. 2018, Alder et al. 2021

Table cont'd.

Model	Description	Citations
DBH	Mean DBH of trees (cm) is a measure of tree age and tree composition. This measure can indicate the presence of roosting habitat (larger DBH is typically chosen for bat species that roost in cavities, under bark, and in foliage).	O’Keefe et al. 2009, Lucas et al. 2015, Castleberry et al. 2020, Monarchino et al. 2020, Alder et al. 2021
Variance in NDVI (50 m) + percent shrub cover	Model that represents edges between open areas and areas with increased understory vegetation structure, which may benefit edge space aerial foragers (i.e., evening, eastern red, tricolored bats) and provide high insect abundances for foraging.	Hunter 2002, O’Keefe et al. 2009
Variance in NDVI (1 km) + DBH	Model that represents older stands within a heterogenous landscape, providing both roosting and foraging opportunities for bats.	Lucas et al. 2015
Basal area + percent shrub cover	Model that represents sites with increased understory vegetation structure, which can negatively affect bat presence for edge and open space aerial foragers.	Bender et al. 2015, Gallagher et al. 2021
Snag density + distance to water	Model that represents riparian areas, which are found to be important to bats in previous studies by providing drinking water, foraging areas, and roosting sites	Cable et al. 2021
DBH + snag density	Model that represents areas that provide roosting habitat for both foliage and cavity roosters.	Lucas et al. 2015, Monarchino et al. 2020

2.3. Results

I collected data from 93 recording nights during the 2020 non-breeding period, 183 recording nights during the 2020 breeding period, 248 recording nights during the 2021 non-breeding period, and 233 recording nights during the 2021 breeding period. SonoBat identified 643 files to species during the 2020 non-breeding period, 17,216 files to species during the 2020 breeding period, 979 files to species during the 2021 non-breeding period, and 24,957 files to species during the 2021 breeding period.

2.3.1. Species Occurrence and Seasonality

I found that species richness was similar among treatments during the 2020 non-breeding period ($H_3 = 1.71$, $P = 0.63$), the 2020 breeding period ($H_3 = 0.54$, $P = 0.91$), the 2021 non-breeding period ($H_5 = 5.61$, $P = 0.35$), and the 2021 breeding period ($H_5 = 10.43$, $P = 0.06$). However, I found that species richness at acoustic points was 6 times greater during the breeding period compared to the non-breeding period ($Z = 7.28$, $P \leq 0.001$). I detected the eastern red/Seminole bat group, the *Myotis* species group, tricolored bats, big brown bats, hoary bats, and Mexican free-tailed bats during both the breeding and non-breeding periods, and additionally detected evening bats during the breeding period. I only detected Rafinesque’s big-eared bats at one thinned point during the 2020 non-breeding period; therefore, I excluded Rafinesque’s big-eared bats from further analysis. I detected silver-haired bats (*Lasionycteris noctivagans*) at two points during the non-breeding period and six points during the breeding period; however, due to their similarity in call structure with hoary bats and their rarity in the Kisatchie National Forest (Stevens et al. 2020), I did not analyze silver-haired bats further. I detected all three foraging groups at almost all treatments during the non-breeding and breeding periods (Table 2.3.). I did not detect narrow space passive gleaning foragers at the control treatment during the non-breeding period (Table 2.3.). In addition, I did not detect narrow space passive gleaning foragers at the clearcut treatment during the breeding period (Table 2.3.).

Table 2.3. The number of acoustic points where I detected at least one open space aerial forager (OS), edge space aerial forager (ES), or narrow space passive gleaning forager (N) during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana. I combined results from 2020 and 2021 for each point, resulting in 36 points total.

	Non-breeding			Breeding		
Treatment	OS	ES	N	OS	ES	N
Control	2	3	0	6	6	3
Group Selection Harvest	2	5	4	6	6	3
Thinned	1	4	4	6	6	2
Clearcut	1	4	2	6	6	0
Red-cockaded woodpecker	5	5	1	6	6	1
Bottomland hardwood	3	4	2	6	6	4

I confirmed species presence for both big brown bats and northern long-eared bats from mist netting activities on three nights in August 2021. I captured one adult male big brown bat individual at a red-cockaded woodpecker site on 18 August 2021 at 0530. This individual had a forearm length of 47 mm, an ear length of 14 mm, a tragus length of 8 mm, a body length of 63 mm, a tail length of 42 mm, and a hindfoot length of 8 mm. I also captured one male adult northern long-eared bat at a control site on 21 August 2021 at 0543. This individual had a forearm length of 38 mm, an ear length of 17 mm, a tragus length of 10 mm, a body length of 45 mm, a tail length of 32 mm, and a hindfoot length of 6 mm.

2.3.2. Activity

I found that, on average, relative activity was 25 times greater for all species during the breeding compared to the non-breeding period (Table 2.4.). In contrast, I found that relative activity was similar among treatments for all species during the 2020 non-breeding period (Table A.1.). During the 2020 breeding period, I found that tricolored bat relative activity was, on average, 6 times greater at group selection harvests compared to the bottomland hardwood and control treatments and 3 times greater at the thinned treatment compared to the bottomland hardwood treatment (H_3= 14.06, P = 0.003; Fig. 2.3.). Similarly, I found that eastern red/Seminole bat relative activity was 23 times greater at the group selection harvest treatment and 4 times greater at the thinned treatment compared to the bottomland hardwood treatment during the 2020 breeding period (H_3= 9.53, P = 0.02; Fig. 2.3.).

Table 2.4. The mean and standard errors for species' relative activity between the non-breeding (January–February 2020 and December 2020–February 2021) and breeding periods (June–August 2020 and 2021) and the Wilcoxon signed-rank test results for each species for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana. I defined relative activity as the mean call files/night at each point.

Species	Non-breeding	Breeding	*Z*-value	*P*-value
Eastern red/Seminole	2.24 ± 0.59	21.56 ± 3.07	6.11	<0.001
Tricolored	0.28 ± 0.06	15.03 ± 3.13	7.20	<0.001
Big brown	0.91 ± 0.63	14.79 ± 3.44	6.83	<0.001
Hoary	0.38 ± 0.17	1.59 ± 0.31	4.18	<0.001
Mexican free-tailed	0.67 ± 0.30	42.76 ± 5.47	7.28	<0.001
Myotis species	0.47 ± 0.17	0.98 ± 0.14	3.86	<0.001

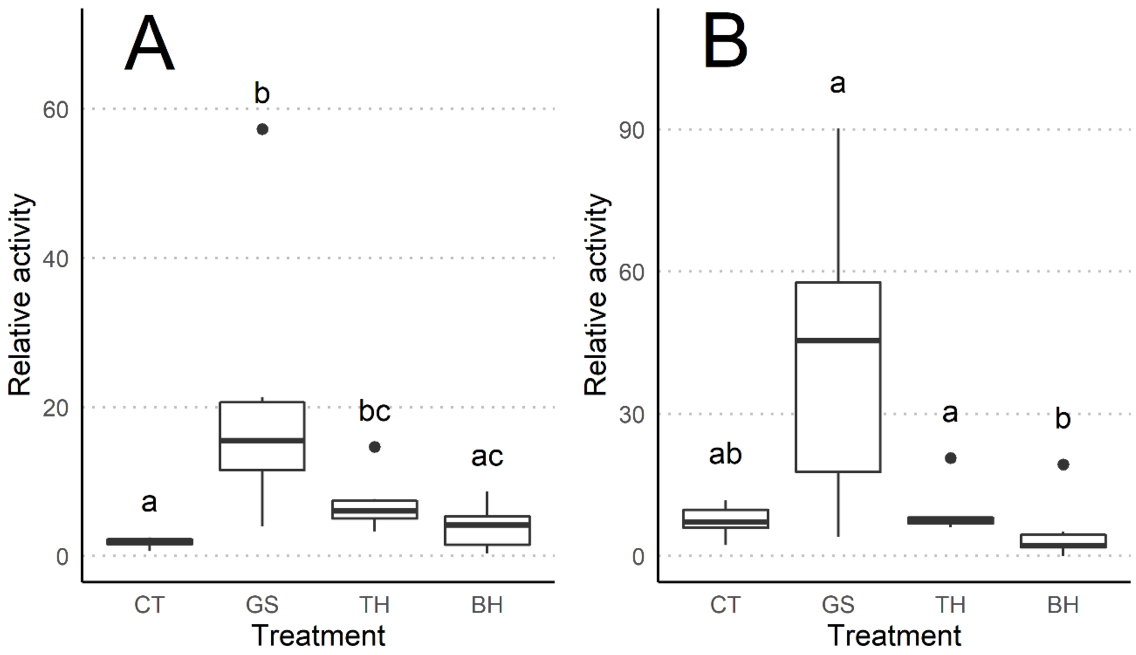

Figure 2.3. Boxplots showing difference in the median of relative activity for tricolored bats (A) and eastern red/Seminole bats (B) during the 2020 breeding period (June–August) for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. Treatments are control (CT), group selection harvest (GS), thinned (TH), and bottomland hardwood (BH).

I found that big brown bat relative activity was greater at the red-cockaded woodpecker treatment compared to control and thinned treatments during the 2021 non-breeding period because big brown bat relative activity was 0 at these treatments ($H_5 = 11.45$, $P = 0.04$; Fig. 2.4.). During the 2021 breeding period, I found that *Myotis* species relative activity was greater at the group selection harvest, thinned, and bottomland hardwood treatments compared to the clearcut treatment because *Myotis* were not active at these treatments ($H_5 = 13.38$, $P = 0.02$; Fig. 2.5.). Similarly, I found that *Myotis* species relative activity was greater at the thinned treatment compared to the red-cockaded woodpecker treatment because *Myotis* species relative activity was 0 at this treatment ($H_5 = 13.38$, $P = 0.02$; Fig. 2.5.).

I found that, during the 2021 breeding period, tricolored bat relative activity was an average of 12 times greater at the bottomland hardwood, group selection harvest, thinned, clearcut, and red-cockaded woodpecker treatments compared to the control treatment (H_5 = 18.93, P = 0.002; Fig. 2.5.). In addition, I found that tricolored bat relative activity was 2 times greater at the clearcut treatment compared to the thinned treatment (H_5 = 18.93, P = 0.002; Fig. 2.5.). I found that eastern red/Seminole bat relative activity was, on average, 4 times greater at group selection harvests compared to thinned, bottomland hardwood, and control treatments during the 2021 breeding period (H_5 = 22.64, $P \leq 0.001$; Fig. 2.5.). In addition, I found that eastern red/Seminole bat relative activity was, on average, 6 times greater at clearcuts compared to thinned, bottomland hardwood, and control treatments (H_5 = 22.64, $P \leq 0.001$; Fig. 2.5.). Last, I found that eastern red/Seminole bat relative activity was 9 times greater at the red-cockaded woodpecker treatment compared to the bottomland hardwood treatment (H_5 = 22.64, $P \leq 0.001$; Fig. 2.5.).

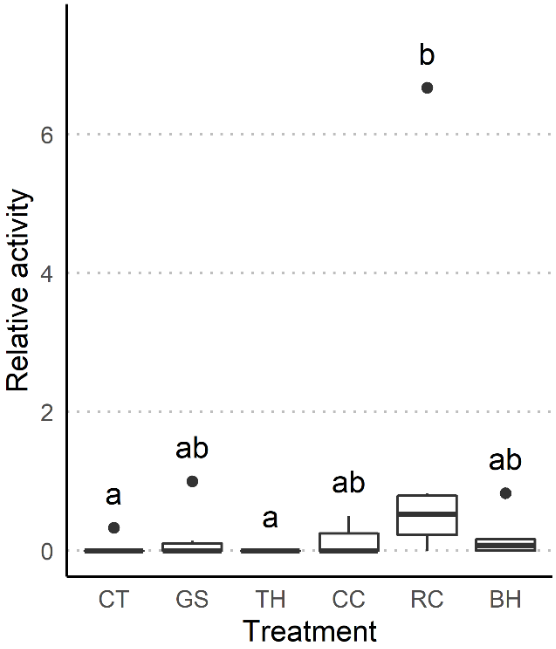

Figure 2.4. Boxplots showing the difference in the median of relative activity among treatments for big brown bats during the 2021 non-breeding period (December 2020–February 2021) for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. Treatments are control (CT), group selection harvest (GS), thinned (TH), clearcut (CC), red-cockaded woodpecker habitat (RC), and bottomland hardwood (BH).

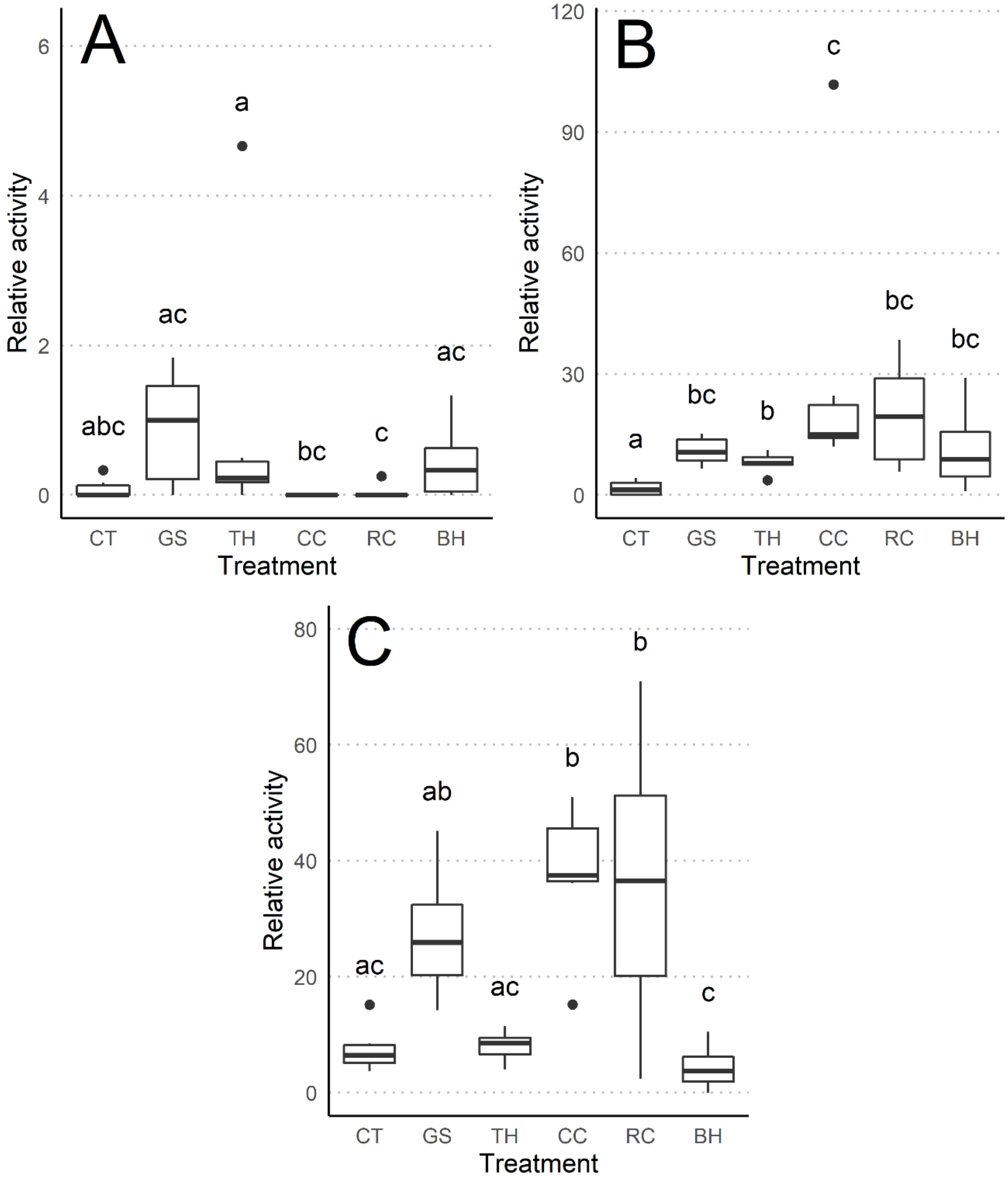

Figure 2.5. Boxplots showing the difference in the median of relative activity among treatments for *Myotis* species (A), tricolored bats (B), and eastern red/Seminole bats (C) during the 2021 breeding period (June–August) for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. Treatments are control (CT), group selection harvest (GS), thinned (TH), clearcut (CC), red-cockaded woodpecker habitat (RC), and bottomland hardwood (BH).

During the non-breeding period, I only used proportional odds modeling to analyze eastern red/Seminole bat group activity. I found that the confidence interval for the group selection harvest treatment compared to the bottomland hardwood treatment did not include 1 for eastern red/Seminole bats during the non-breeding period (Table 2.5.). Similarly, I found that the confidence intervals for the thinned treatment compared to the group selection harvest treatment and compared to the red-cockaded woodpecker treatment did not include 1 during the non-breeding period (Table 2.5.). My results suggested that the odds of detecting high activity from an eastern red/Seminole bat in the group selection harvest treatment was 6 times the odds of detecting high activity from an eastern red/Seminole bat in the bottomland hardwood treatment during the non-breeding period (Table 2.5.). My results also suggested that the odds of detecting high activity from an eastern red/Seminole bat at the thinned treatment was 0.1 times the odds of detecting high activity from an eastern red/Seminole bat in the group selection harvest and in the red-cockaded woodpecker treatments during the non-breeding period (Table 2.5.).

Table 2.5. Eastern red/Seminole bat proportional odds model results for the non-breeding period (January–February 2020 and December 2020–February 2021), including odds ratios ± SE and 95% confidence intervals, for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. Asterisks denote confidence intervals that do not include 1. Treatments are bottomland hardwood (BH), clearcut (CC), control (CT), group selection harvest (GS), red-cockaded woodpecker habitat (RC), and thinned (TH).

	Reference treatment					
Treatment	BH	CC	CT	GS	RC	TH
BH	0	-	-	-	-	-
	-	-	-	-	-	-
CC	1.48 ± 3.06	0	-	-	-	-
	(0.15, 15.08)	-	-	-	-	-
CT	2.81 ± 2.88	1.90 ± 2.94	0	-	-	-
	(0.36, 26.85)	(0.23, 18.68)	-	-	-	-
GS	6.43 ± 2.66	4.34 ± 2.72	2.29 ± 2.52	0	-	-
	(1.06, 55.73)*	(0.67, 38.77)	(0.38, 15.59)	-	-	-
RC	3.59 ± 2.46	2.42 ± 2.53	1.28 ± 2.33	0.56 ± 2.10	0	-
	(0.69, 27.68)	(0.44, 19.27)	(0.25, 7.54)	(0.13, 2.41)	-	-
TH	0.36 ± 3.66	0.25 ± 3.72	0.13 ± 3.53	0.06 ± 3.31	0.10 ± 3.11	0
	(0.02, 4.34)	(0.01, 3.02)	(0.01, 1.26)	(0.003, 0.44)*	(0.01, 0.68)*	-

I was able to use proportional odds modeling to analyze all species that were present during the breeding period, excluding *Myotis* species. I found that the confidence intervals for the clearcut, group selection harvest, and red-cockaded woodpecker treatments compared to the bottomland hardwood treatment did not include 1 for eastern red/Seminole bats (Table 2.6.). I also found that the confidence intervals for the control and thinned treatments compared to the clearcut treatment, the group selection harvest and red-cockaded woodpecker treatments compared to the control treatment, and the thinned treatment compared to the group selection harvest and red-cockaded woodpecker treatments did not include 1 for eastern red/Seminole bats (Table 2.6.). My estimates suggested that the odds of detecting high activity from an eastern red/Seminole bat in the clearcut, group selection harvest, and red-cockaded woodpecker treatments was 7–10 times the odds of detecting high activity from an eastern red/Seminole bat in the bottomland hardwood treatment (Table 2.6.). In addition, I found that the odds of detecting high activity from an eastern red/Seminole bat in the control and thinned treatments was 0.1 times the odds of detecting high activity in the clearcut treatment (Table 2.6.). My estimates also suggested that the odds of detecting high activity from an eastern red/Seminole bat in the group selection harvest and red-cockaded woodpecker treatments was 3 and 2 times the odds of detecting high activity in the control treatment, respectively (Table 2.6.). Last, my results suggested that the odds of detecting high activity from an eastern red/Seminole bat in the thinned treatment was 0.1 times the odds of detecting high activity in the group selection harvest and red-cockaded woodpecker treatments (Table 2.6.).

I found that the confidence intervals for the clearcut, group selection harvest, and red-cockaded woodpecker treatments compared to the bottomland hardwood treatment did not include 1 for big brown bats (Table 2.6.). I also found that the confidence intervals for the red-cockaded woodpecker treatment compared to the control treatment and for the thinned treatment compared to the red-cockaded woodpecker treatment did not include 1 for big brown bats (Table 2.6.). My results suggested that the odds of detecting high activity from a big brown bat in the clearcut, group selection harvest, and red-cockaded treatments was 2–4 times the odds of detecting high activity from a big brown bat in the bottomland hardwood treatment (Table 2.6.). My results also suggested that the odds of detecting high activity from a big brown bat in the red-cockaded woodpecker treatment was 2 times the odds of detecting high activity from a big brown bat in the control treatment (Table 2.6.). Last, my results suggested that the odds of detecting high activity from a big brown bat in the thinned treatment was 0.4 times the odds of detecting high activity from a big brown bat in the red-cockaded woodpecker treatment (Table 2.6.).

I found that the confidence intervals for the clearcut, group selection harvest, and red-cockaded woodpecker treatments compared to the bottomland hardwood treatment did not include 1 for tricolored bats (Table 2.6.). In addition, I found that the confidence intervals for the control and thinned treatments compared to the clearcut treatment and the group selection harvest, red-cockaded woodpecker, and thinned treatments compared to the control treatment did not include 1 for tricolored bats (Table 2.6.). Last, I found that the confidence intervals for the thinned treatment compared to the group selection harvest and red-cockaded woodpecker treatments did not include 1 for tricolored bats (Table 2.6.). My estimates suggested that the odds of detecting high activity from a tricolored bat in the clearcut, group selection harvest, and red-cockaded woodpecker treatments was 6–8 times the odds of detecting high activity from a tricolored bat in the bottomland hardwood treatment (Table 2.6.). In addition, my estimates suggested that the odds of detecting high activity from a tricolored bat in the control and thinned treatments was 0.03 and 0.2 times the odds of detecting high activity from a tricolored bat in the clearcut treatment (Table 2.6.). Further, my estimates suggested that the odds of detecting high activity from a tricolored bat in the group selection harvest, red-cockaded woodpecker, and thinned treatments was 6–30 times the odds of detecting high activity from a tricolored bat in the control treatment (Table 2.6.). Last, I found that the odds of detecting high activity from a tricolored bat in the thinned treatment was 0.1 times the odds of detecting high activity from a tricolored bat in the group selection harvest and red-cockaded woodpecker treatments (Table 2.6.).

I found that the confidence intervals for the clearcut, group selection harvest, and red-cockaded woodpecker treatments compared to the bottomland hardwood treatment did not include 1 for evening bats (Table 2.6.). I also found that the confidence intervals for the control and group selection harvest treatments compared to the clearcut treatment did not include 1 for evening bats (Table 2.6.). In addition, I found that the confidence intervals for the group selection harvest and red-cockaded woodpecker treatments compared to the control treatment did not include 1 for evening bats (Table 2.6.). Last, I found that the confidence intervals for the thinned treatment compared to the group selection harvest and red-cockaded woodpecker treatments did not include 1 for evening bats (Table 2.6.). My estimates suggested that the odds of detecting high activity from an evening bat in the clearcut, group selection harvest, and red-cockaded woodpecker treatments was 3–7 times the odds of detecting high activity from an evening bat in the bottomland hardwood treatment (Table 2.6.). My estimates also suggested that the odds of detecting high activity from an evening bat in the control and group selection harvest treatments was 0.3 and 2 times the odds of detecting high activity from an evening bat in the clearcut treatment, respectively (Table 2.6.). In addition, my results suggested that the odds of detecting high activity from an evening bat in the group selection harvest and red-cockaded woodpecker treatments was 7 and 4 times the odds of detecting high activity from an evening bat in the control treatment, respectively (Table 2.6.). Last, my results suggested that the odds of detecting high activity from an evening bat in the thinned treatment was 0.2 and 0.4 times the odds of detecting high activity from an evening bat in the group selection harvest and red-cockaded woodpecker treatments, respectively (Table 2.6.).

I found that the confidence intervals for the control, clearcut, and group selection harvest treatments compared to the bottomland hardwood treatment did not include 1 for Mexican free-tailed bats (Table 2.6.). In addition, I found that the confidence intervals for the group selection harvest, red-cockaded woodpecker, and thinned treatments compared to the clearcut treatment did not include 1 for Mexican free-tailed bats (Table 2.6.). I also found that the confidence intervals for the red-cockaded woodpecker and thinned treatment compared to the control treatment did not include 1 for Mexican free-tailed bats (Table 2.6.). My estimates suggested that the odds of detecting high activity from a Mexican free-tailed bat in the control, clearcut, and group selection harvest treatments was 2–4 times the odds of detecting high activity from a Mexican free-tailed bat in the bottomland hardwood treatment (Table 2.6.). My results also suggested that the odds of detecting high activity from a Mexican free-tailed bat in the group selection harvest, red-cockaded woodpecker, and thinned treatments was 0.3–0.5 times the odds of detecting high activity from a Mexican free-tailed bat in the clearcut treatment (Table 2.6.). Similarly, my results suggest that the odds of detecting high activity from a Mexican free-tailed bat in the red-cockaded woodpecker and thinned treatments was 0.4 times the odds of detecting high activity from a Mexican free-tailed bat in the control treatment (Table 2.6.). Last, I found that the confidence intervals for all treatment comparisons included 1 for hoary bats (Table 2.6.).

Table 2.6. Proportional odds model results for the breeding period, including odds ratios ± SE and 95% confidence intervals, for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. Asterisks denote confidence intervals that do not include 1. Treatments are bottomland hardwood (BH), clearcut (CC), control (CT), group selection harvest (GS), red-cockaded woodpecker habitat (RC), and thinned (TH).

		Reference treatment					
Species	Treatment	BH	CC	CT	GS	RC	TH
Eastern red/Seminole	BH	0	-	-	-	-	-
		-	-	-	-	-	-
	CC	7.21 ± 1.54	0	-	-	-	-
		(3.14, 17.15)*	-	-	-	-	-
	CT	1.67 ± 1.47	0.23 ± 1.50	0	-	-	-
		(0.78, 3.60)	(0.10, 0.50)*	-	-	-	-
	GS	9.72 ± 1.51	1.35 ± 1.52	5.82 ± 1.47	0	-	-
		(4.40, 22.22)*	(0.59, 3.06)	(2.78, 12.55)*	-	-	-
	RC	7.48 ± 1.53	1.04 ± 1.54	4.48 ± 1.49	0.77 ± 1.51	0	-
		(3.31, 17.49)*	(0.44, 2.41)	(2.09, 9.90)*	(0.34, 1.72)	-	-
	TH	1.43 ± 1.44	0.20 ± 1.47	0.86 ± 1.40	0.15 ± 1.44	0.19 ± 1.46	0
		(0.70, 2.96)	(0.09, 0.42)*	(0.44, 1.66)	(0.07, 0.29)*	(0.09, 0.39)*	-
Big brown	BH	0	-	-	-	-	-
		-	-	-	-	-	-
	CC	2.56 ± 1.51	0	-	-	-	-
		(1.16, 5.79)*	-	-	-	-	-
	CT	1.75 ± 1.51	0.68 ± 1.46	0	-	-	-
		(0.78, 3.99)	(0.32, 1.44)	-	-	-	-
	GS	2.19 ± 1.47	0.86 ± 1.41	1.25 ± 1.42	0	-	-
		(1.04, 4.73)*	(0.43, 1.68)	(0.63, 2.51)	-	-	-
	RC	4.12 ± 1.49	1.61 ± 1.43	2.35 ± 1.44	1.88 ± 1.39	0	-
		(1.90, 9.14)*	(0.80, 3.26)	(1.15, 4.85)*	(0.98, 3.62)	-	-
	TH	1.58 ± 1.49	0.62 ± 1.43	0.90 ± 1.44	0.72 ± 1.40	0.38 ± 1.42	0
		(0.73, 3.49)	(0.30, 1.25)	(0.44, 1.86)	(0.37, 1.39)	(0.19, 0.76)*	-

Table cont'd.

		Reference treatment					
Species	Treatment	BH	CC	CT	GS	RC	TH
Tricolored	BH	0	-	-	-	-	-
		-	-	-	-	-	-
	CC	7.99 ± 1.53	0	-	-	-	-
		(3.53, 18.71)*	-	-	-	-	-
	CT	0.23 ± 1.60	0.03 ± 1.66	0	-	-	-
		(0.09, 0.55)*	(0.01, 0.07)*	-	-	-	-
	GS	6.81 ± 1.46	0.85 ± 1.48	29.94 ± 1.60	0	-	-
		(3.26, 14.58)*	(0.39, 1.84)	(12.35, 78.64)*	-	-	-
	RC	6.49 ± 1.51	0.81 ± 1.53	28.54 ± 1.64	0.95 ± 1.47	0	-
		(2.93, 14.77)*	(0.35, 1.86)	(11.22, 78.52)*	(0.45, 2.03)	-	-
	TH	1.33 ± 1.43	0.17 ± 1.48	5.83 ± 1.56	0.19 ± 1.41	0.20 ± 1.46	0
		(0.66, 2.67)	(0.08, 0.35)*	(2.53, 14.50)*	(0.10, 0.38)*	(0.10, 0.42)*	-
Evening	BH	0	-	-	-	-	-
		-	-	-	-	-	-
	CC	2.70 ± 1.66	0	-	-	-	-
		(1.02, 7.59)*	-	-	-	-	-
	CT	0.89 ± 1.77	0.33 ± 1.65	0	-	-	-
		(0.29, 2.76)	(0.12, 0.86)*	-	-	-	-
	GS	6.64 ± 1.61	2.46 ± 1.47	7.44 ± 1.60	0	-	-
		(2.72, 17.68)*	(1.16, 5.31)*	(3.08, 19.70)*	-	-	-
	RC	3.57 ± 1.63	1.32 ± 1.50	4.00 ± 1.62	0.54 ± 1.43	0	-
		(1.42, 9.70)*	(0.60, 2.94)	(1.60, 10.81)*	(0.26, 1.08)	-	-
	TH	1.46 ± 1.63	0.54 ± 1.51	1.64 ± 1.63	0.22 ± 1.44	0.41 ± 1.47	0
		(0.57, 3.98)	(0.24, 1.21)	(0.65, 4.43)	(0.11, 0.45)*	(0.19, 0.87)*	-

Table cont’d.

		Reference treatment					
Species	Treatment	BH	CC	CT	GS	RC	TH
Mexican free-tailed	BH	0	-	-	-	-	-
		-	-	-	-	-	-
	CC	4.34 ± 1.49	0	-	-	-	-
		(2.01, 9.77)*	-	-	-	-	-
	CT	3.30 ± 1.46	0.76 ± 1.54	0	-	-	-
		(1.60, 7.02)*	(0.32, 1.76)	-	-	-	-
	GS	1.94 ± 1.40	0.45 ± 1.49	0.59 ± 1.45	0	-	-
		(1.003, 3.80)*	(0.20, 0.96)*	(0.28, 1.21)	-	-	-
	RC	1.25 ± 1.41	0.29 ± 1.49	0.38 ± 1.46	0.64 ± 1.40	0	-
		(0.64, 2.45)	(0.13, 0.62)*	(0.18, 0.79)*	(0.33, 1.25)	-	-
	TH	1.20 ± 1.42	0.28 ± 1.50	0.36 ± 1.47	0.62 ± 1.41	0.96 ± 1.42	0
		(0.60, 2.38)	(0.12, 0.60)*	(0.17, 0.76)*	(0.31, 1.21)	(0.48, 1.90)	-
Hoary	BH	0	-	-	-	-	-
		-	-	-	-	-	-
	CC	2.41 ± 2.36	0	-	-	-	-
		(0.51, 17.51)	-	-	-	-	-
	CT	0.74 ± 2.92	0.31 ± 2.33	0	-	-	-
		(0.08, 6.93)	(0.04, 1.40)	-	-	-	-
	GS	0.92 ± 2.68	0.38 ± 2.09	1.24 ± 2.65	0	-	-
		(0.13, 7.77)	(0.08, 1.49)	(0.18, 10.31)	-	-	-
	RC	1.96 ± 2.35	0.81 ± 1.73	2.64 ± 2.32	2.13 ± 2.07	0	-
		(0.42, 14.16)	(0.27, 2.41)	(0.59, 18.78)	(0.55, 10.54)	-	-
	TH	1.17 ± 2.70	0.48 ± 2.10	1.58 ± 2.67	1.27 ± 2.42	0.60 ± 2.09	0
		(0.17, 9.98)	(0.10, 1.93)	(0.23, 13.25)	(0.21, 7.74)	(0.12, 2.35)	-

The 2020 non-breeding period NMDS converged with 2 dimensions after 20 tries with a stress value of 0.13. I excluded big brown bats from the 2020 non-breeding period NMDS analysis because this species was only active at 3 points. Additionally, I excluded evening bats from the 2020 non-breeding period NMDS analysis because this species was not detected during this sampling period. I found that, during the 2020 non-breeding period, shrub cover and DBH were significantly correlated ($P < 0.10$) with the ordination and explained 38% and 27% of variation in the proportional activity data, respectively (Table 2.7.). In addition, I found that the acoustic points did not group based on treatment during this sampling period, suggesting that activity was not explained by treatment (Fig. 2.6.). I also found that tricolored bats, Mexican free-tailed bats, hoary bats, and eastern red/Seminole bats were positively associated with DBH and negatively associated with percent shrub cover (Fig. 2.6.). Last, I found that *Myotis* species were not associated with any environmental characteristics but were grouped with two thinned points and one group selection harvest point (Fig. 2.6.).

The 2020 breeding period NMDS output a best solution after 100 tries with 2 dimensions and a stress value of 0.18. During the 2020 breeding period, I found that snag density, the proportion of forest >30 years old (1 km), DBH, and variation of NDVI (50 m) were significantly correlated ($P < 0.10$) with the ordination and explained 16–28% of variation in the proportional activity data (Table 2.7.). In addition, I found that tricolored bats, evening bats, and eastern red/Seminole bats were associated with each other and with group selection harvest points (Fig. 2.6.). Further, I found that these species were positively associated with the variation in NDVI (50 m) and negatively associated with the proportion of forest >30 years old (1 km) and snag density (Fig. 2.6.). I also found that *Myotis* species were positively associated with DBH but were not strongly associated with any specific treatments (Fig. 2.6.). I found that Mexican free-tailed bats and hoary bats were negatively associated with DBH, and that Mexican free-tailed bats were associated with thinned points (Fig. 2.6.). Last, I found that big brown bats were not associated with any treatments or environmental characteristics (Fig. 2.6.).

The 2021 non-breeding period NMDS output a best solution after 100 tries with 3 dimensions and a stress value of 0.15. I excluded evening bats from the 2021 non-breeding period NMDS analysis because this species was not present during this sampling period. During the 2021 non-breeding period, I found that snag density and distance to an incorporated area were significantly correlated ($P < 0.10$) with the ordination and explained 23% and 24% of variation in the proportional activity data, respectively (Table 2.7.). In addition, I found that clearcut, red-cockaded woodpecker habitat, and bottomland hardwood points were loosely associated with one another, suggesting that they experienced similar activity patterns (Fig. 2.7.). I found that eastern red/Seminole, Mexican free-tailed, *Myotis* species, and big brown bats were positively associated with distance to an incorporated area and negatively associated with snag density (Fig. 2.7.). In addition, I found that tricolored bats were positively associated with snag density and negatively associated with the distance to an incorporated area (Fig. 2.7.). I also found that tricolored bats were associated with clearcut and thinned points, eastern red/Seminole bats were associated with group selection harvest and bottomland hardwood points, and Mexican free-tailed bats were associated with red-cockaded woodpecker habitat points (Fig. 2.7.). In addition, I found that *Myotis* species and big brown bats were associated with one another, suggesting that they were active in the same areas (Fig. 2.7.). Further, I found that these two species were associated with red-cockaded woodpecker habitat points (Fig. 2.7.). Last, I found that hoary bats were not strongly associated with any treatments or environmental characteristics (Fig. 2.7.).

The 2021 breeding period NMDS converged with 2 dimensions after 20 tries with a stress value of 0.18. During the 2021 breeding period, I found that snag density, basal area, total edge (1 km), and variance in NDVI (50 m) were significantly correlated ($P < 0.10$) with the ordination and explained 16–35% of variation in the proportional activity data (Table 2.7.). In addition, I found that points were loosely associated with one another by treatment, suggesting that they explained similar amounts of activity (Fig. 2.7.). I found that evening, *Myotis* species, eastern red/Seminole, and tricolored bats were positively associated with the variance in NDVI (50 m) and negatively associated with snag density and basal area (Fig. 2.7.). In addition, I found that big brown bats were positively associated with total edge (1 km) (Fig. 2.7.). I also found that evening bats were associated with group selection harvest and clearcut points. Additionally, I found that *Myotis* species were associated with clearcut and group selection harvest points and one red-cockaded woodpecker habitat point, and that Eastern red/Seminole bats were associated with clearcut points (Fig. 2.7.). Last, I found that hoary bats, big brown bats, and tricolored bats were associated with red-cockaded woodpecker habitat points (Fig. 2.7.).

Table 2.7. Significant environmental variables in NMDS analyses ($P < 0.10$) for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. I found that all environmental variables had $r^2 \geq 0.15$ (McCune and Grace 2002).

Variable	Axis 1	Axis 2	Axis 3	r^2
Non-breeding 2020				
Shrub cover	0.60	0.80	-	0.38
DBH	−0.32	−0.95	-	0.27
Breeding 2020				
Variance in NDVI (50 m)	−0.76	0.65	-	0.28
DBH	0.73	0.69	-	0.22
Proportion forest >30 years (1 km)	0.38	−0.93	-	0.17
Snag density	0.11	−0.99	-	0.16
Non-breeding 2021				
Distance to developed	−0.65	0.76	−0.04	0.24
Snag density	0.76	−0.43	0.49	0.23
Breeding 2021				
Total edge (1 km)	0.87	0.50	-	0.25
Basal area	−0.80	0.60	-	0.21
Variance in NDVI (50 m)	0.72	−0.69	-	0.18
Snag density	−0.64	0.77	-	0.16

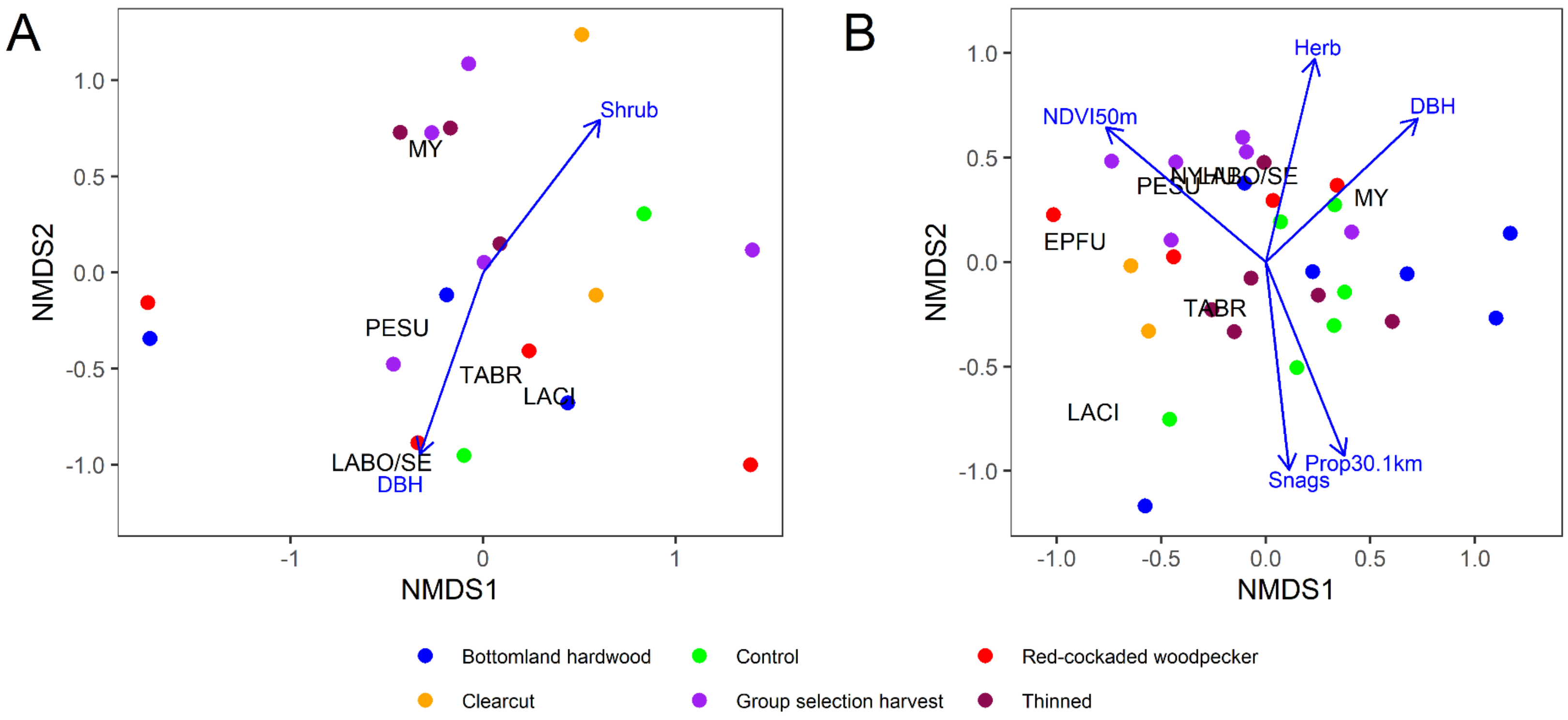

Figure 2.6. NMDS for the 2020 non-breeding (A) and breeding (B) periods for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. Bat species include the eastern red/Seminole bat group (LABO/SE), Mexican free-tailed bats (TABR), hoary bats (LACI), tricolored bats (PESU), the *Myotis* species group (MY), big brown bats (EPFU), and evening bats (NYHU). I included variables that were significantly correlated ($P < 0.10$) with the ordination, which included DBH and percent shrub cover (Shrub) during the 2020 non-breeding period and the variance in NDVI in 50 m (NDVI50m), percent herbaceous cover (Herb), DBH, snag density (Snags), and the proportion of forest in 1 km (Prop30.1km) during the 2020 breeding period.

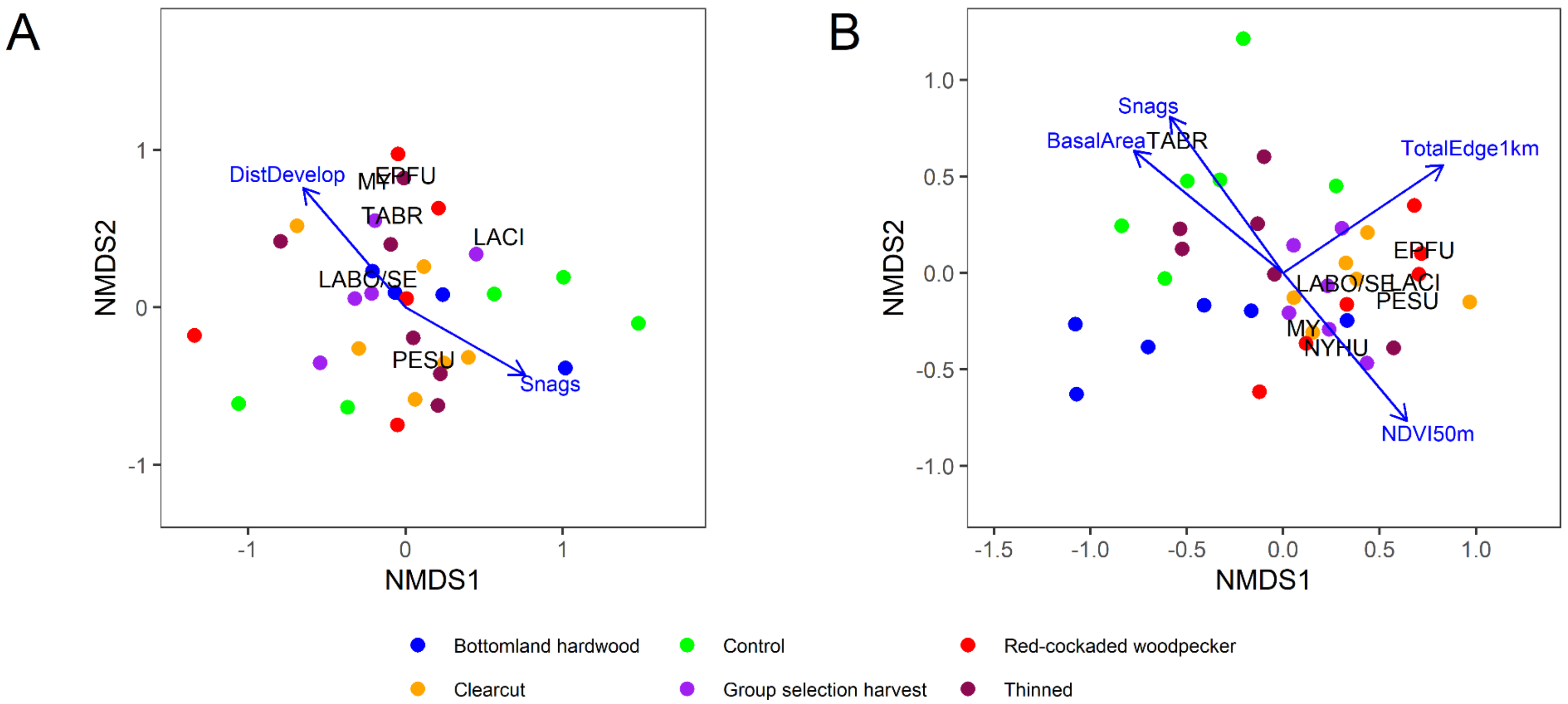

Figure 2.7. NMDS for the 2021 non-breeding (A) and breeding (B) periods for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. Bat species include the eastern red/Seminole bat group (LABO/SE), Mexican free-tailed bats (TABR), hoary bats (LACI), tricolored bats (PESU), the *Myotis* species group (MY), big brown bats (EPFU), and evening bats (NYHU). I included variables that were significantly correlated ($P < 0.10$) with the ordination, which included the distance to an incorporated area (DistDevelop) and snag density (Snags) during the 2021 non-breeding period and the variance in NDVI in 50 m (NDVI50m), the total edge in 1 km (TotalEdge1km), snag density (Snags), and basal area during the 2021 breeding period.

2.3.3. Foraging Activity

I found that the confidence intervals for all treatment comparisons included 1 for foraging activity during the non-breeding period (Table 2.8.). During the breeding period, I found that the confidence intervals for the clearcut, control, group selection harvest, and red-cockaded woodpecker treatments compared to the bottomland hardwood treatment did not include 1 for foraging activity (Table 2.8.). In addition, I found that the confidence intervals for control and thinned treatments compared to the clearcut treatment did not include 1 and that the group selection harvest, red-cockaded woodpecker, and thinned treatments compared to the control treatment did not include 1 during the breeding period (Table 2.8.). Last, I found that the thinned treatment compared to the group selection harvest and red-cockaded woodpecker treatments did not include 1 during the breeding period (Table 2.8.).

My results suggested that the odds of detecting high foraging activity in the clearcut, group selection harvest, and red-cockaded woodpecker treatments was 6–10 times the odds of detecting high foraging activity in the bottomland hardwood treatment (Table 2.8.). Further, my results suggested that the odds of detecting high foraging activity in the group selection harvest, red-cockaded woodpecker, and thinned treatments was 4–40 times the odds of detecting high foraging activity in the control treatment (Table 2.8.). In addition, my results suggested that the odds of detecting high foraging activity in the control treatment was 0.2 times the odds of detecting high foraging activity in the bottomland hardwood treatment (Table 2.8.). My results also suggested that the odds of detecting high foraging activity in the control and thinned treatments was 0.02 and 0.07 times the odds of detecting high foraging activity in the clearcut treatment, respectively (Table 2.8.). Last, my results suggested that the odds of detecting high foraging activity at the thinned treatment was 0.1 times the odds of detecting high foraging activity at the group selection harvest and red-cockaded woodpecker treatments, respectively (Table 2.8.).

Table 2.8. Foraging activity proportional odds model results for the non-breeding and breeding periods, including odds ratios ± SE and 95% confidence intervals, for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. Asterisks denote confidence intervals that do not include 1. Treatments are bottomland hardwood (BH), clearcut (CC), control (CT), group selection harvest (GS), red-cockaded woodpecker habitat (RC), and thinned (TH).

		Reference treatment					
Period	Treatment	BH	CC	CT	GS	RC	TH
Non-breeding	BH	0	-	-	-	-	-
		-	-	-	-	-	-
	CC	0.46 ± 3.49	0	-	-	-	-
		(0.02, 4.36)	-	-	-	-	-
	CT	0.46 ± 3.49	1.00 ± 4.73	0	-	-	-
		(0.02, 4.36)	(0.03, 30.60)	-	-	-	-
	GS	1.13 ± 2.34	2.44 ± 3.51	2.44 ± 3.51	0	-	-
		(0.21, 6.20)	(0.26, 55.29)	(0.26, 55.29)	-	-	-
	RC	1.08 ± 2.17	2.34 ± 3.33	2.34 ± 3.33	0.96 ± 2.18	0	-
		(0.24, 5.25)	(0.28, 50.17)	(0.28, 50.17)	(0.21, 4.70)	-	-
	TH	0.52 ± 2.68	1.12 ± 3.86	1.12 ± 3.86	0.46 ± 2.70	0.48 ± 2.53	0
		(0.06, 3.39)	(0.08, 27.91)	(0.08, 27.91)	(0.05, 3.04)	(0.60, 2.67)	-
Breeding	BH	0	-	-	-	-	-
		-	-	-	-	-	-
	CC	9.50 ± 1.56	0	-	-	-	-
		(4.03, 23.30)*	-	-	-	-	-
	CT	0.18 ± 1.78	0.02 ± 1.78	0	-	-	-
		(0.05, 0.51)*	(0.01, 0.05)*	-	-	-	-
	GS	6.93 ± 1.51	0.73 ± 1.47	39.56 ± 1.74	0	-	-
		(3.14, 15.77)*	(0.34, 1.54)	(14.44, 129.60)*	-	-	-
	RC	5.54 ± 1.53	0.58 ± 1.49	31.59 ± 1.75	0.80 ± 1.43	0	-
		(2.45, 12.89)*	(0.26, 1.27)	(11.30, 105.01)*	(0.39, 1.62)	-	-
	TH	0.67 ± 1.53	0.07 ± 1.53	3.85 ± 1.74	0.10 ± 1.47	0.12 ± 1.49	0
		(0.29, 1.55)	(0.03, 0.16)*	(1.37, 12.64)*	(0.04, 0.20)*	(0.05, 0.26)*	-

2.3.4. Occupancy

I detected too few Mexican free-tailed bats during the non-breeding period (n = 6) to model this species' predicted probability of occupancy. Similarly, I had too few non-detections ($n \leq 6$) of Mexican free-tailed bats, tricolored bats, and eastern red/Seminole bats to model these species' predicted probability of occupancy during the breeding period. Therefore, I modeled the predicted probability of occupancy for eastern red/Seminole bats, tricolored bats, big brown bats, *Myotis* species, and hoary bats during the non-breeding period and big brown bats, *Myotis* species, hoary bats, and evening bats during the breeding period. None of my global models had $\hat{c}$ >2, so I used AICc model selection procedures for the following analyses.

During the non-breeding period, I found that the models that best described eastern red/Seminole bat occupancy included the model for forest diversity in 1 km and the model including only year as a parameter (Table 2.9.). I found that the model weight for the forest diversity model was >2 times the weight of the year model (Table 2.9.). Additionally, I found that both the forest diversity and year (2021) parameters had confidence intervals that did not include 0 (Table 2.11.). My results suggested that, as forest diversity increased, the predicted probability of eastern red/Seminole bat occupancy decreased during the non-breeding period (Table 2.11.; Fig. 2.8.). Also, during the 2021 non-breeding period, I found that the predicted probability of eastern red/Seminole bat occupancy increased compared to the 2020 non-breeding period (Table 2.11.).

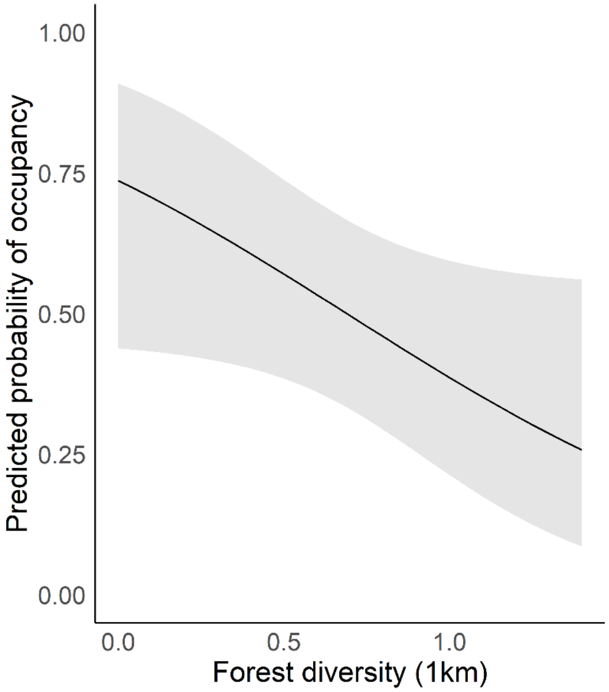

Figure 2.8. The predicted probability of eastern red/Seminole bat occupancy in relation to forest diversity (1 km) during the 2021 non-breeding period for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

During the non-breeding period, I found that there were five models in the top model set for tricolored bats (Table 2.9.). In addition, I found that the model including total edge in 1 km had the highest AICc weight but was similar in weight to the next top model (Table 2.9.). Further, I found that only year (2021) had a 95% confidence interval that did not include 0 (Table 2.11.). My results showed that during the 2021 non-breeding period, the predicted probability of tricolored bat occupancy increased compared to the 2020 non-breeding period (Table 2.11.).

During the non-breeding period, I found that the models that best described big brown bat occupancy included percent shrub cover, treatment, total edge in 1 km, and basal area + percent shrub cover (Table 2.9.). In contrast, I found that the only model in the top model set during the breeding period was forest diversity in 1 km (Table 2.10.). I found that the AICc weight was similar for both the percent shrub cover and treatment models, whereas the total edge and basal area + percent shrub cover models had less than half the weight of the top models during this period (Table 2.9.). I found that the forest diversity model had high AICc weight during the breeding period, suggesting that it was able to explain big brown bat occupancy much better than other models during this period (Table 2.10.). During the non-breeding period, I found that the 95% confidence intervals for percent shrub cover, the red-cockaded woodpecker treatment, and total edge did not include 0 (Table 2.11.). Additionally, I found that the 95% confidence interval for forest diversity did not include 0 during the breeding period (Table 2.12.).

My results suggest that, during the non-breeding period, as percent shrub cover increased, the predicted probability of big brown bat occupancy decreased (Table 2.11.). However, I found that the main effect of shrub cover on the predicted probability of big brown bat occupancy occurs under 20% shrub cover with a large confidence interval, which suggests that that this variable does not have a large effect on big brown bat occupancy (Fig. 2.9.). Additionally, I found that the predicted probability of big brown bat occupancy increased in the red-cockaded woodpecker treatment compared to the bottomland hardwood treatment during the non-breeding period (Table 2.11.). I also found that the predicted probability of big brown bat occupancy increased with increasing total edge (Table 2.11.; Fig. 2.9.). Last, during the breeding period, I found that the predicted probability of big brown bat occupancy decreased with increasing forest diversity (Table 2.12.; Fig. 2.9.).

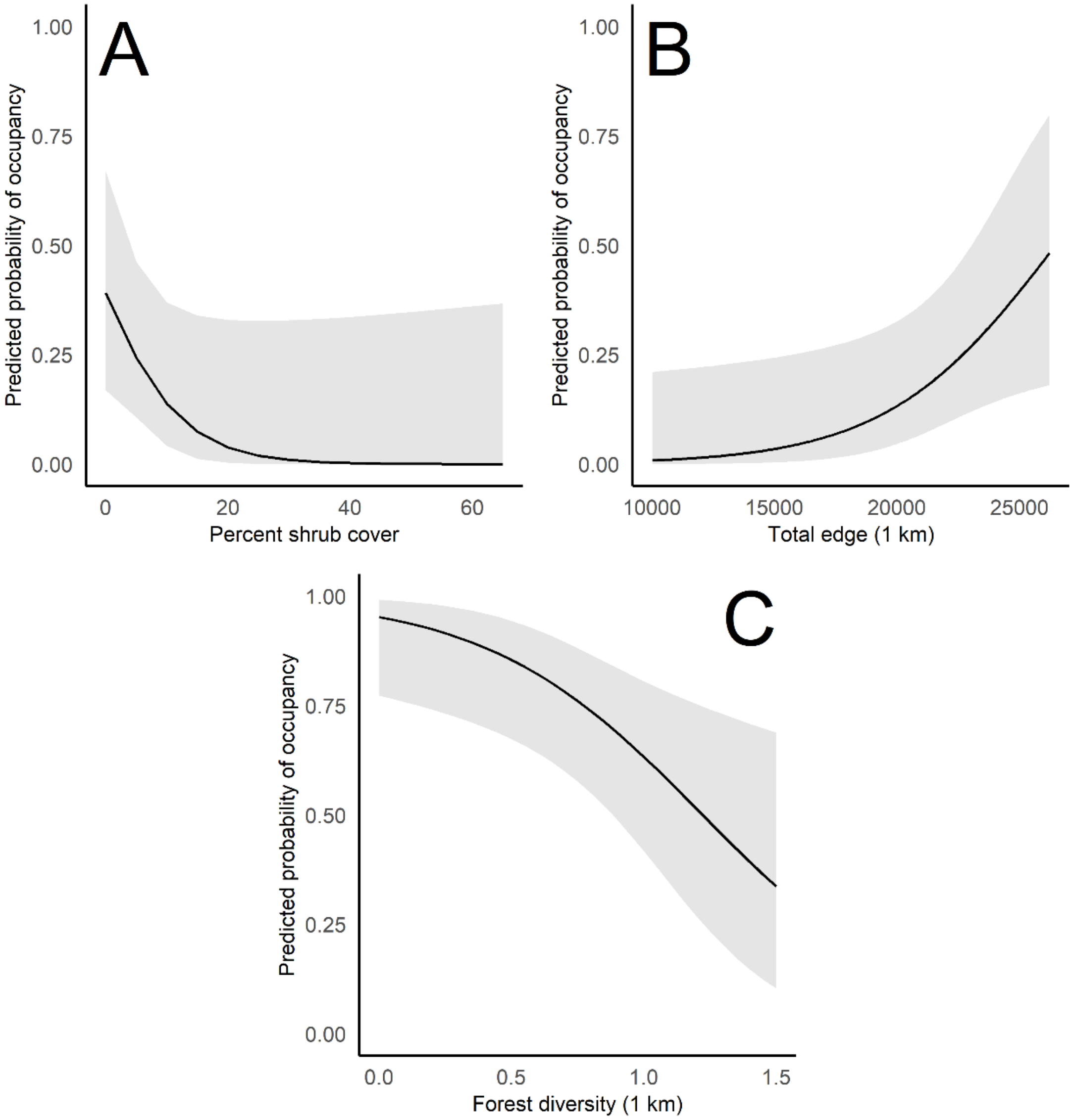

Figure 2.9. The predicted probability of big brown bat occupancy in relation to percent shrub cover (A) and total edge (1 km) (B) during the 2021 non-breeding period and forest diversity (1 km) (C) during the 2021 breeding period for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

For the *Myotis* species group, I found that there were two models in the top model set during the non-breeding period (Table 2.9.) and six models in the top model set during the breeding period (Table 2.10.). During the non-breeding period, I found that the weight of the model including the proportion of forest aged >30 years was more than twice the weight of the next top model (snag density + DBH; Table 2.9). In contrast, I found that the top two models for the breeding period had similar AICc weights, with the second-top model being the constant (null) model (Table 2.10.). I found that the 95% confidence intervals for the parameters proportion of forest aged >30 years and snag density did not include 0 during the non-breeding period (Table 2.11.), whereas all parameters included 0 during the breeding period (Table 2.12.). My results suggest that none of the models were able to explain *Myotis* species occupancy during the breeding period. In contrast, during the non-breeding period, I found that the predicted probability of *Myotis* species occupancy decreased with both increasing proportion of forest aged >30 years and increasing snag density (Table 2.11.; Fig. 2.10.).

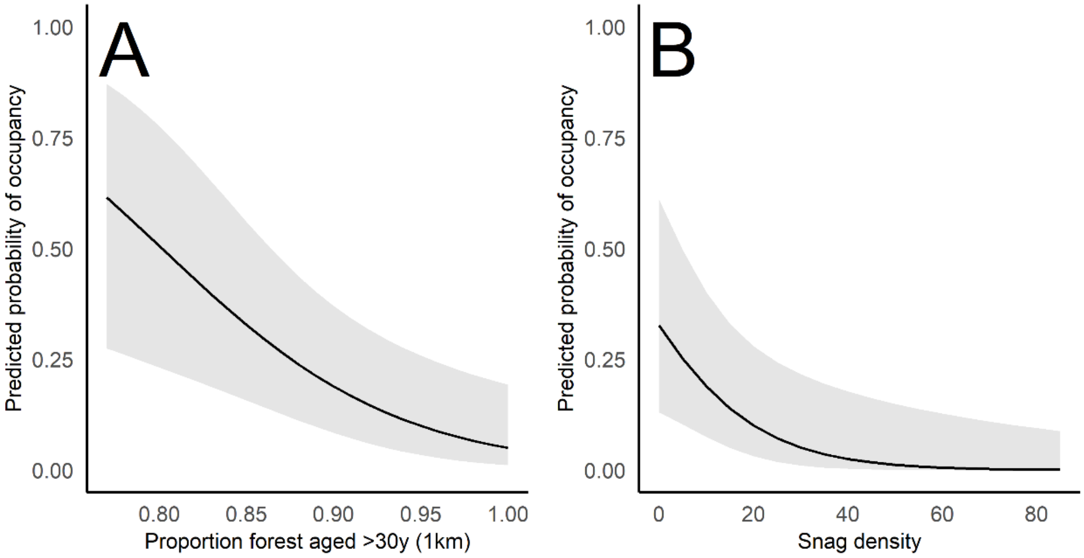

Figure 2.10. The predicted probability of *Myotis* species occupancy in relation to the proportion of forest >30 years old (1 km) (A) and snag density (B) during the 2021 non-breeding period for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

During the non-breeding period, I found that there were two models in the top model set for hoary bats, which both included the parameter for DBH (Table 2.9.). Additionally, I found that the top model (DBH) had more than twice the AICc weight compared to the next top model, suggesting that DBH was controlling the weight of the second model (Table 2.9.). I found that the 95% confidence interval for DBH did not include 0, which suggests that the predicted probability of hoary bat occupancy increased with increasing DBH during the non-breeding period (Table 2.11.); however, I found that the confidence interval was large above 30 cm DBH, suggesting that this variable is unable to accurately predict hoary bat occupancy above this value (Fig. 2.11.). During the breeding period, I found that the top two models had the same AICc weight and were similar in weight to the year model (Table 2.10.). I found that the only parameter with a 95% confidence interval that did not include 0 during the breeding period was year (2021), which suggested that the predicted probability of hoary bat occupancy was lower during the 2021 breeding period compared to the 2020 breeding period (Table 2.12.).

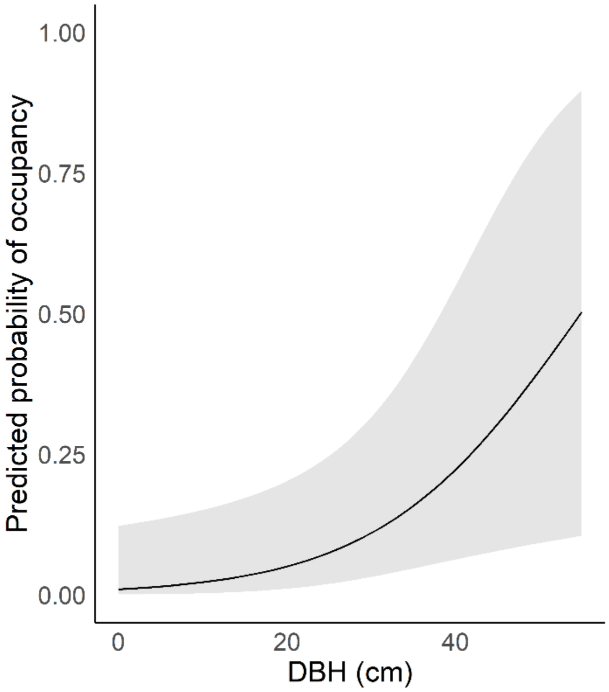

Figure 2.11. The predicted probability of hoary bat occupancy in relation to DBH during the 2021 non-breeding period for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

During the breeding period, I found that there were three models in the top model set for evening bats, with two of the models including distance to water as a parameter (Table 2.10.). I found that the top-most model (distance to water) had an AICc weight that was almost twice the weight of the second model and was over twice the weight of the third model (Table 2.10.). I found that the 95% confidence intervals for distance to water and the proportion of forest aged >30 years did not include 0 (Table 2.12.). My results suggest that the predicted probability of evening bat occupancy increased with increasing distance to water, whereas the predicted probability of evening bat occupancy decreased with increasing proportion of forest aged >30 years (Table 2.12.; Fig. 2.12.).

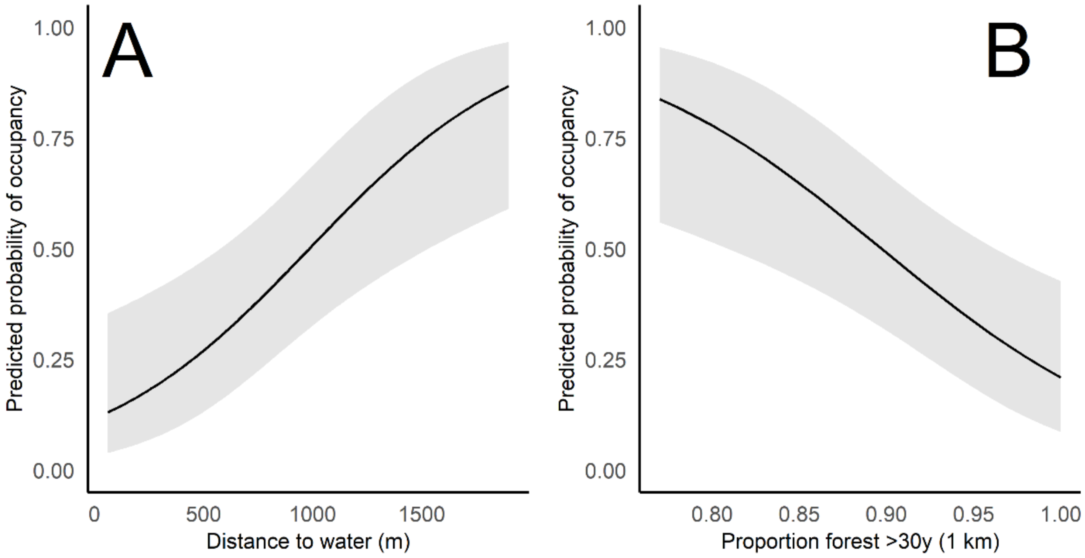

Figure 2.12. The predicted probability of evening bat occupancy in relation to the distance to water (A) and the proportion of forest >30 years old (1 km) (B) during the 2021 breeding period for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

Table 2.9. The top model set of occupancy models (ΔAICc <2) for species and species groups during the non-breeding period for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. I created these models using generalized linear modeling with a binomial distribution and included year as a parameter in each model.

Model	K	AICc	ΔAICc	AICc Wt.	Cumul. Wt.	LL
Eastern red/Seminole						
Forest diversity (1 km)	3	75.86	0.00	0.23	0.23	−34.74
Year	2	77.66	1.80	0.10	0.33	−36.73
Tricolored						
Total edge (1 km)	3	67.47	0.00	0.15	0.15	−30.54
Year	2	68.01	0.54	0.12	0.27	−31.91
DBH	3	69.12	1.65	0.07	0.34	−31.37
Distance to developed	3	69.30	1.82	0.06	0.40	−31.45
Percent shrub cover	3	69.32	1.84	0.06	0.46	−31.46
Big brown						
Percent shrub cover	3	39.17	0.00	0.26	0.26	−16.39
Treatment	7	39.29	0.12	0.25	0.51	−11.68
Total edge (1km)	3	40.85	1.69	0.11	0.62	−17.23
Basal area + Percent shrub cover	4	41.13	1.97	0.10	0.72	−16.24
Myotis species						
Proportion forest >30 years (1 km)	3	66.06	0.00	0.45	0.45	−29.83
Snag density + DBH	4	68.05	1.99	0.17	0.62	−29.70
Hoary						
DBH	3	48.06	0.00	0.33	0.33	−20.84
Variance in NDVI (1 km) + DBH	4	49.50	1.43	0.16	0.48	−20.42

Table 2.10. The top model set of occupancy models (ΔAICc <2) for species and species groups during the breeding period for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. I created these models using generalized linear modeling with a binomial distribution and included year as a parameter in each model.

Model	K	AICc	ΔAICc	AICc Wt.	Cumul. Wt.	LL
Big brown						
Forest diversity (1 km)	3	68.24	0.00	0.41	0.41	−30.93
Myotis species						
Basal area	3	72.10	0.00	0.15	0.15	−32.86
Constant	1	72.81	0.71	0.11	0.26	−35.37
Year	2	73.28	1.18	0.08	0.34	−34.55
Forest diversity (1 km)	3	73.57	1.47	0.07	0.42	−33.59
Total edge (1 km)	3	73.86	1.76	0.06	0.48	−33.74
Snag density	3	74.00	1.90	0.06	0.54	−33.80
Evening						
Distance to water	3	84.48	0.00	0.49	0.49	−39.05
Snag density + distance to water	4	85.75	1.27	0.26	0.74	−38.55
Proportion forest >30 years (1 km)	3	86.15	1.67	0.21	0.95	−39.88
Hoary						
Variance in NDVI (1 km)	3	42.92	0.00	0.14	0.14	−18.27
Total edge (1 km)	3	42.98	0.06	0.14	0.28	−18.30
Year	2	43.35	0.43	0.11	0.39	−19.58
Variance in NDVI (50 m)	3	44.42	1.50	0.07	0.45	−19.02
Percent herbaceous	3	44.55	1.73	0.06	0.51	−19.14
Variance in NDVI (1 km) + DBH	4	44.69	1.77	0.06	0.57	−18.02

Table 2.11. Parameter estimates for occupancy models in the top model set (ΔAICc <2) during the non-breeding period for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. A 95% confidence interval that includes 0 suggests that the parameter is uninformative.

Model	Parameter	Estimate	Lower (2.5%)	Upper (97.5%)
Eastern red/Seminole				
Forest diversity (1 km)	Intercept	−2.00	−3.27	−1.01
	Forest diversity (1 km)	−0.60	−1.26	−0.01
	Year 2021	1.98	0.77	3.42
Year	Intercept	−1.87	−3.09	−0.93
	Year 2021	1.87	0.71	3.24
Tricolored				
Total edge (1 km)	Intercept	−2.35	−3.83	−1.26
	Total edge (1 km)	−0.53	−1.21	0.10
	Year 2021	1.52	0.18	3.13
Year	Intercept	−2.20	−3.63	−1.16
	Year 2021	1.38	0.08	2.95
DBH	Intercept	−2.34	−3.81	−1.25
	DBH	0.33	−0.29	1.01
	Year 2021	1.59	0.23	3.23
Distance to developed	Intercept	−2.26	−3.71	−1.20
	Distance to developed	0.31	−0.31	1.02
	Year 2021	1.44	0.13	3.02
Percent shrub cover	Intercept	−2.16	−3.60	−1.11
	Percent shrub cover	−0.31	−1.02	0.33
	Year 2021	1.26	−0.06	2.85

Table cont’d.

Model	Parameter	Estimate	Lower (2.5%)	Upper (97.5%)
Big brown				
Percent shrub cover	Intercept	−4.11	−7.65	−2.15
	Percent shrub cover	−1.99	−4.98	−0.50
	Year 2021	1.17	−0.88	4.20
Treatment	Intercept	−3.90	−7.77	−1.44
	Clearcut	0.10	−3.29	3.48
	Control	−18.11	NA	499.87
	Group selection	−18.11	NA	499.87
	RCW	2.58	0.29	5.83
	Thinned	−18.11	NA	499.87
	Year 2021	2.15	−0.12	5.40
Total edge (1 km)	Intercept	−4.00	−7.11	−2.17
	Total edge (1 km)	1.37	0.30	2.86
	Year 2021	1.78	−0.17	4.79
Basal area + percent shrub cover	Intercept	−3.95	−7.47	−2.04
	Basal area	−0.31	−1.43	0.84
	Percent shrub cover	−1.75	−4.84	−0.29
	Year 2021	1.05	−1.08	4.11
Myotis species				
Proportion forest >30 years (1 km)	Intercept	−1.23	−2.26	−0.35
	Proportion forest >30 years (1 km)	−1.04	−1.81	−0.39
	Year 2021	−0.50	−1.81	0.76
Snag density + DBH	Intercept	−0.98	−2.01	−0.09
	DBH	−0.63	−1.44	0.07
	Snag density	−1.29	−2.44	−0.39
	Year 2021	−1.21	−2.84	0.23

Table cont'd.

Model	Parameter	Estimate	Lower (2.5%)	Upper (97.5%)
Hoary				
DBH	Intercept	−2.18	−3.52	−1.12
	DBH	1.06	0.21	2.06
	Year 2021	−0.42	−2.18	1.20
Variance in NDVI (1 km) + DBH	Intercept	−2.11	−3.44	−1.03
	Variance in NDVI (1 km)	−0.54	−1.99	0.54
	DBH	1.29	0.30	2.55
	Year 2021	−0.85	−2.88	1.01

Table 2.12. Parameter estimates for occupancy models in the top model set during the breeding period for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. A 95% confidence interval that includes 0 suggests that the parameter is uninformative.

Model	Parameter	Estimate	Lower (2.5%)	Upper (97.5%)
Big brown				
Forest diversity (1 km)	Intercept	1.70	0.75	2.87
	Forest diversity (1 km)	−0.99	−1.80	−0.31
	Year 2021	−0.42	−1.73	0.83
Myotis species				
Basal area	Intercept	−1.03	−1.94	−0.23
	Basal area	0.61	−0.04	1.37
	Year 2021	−0.59	−1.84	0.62
Constant	Intercept	−1.22	−1.83	−0.67
Year	Intercept	−0.85	−1.68	−0.10
	Year 2021	−0.76	−1.98	0.40
Forest diversity (1 km)	Intercept	−0.90	−1.76	−0.13
	Forest diversity (1 km)	0.42	−0.17	1.07
	Year 2021	−0.76	−2.00	0.42
Total edge (1 km)	Intercept	−0.85	−1.69	−0.08
	Total edge (1 km)	0.40	−0.21	1.06
	Year 2021	−0.84	−2.09	0.34
Snag density	Intercept	−0.90	−1.75	−0.13
	Snag density	0.34	−0.22	0.92
	Year 2021	−0.73	−1.96	0.45

Table cont’d.

Model	Parameter	Estimate	Lower (2.5%)	Upper (97.5%)
Evening				
Distance to water	Intercept	0.15	−0.65	0.96
	Distance to water	1.01	0.43	1.70
	Year 2021	−0.37	−1.47	0.70
Snag density + distance to water	Intercept	0.15	−0.66	0.98
	Snag density	−0.27	−0.88	0.26
	Distance to water	1.01	0.43	1.71
	Year 2021	−0.42	−1.53	0.67
Proportion forest >30 years (1 km)	Intercept	0.21	−0.58	1.02
	Proportion forest >30 years (1 km)	−0.91	−1.53	−0.36
	Year 2021	−0.48	−1.58	0.58
Hoary				
Variance in NDVI (1 km)	Intercept	−1.61	−2.67	−0.72
	Variance in NDVI (1 km)	0.67	−0.16	1.55
	Year 2021	−2.35	−5.92	−0.31
Total edge (1 km)	Intercept	−1.58	−2.76	−0.67
	Total edge (1 km)	−0.70	−1.68	0.15
	Year 2021	−2.15	−5.14	−0.27
Year	Intercept	−1.39	−2.38	−0.56
	Year 2021	−2.17	−5.14	−0.32
Variance in NDVI (50 m)	Intercept	−1.49	−2.55	−0.62
	Variance in NDVI (50 m)	0.36	−0.35	1.01
	Year 2021	−2.08	−5.06	−0.21
Percent herbaceous	Intercept	−1.43	−2.48	−0.58
	Percent herbaceous	−0.43	−1.43	0.44
	Year 2021	−2.21	−5.19	−0.34
Variance in NDVI (1 km) + DBH	Intercept	−1.61	−2.68	−0.72
	Variance in NDVI (1 km)	0.76	−0.10	1.73
	DBH	−0.37	−1.44	0.69
	Year 2021	−2.51	−6.23	−0.39

2.3.5. Roosting Habitat

I found 51 potential tree roosts. Of these, 23 were in bottomland hardwoods, 10 were in controls, 7 were in thinned sites, 6 were in group selection harvests, and 5 were in red-cockaded woodpecker habitat. I did not find any potential tree roosts in the few trees that were still standing at my clearcut sites. Additionally, a hurricane passed through my study area during August 2020, destroying 8 potential roosts in bottomland hardwoods, 3 in controls, 2 in thinned sites, one in a group selection harvest, and one in a red-cockaded woodpecker site. I found the most potential roosts in decay cavities of sweet gums (n = 17). The most common roost type that I found were decay cavities (n = 38), followed by both peeling bark (n = 6) and crevices (n = 6). I revisited potential roosts seven times during the 2020 breeding period, two times during the 2021 non-breeding period, and once during the 2021 breeding period to survey for bat presence. However, I did not detect any bats during my tree roost surveys.

I found 38 structures that I characterized as potential roosting sites within 2 km of my acoustic points. Of these, the most common structure types were double-T bridges made from sectioned cement tops and wooden beams (n = 21). I surveyed structures for bat presence once during the 2021 non-breeding period and once during the 2021 breeding period. I detected both tricolored bats and Rafinesque’s big-eared bats in structures during the 2021 non-breeding period, but only detected Rafinesque’s big-eared bats during the 2021 breeding period. I detected all but one bat in double-T bridges. I detected the remaining bat (a tricolored bat) in a circular culvert during the 2021 non-breeding period (Fig. A.2.). During the non-breeding period, I detected three Rafinesque’s big-eared bats and 25 tricolored bats. During the breeding period, I detected three Rafinesque’s big-eared bats. These detections were spread out throughout my study area during the non-breeding period (Fig. 2.13.). However, during the breeding period, I found one Rafinesque's big-eared bat near a group selection harvest and a bottomland hardwood, and I found the remaining two Rafinesque's big eared bats near two thinned sites and one bottomland hardwood site (Fig. 2.13.).

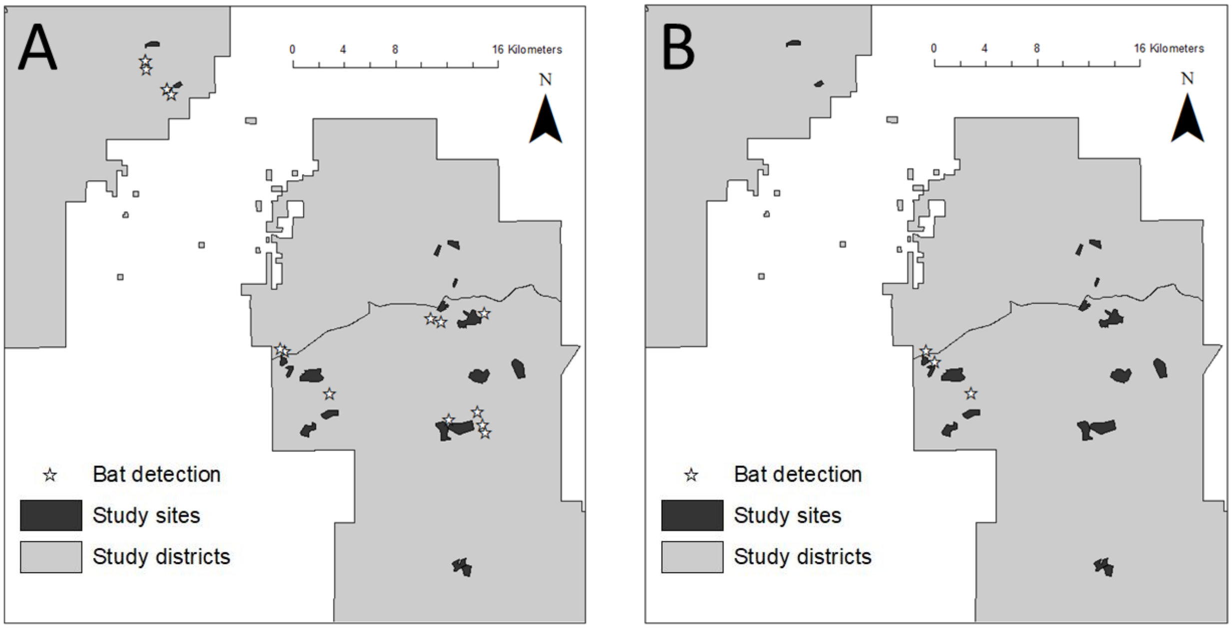

Figure 2.13. Detections of bats in manmade structures during the non-breeding (A) and breeding (B) periods of 2021 for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

2.4. Discussion

I found that species composition was similar between the non-breeding and breeding periods, but that I only detected evening bats during the breeding period. Further, I found that foraging group composition and species richness did not differ among treatments during any period. In addition, I found that group selection harvests, red-cockaded woodpecker habitat, and clearcuts promoted high bat activity and high foraging activity for multiple species, suggesting that these treatments provide important habitat for bats. I also found that different habitat variables influenced the predicted probability of occupancy for big brown bats and *Myotis* species between periods, and that variables associated with riparian habitat negatively affected the predicted probability of occupancy for several species. Last, I found that double-T bridges provided roosting habitat for tricolored bats and Rafinesque's big-eared bats during the non-breeding period, but that only a small number of bats utilized these bridges during the breeding period.

I expected that species composition would differ between the breeding and non-breeding periods due to migratory behavior and seasonal differences in habitat use. However, I detected eastern red/Seminole bats, tricolored bats, hoary bats, *Myotis* species, big brown bats, and Mexican free-tailed bats during both periods. In addition, during the breeding period, I detected eastern red/Seminole bats, tricolored bats, and Mexican free-tailed bats at almost all points, suggesting that they are ubiquitous on the landscape during this period. The only difference in species composition I found was during the breeding period, when I additionally detected evening bats. In contrast, Stevens et al. (2020) previously detected evening bats in the Kisatchie National Forest during the non-breeding period. This suggests that evening bats are present in the study area during the non-breeding period, but do not utilize the treatments I studied during this period. Evening bats are often found roosting in infrastructure (Mering and Chambers 2014), so it is possible that evening bats were using developed areas in the Kisatchie National Forest during the non-breeding period. Alternatively, it is possible that SonoBat misidentified evening bat echolocation calls during this period as different species or as background noise; however, with SonoBat's success in identifying evening bats during the breeding period, I believe that this is unlikely.

Contrary to my expectations, foraging group composition did not differ among treatments, as I detected all three groups at almost all treatments during both periods. I expected to detect more open space aerial foragers in open habitats, such as my clearcut and group selection harvest treatments, because they typically occur in open habitats like fields and large forest cuts (Menzel et al. 2002, Menzel et al. 2005). However, open space aerial foragers can occur in forests with increased understory vegetation structure (Menzel et al. 2002, Morris et al. 2010, Caldwell et al. 2019), and it is possible that I detected open space aerial foragers navigating above the canopy, which is better suited for their echolocation characteristics and morphology (Menzel et al. 2005). In contrast to my findings, I expected to detect more narrow space passive gleaning foragers (i.e., *Myotis* species) in the forest treatments with increased understory vegetation structure, such as the control and bottomland hardwood treatments, as these species are often negatively associated with open habitat (Morris et al. 2010, Jantzen and Fenton 2013). However, *Myotis* species can forage at forest edges (Jantzen and Fenton 2013, Ober et al. 2020), which treatments like group selection harvests and clearcuts provide. Further, my detections of *Myotis* species in clearcuts may be attributed to the relatively small size of clearcuts in my study (<50 ha) compared to many commercial forest clearcuts (>90 ha; Boston and Bettinger 2001). My species composition results show that treatment alone did not affect the species or foraging group composition in my study area, suggesting that all the bat species that I analyzed can occur in a variety of management practices in the forests of central Louisiana.

In line with my expectations, I found that bat activity was high at group selection harvests for several species. Group selection harvests provide edge habitat, which bats can use for foraging (Jantzen and Fenton 2013, Morris et al. 2010) and as conduits for navigation (Kalcounis-Rueppell et al. 2013). In particular, edge space aerial foragers like eastern red bats, Seminole bats, and tricolored bats have echolocation and body characteristics that make them better adapted for foraging and navigating in edge habitat (Denzinger and Schnitzler 2013), and I found that these species had higher activity at group selection harvests. Therefore, my results suggest that the habitat provided by group selection harvests is beneficial to multiple species of bats throughout the year, but particularly the edge space aerial foragers. Further, recommendations to increase wildlife abundance in forests often include the use of small (<1 ha) cuts like those used in group selection harvests, and management of these cuts include maintaining herbaceous cover and limiting growth of woody plants (Healy 1989, Bromley et al. 2009). These cuts, often termed “wildlife openings” (Healy 1989), and their associated edge habitat encourage the presence of game species like white-tailed deer (*Odocoileus virginianus*; Stewart et al. 2000, Massé and Côté 2012), wild turkey (*Meleagris gallopavo*; Pollentier et al. 2017), and American woodcock (*Scolopax minor*; Shartell 2021), as well as non-game species (e.g., early-successional birds; Campbell et al. 2012, Smetzer et al. 2014). Therefore, group selection harvests have the potential to increase overall biodiversity, particularly when managers maintain herbaceous forage and reduce woody vegetation inside of cuts.

In line with my expectations, I found that activity was high at the red-cockaded woodpecker treatment for multiple species. One of the red-cockaded woodpecker treatment sites in my study was a longleaf pine-grassland ecosystem, which are unique in their dependence on frequent fire regimes (Van Lear et al. 2005). Prescribed burns in these ecosystems can provide roosting opportunities for both cavity (Boyles and Aubrey 2006, Johnson et al. 2009) and foliage roosting bats (Jorge et al. 2021) and can decrease the density of trees and tall shrubs (Armitage and Ober 2012). The resulting open habitat often increases bat activity, including for open space aerial foragers like big brown bats (Armitage and Ober 2012, Braun de Torrez et al. 2018). The other two red-cockaded woodpecker treatment sites in my study were composed of loblolly but also had an open habitat structure; however, these sites were recently created and had no history of burning after their conversion to red-cockaded woodpecker habitat. Though the ground cover and snag composition of these sites differed from the burned longleaf red-cockaded woodpecker site, bats were also highly active at the loblolly red-cockaded woodpecker sites, suggesting that bat activity was driven by the open habitat structure rather than ground cover or snag densities. This is unsurprising, as bat activity is often primarily driven by the physical structure of a forest (Grindal and Brigham 1999, Ober and Hayes 2008). Thus, my results affirm that the openness of red-cockaded woodpecker habitat is beneficial to a variety of bat species.

I expected that bat activity would be lower at clearcuts compared to other treatments because they only provide homogenous open habitat. However, I found that activity was high at clearcuts for multiple species. The open habitat that characterizes clearcuts may increase activity by providing foraging habitat for less-maneuverable open space aerial foragers (e.g., big brown bats) and generalist edge space aerial foragers (e.g., eastern red bats; Grindal and Brigham 1999, Caldwell et al. 2019). Because bat activity in open areas is often related to the amount of edge habitat available (Jantzen and Fenton 2013), the smaller size of clearcuts in my study may have decreased the distance to edge habitat at any given point within the clearcut, which would explain this treatment's association with high activity from multiple foraging groups. Thus, clearcuts that are <50 ha may promote high bat activity for multiple species in central Louisiana, but larger clearcuts could potentially have negative effects on the activity of species that tend to occur away from homogenous open habitat (e.g., *Myotis* species).

From my occupancy analyses, I found that there were differences in habitat use between periods for multiple species, including big brown bats, *Myotis* species, and eastern red/Seminole bats. As demonstrated through my study and others, bats are less active during the non-breeding period but still maintain some activity in the United States (Boyles et al. 2006, Parker et al. 2020, Stevens et al. 2020). Excursions during the non-breeding period are often for foraging (Dunbar et al. 2007, Bernard et al. 2021) and hydrating (Ben-Hamo et al. 2013) on nights with warmer temperatures at dusk (Bender and Hartman 2015, Arndt and Lima 2020). Bats may selectively use areas with high insect abundances and water availability rather than habitats that align with their foraging group during the non-breeding period to conserve energy and optimize foraging success during these emergences (e.g., open habitats for open space aerial foragers; Shute et al. 2021), which could explain the differences in species' habitat use between periods in my study. Therefore, a variety of forest types are necessary to provide habitat for bats during the non-breeding and breeding periods in central Louisiana, and wildlife managers need to consider the differences in bat habitat use between periods to accurately manage for high bat diversity in the landscape.

I also found that habitat characteristics that described proximity to riparian habitat in my study area (i.e., increasing forest diversity in 1 km and proportion of forest >30 years in 1 km and decreasing distance to water) decreased the predicted probability of occupancy for eastern red/Seminole bats, big brown bats, *Myotis* species, and evening bats. In contrast to my findings, eastern red/Seminole bats are often associated with riparian areas and hardwood forests during the breeding period (Owen et al. 2004, Menzel et al. 2005, Morris et al. 2010). Similarly, big brown bats are sometimes associated with riparian habitat (Owen et al. 2004, Menzel et al. 2005) where their activity is often higher in the open space above the canopy (Menzel et al. 2005); however, Ketzler et al. (2018) found that big brown bats were not active in bottomland hardwoods in the Mississippi Alluvial Valley. Further, *Myotis* species often occur and are active in dense, old-growth forests with high snag densities due to their maneuverability and high-frequency calls (Ketzler et al. 2018) and are usually associated with bottomland hardwood forests, particularly for roosting activities (Carver and Ashley 2008, Stuemke et al. 2014). Last, though evening bats were associated with riparian habitat in South Carolina (Menzel et al. 2005), evening bat occupancy is often negatively associated with increased complexity of understory vegetation (Bender et al. 2015, Brooks et al. 2017, Bender et al. 2021). Due to the increased complexity of understory vegetation of the riparian bottomland hardwoods in my study, these species may primarily use the available managed pine habitat outside of these areas, which provide more open habitat and linear landscape features (e.g., edges). In support of this idea, researchers found that the primary prey of bats in our study area (i.e., Coleoptera and Lepidoptera) were abundant in bottomland hardwoods in the southeast during the breeding period (Ketzler et al. 2017, Weinkauf et al. 2018), suggesting that prey availability did not influence the use of bottomland hardwoods for these species. However, bottomland hardwoods still provide important habitat for bats in Louisiana, as evidenced by species occurrence and activity in these areas during my study.

I did not detect bats during my tree roost surveys, which is consistent with previous research describing the high search effort necessary to find occupied roost trees (Comer et al. 2014). However, I found that bottomland hardwoods and controls had the most potential tree roosts based on my search criteria, suggesting that these areas likely provide suitable roosting habitat for bats. Riparian habitat like bottomland hardwoods provide snags and cavity roosting opportunities, and some species primarily roost in bottomland hardwoods (e.g., Rafinesque’s big eared bats; Gooding and Langford 2004, Clement and Castleberry 2013). In addition, my controls were composed of a mix of old loblolly pine and hardwoods that provided cavities, crevices, and peeling bark that could be used by species like northern long-eared bats (Johnson et al. 2009, Rojas et al. 2017) and evening bats (Boyles and Robbins 2006). Therefore, although I did not detect any bats during my tree roost surveys, it is likely that both my bottomland hardwood and control treatments provided adequate roosting habitat. In addition, some bats in my study area (i.e., eastern red bats, Seminole bats, hoary bats, and tricolored bats) are considered foliage roosting, meaning that they roost in dense foliage rather than in cavities or under bark during at least a portion of the year (Kalcounis-Rüppell et al. 2005). Foliage roosting bats typically choose large DBH trees and roost high in the canopy (Kalcounis-Rüppell et al. 2005, Drake et al. 2020), so treatments with old, tall trees (e.g., controls, bottomland hardwoods, red-cockaded woodpecker habitat) may provide foliage roosting habitat that I did not investigate in this study.

During my infrastructure surveys, I detected tricolored bats and Rafinesque’s big-eared bats, which are both documented as utilizing infrastructure like culverts and bridges in Louisiana (Ferrara and Leberg 2005) and other areas (Wolters and Martin 2011, Meierhofer et al. 2019). However, I only detected tricolored bats in infrastructure during the non-breeding period, likely because these bats were moving into foliage roosts during the summer (Drake et al. 2020). In addition, the distribution of bats in infrastructure did not follow any treatment patterns during the non-breeding period, suggesting that these structures are beneficial during this period regardless of the forested habitat available in the surrounding landscape. Further, the design of the occupied structures provided crevices and cracks that bats used (Fig. A.3.). Therefore, structures with these characteristics can provide roosting habitat for bats, particularly during the colder non-breeding period when solitary roosting bats need protection from the elements. Last, though I detected few Rafinesque’s big-eared bats using structures as roosts, these detections show that this species was present in the forest and was therefore under-detected in my acoustic surveys due to their echolocation characteristics (i.e., low amplitude and low-frequency calls; Sherwin et al. 2000). My low acoustic detections of Rafinesque’s big-eared bats are in line with findings from other acoustic studies (Bender et al. 2015, Jorge et al. 2021), highlighting the importance of live capture and roost surveys to study this species.

Overall, my results suggest that a mosaic of forest management practices support the diversity and activity of bats in central Louisiana throughout the year. However, group selection harvests, red-cockaded woodpecker habitat, and clearcuts had the greatest influence on bat species activity and foraging activity in my study area, suggesting that these practices can benefit multiple species. Thus, my results suggest that bat conservation and management efforts in our region should incorporate forest management practices that increase edge habitat (e.g., group selection harvest) and decrease understory vegetation structure (e.g., thinning in red-cockaded woodpecker stands) while preserving natural roosts like snags. Further, my group selection harvest results suggest that managers can implement wildlife openings to benefit bats while also increasing the abundance of game species like white-tailed deer and wild turkey. Additionally, my research demonstrates the importance of studying bats throughout the year, as occupancy and activity were influenced by different treatments and habitat characteristics between the breeding and non-breeding periods. By studying bats during more than one season, researchers can provide more accurate information on bat-habitat relationships to inform species status assessments for declining species. In addition, researchers should conduct more live capture activities in central Louisiana, particularly targeting areas where *Myotis* species have high activity (e.g., group selection harvests, bottomland hardwoods) to distinguish habitat use between the southeastern bat and the Threatened northern long-eared bat. My results provide important information on the life histories, distributions, and habitat relationships of bat species that occur widely in the southeastern United States, which can help to inform wildlife management efforts to promote bat biodiversity and ecosystem services within southeastern forests.

Chapter 3. A Comparison of Bat Detections Recorded by Two Acoustic Monitors

3.1. Introduction

Wildlife survey and monitoring data are necessary to evaluate population trends, inform management efforts, and advise conservation policy decisions (Honrado et al. 2016). The methods we use to collect these data generally fall into two categories of observation: direct and indirect. Direct methods (e.g., trapping, manual counts) allow researchers to identify species distributions, estimate population sizes, and track individual movements, among others, but can be invasive (Romero 2004), biased (De Bondi et al. 2010, Greene et al. 2016) and difficult to implement based on logistics, funding, and expertise (Prosekov et al. 2020). Indirect methods (e.g., camera surveys, radar) have their own limitations (e.g., require certain assumptions, data may be difficult to interpret) (McDonald and Harris 1999). However, indirect methods can be an effective alternative or complement to direct observations for some species and research questions.

Passive acoustic monitoring, whereby autonomous recording units (ARUs; also called acoustic monitors) are deployed in the field to collect data, is an increasingly popular indirect method that biologists use to determine species presence, mean activity, species richness, and habitat use, among others (Gibb et al. 2019). ARUs cannot provide information on densities, home ranges, survival, or physiological responses (Gibb et al. 2019), but are powerful tools because they can be deployed to sample multiple locations at once (Gibb et al. 2019), can detect rare species with greater success than active techniques (De Bondi et al. 2010, Diggins et al. 2016; but see Rojas et al. 2019), and are typically less expensive to implement than direct methods (Diggins et al. 2016).

Passive monitoring with ARUs has been particularly useful for bat research given the difficulties of surveying for nocturnal species that emit inaudible vocalizations. Most insectivorous bats echolocate to navigate and find food sources at night (Metzner 1991), and each species' echolocation pulses have specific frequency, bandwidth, duration, and slope characteristics (Fenton and Bell 1981). ARUs that can detect and record ultrasound (>20 kHz) provide us with a non-invasive method to monitor bats using these "calls". Passive monitoring with ARUs has a high probability of detecting bats compared to active acoustic monitoring with ARUs (e.g., walking transects; Teets et al. 2019) and capture (e.g., mistnetting; O'Farrell and Gannon 1999). Furthermore, recent studies show that passive acoustic monitoring is effective in estimating bat species occupancy (Austin et al. 2020, Law et al. 2020) and activity (Bender and Hartman 2015, Braun de Torrez et al. 2018), and the information gained from such studies can identify habitat characteristics that promote bat abundance and species diversity. After acoustic data are collected, researchers can use automated classification software (e.g., SonoBat [SonoBat, Arcata, California, U.S.], Kaleidoscope Pro [Wildlife Acoustics, Inc., Maynard, Massachusetts, U.S.]) that identify bat calls to species using reference libraries to process the large collections of acoustic files that are generated by passive monitoring techniques.

In terms of audio specifications, ARUs designed to record bat calls can typically filter noise below a user-specified amplitude threshold and be set to record when the monitors detect sounds above a user-specified frequency threshold. Zero-crossing ARUs, which record a single frequency when the amplitude of a call crosses a certain threshold (determined by the signal-to-noise ratio during the recording), do not record the entire spectrum of amplitude (Limpens and McCracken 2004), and, thus, require less storage space compared to full-spectrum ARUs that record the entire spectrum of sound (i.e., all frequencies and amplitudes). However, zero-crossing files do not provide as much information about call structure (Britzke et al. 2013). The limited information contained in zero-crossing files can result in less accurate species identification, missing data for low-amplitude calls, and the absence of other important information like feeding buzzes and call harmonics (Fenton et al. 2001). Most zero-crossing and full spectrum ARUs are designed specifically for passive surveys: they can record bat calls for several weeks or more at a time because they have limited power requirements, can connect to external power sources (e.g., larger batteries, solar), and are weather-resistant. Researchers can choose whether they want to use directional or omnidirectional microphones; directional microphones typically record bats further away but in a limited area, and omnidirectional microphones record bats in a larger area but at a shorter distance (Limpens and McCracken 2004).

Though ARUs are useful in estimating bat species occurrence and activity and allow for increased sampling effort compared to direct methods, there are limitations to each device. ARUs that record ultrasound exceed $500 USD per unit. As such, funding can limit the use of acoustic monitors for large-scale bat research. In addition, microphone sensitivities and detection algorithms differ across ARUs; some microphones can record bats at further distances than others when deployed in the same conditions (Adams et al. 2012). Also, while automatic classifiers can reduce the time necessary to analyze call files, many study designs require trained professionals to verify classified files (Rydell et al. 2017). With the large number of files recorded by some ARUs based on their sensitivity and triggering thresholds, this process can limit the number of nights that can be surveyed and, thus, increase the possibility of missed species and activity patterns. Last, the battery life of these devices varies based on their recording capabilities and processing power, so some ARUs require more maintenance than others.

The recently developed AudioMoth (Open Acoustic Devices 2020*a*) may be a more accessible ARU for large-scale passive monitoring projects. Released in 2017, the AudioMoth is a small (58 x 48 x 15 mm) and inexpensive (~$50) full-spectrum acoustic monitor with a built-in omnidirectional microphone that is customizable and can record sounds up to 192 kHz (Hill et al. 2019). In contrast to many other ARUs, the base AudioMoth does not filter noise and records cyclically during programmed time periods, though a firmware update in 2020 added amplitude and triggering thresholds (Open Acoustic Devices 2020*b*). Researchers and hobbyists have purchased >22,000 AudioMoths since 2017 (GroupGets, LLC 2021) and have used AudioMoths to study cicadas (superfamily Cicadoidea; Hill et al. 2018), gunshots (Hill et al. 2018), gray wolves (*Canis lupus*; Barber-Meyer et al. 2020), and anurans (order Anura; Lapp et al. 2021). In addition, researchers have used AudioMoths to investigate bat species occurrence (Katunzi et al. 2021), bat call plasticity (Montauban et al. 2021), and cave bat emergences (Revilla-Martín et al. 2020). The abovementioned studies utilized automated classification software to analyze their data. However, when I was investigating whether I could use AudioMoths for my own bat research, I could not find any peer-reviewed studies that compared classification results obtained using files recorded by AudioMoths with any other ARUs that record ultrasound.

Variation in ARU performance could influence the results of bat research, inventory, and monitoring programs, and the degree to which studies can be compared. Thus, understanding potential variation across monitors is critically important and should be considered when researchers select an ARU for their projects. My goal was to compare the number of bat call files, the number of echolocation pulses, and the number of species recorded by AudioMoths and full-spectrum SM4BAT monitors (with omnidirectional SMM-U2 microphones; Wildlife Acoustics, Inc., Maynard, Massachusetts, USA) and identified using automated classification software. I expected that the automated classification software would identify a similar number of species from call files recorded by AudioMoths and SM4BATs. However, during a pilot study, I observed that AudioMoths recorded more background noise than my SM4BATs because my AudioMoths didn’t have amplitude and frequency filters, which seemed to decrease the number of files that I could automatically classify. As such, I expected that the automated classification software would identify fewer bat call files and echolocation pulses from recordings made by AudioMoths compared to SM4BATs.

3.2. Study Area and Sampling Periods

I conducted my research in Louisiana, U.S. at the Catahoula and Winn Ranger Districts in the Kisatchie National Forest. I deployed pairs of AudioMoth and SM4BAT monitors at 18 sites 10–140 ha in size, representing 6 treatments. My study sites included three sites that were clearcut within five years from the start of my project (hereafter clearcut treatment); three sites composed of thinned loblolly pine (hereafter thinned treatment); three sites composed of group selection harvested loblolly pine (hereafter group selection harvest treatment); three sites composed of either loblolly or longleaf pine that were managed for red-cockaded woodpeckers (*Leuconotopicus borealis*) (hereafter red-cockaded woodpecker treatment); three sites that were composed of mature loblolly pine >40 years old that were not managed with thinning, burning, or herbicides for ≥5 years (hereafter control treatment); and three sites composed of bottomland hardwood forests ≥40 years old that were characterized by intermittent flooding, and predominated by oaks (*Quercus* spp.), gums (*Nyssa* spp.), and bald cypress (*Taxodium distichum*) (hereafter bottomland hardwood treatment). At each study site, I positioned two acoustic sampling points >200 m from other acoustic sampling points and >100 m from edge habitat (excluding group selection harvest treatments, where I positioned acoustic sampling points at the edge of the group cut, and one study site managed for red-cockaded woodpeckers, where I positioned acoustic sampling points ~80 m from edge habitat due to area constraints). I also positioned all acoustic sampling points to have minimal vegetation above the microphones.

I conducted my acoustic sampling during two breeding (June to August 2020 and 2021) and two non-breeding (January to February 2020 and December to February 2021) periods (Harvey et al. 2011), hereafter called sampling periods. The climate of my study area is subtropical and has an average annual rainfall of 150 cm, and daily temperatures range from approximately 5˚C in January to approximately 35˚C in July (National Weather Service 2021). During the 2020 non-breeding period, the average dry bulb temperature was 9˚C and the average relative humidity was 83% (NOAA 2022). During the 2020 breeding period, the average dry bulb temperature was 25˚C and the average relative humidity was 79% (NOAA 2022). During the 2021 non-breeding period, the average dry bulb temperature was 4˚C and the average relative humidity was 83% (NOAA 2022). Last, during the 2021 breeding period, the average dry bulb temperature was 26˚C and the average relative humidity was 78% (NOAA 2022).

3.3. Methods

I deployed SM4BAT monitor microphones and AudioMoths ~15 cm apart on a 2-m tall polyvinyl chloride (PVC) pole with both microphones facing up (in accordance with Wildlife Acoustics, Inc. 2018) at the acoustic sampling points. Though I included 36 acoustic sampling points in my project, I did not sample all acoustic sampling points during each sampling period due to logistical constraints. I housed my AudioMoths in one of three housing units, including disposable plastic bags (recommended for deployment by Open Acoustic Devices; Open

Acoustic Devices 2021), plastic containers with the microphone hole covered with clear tape, and waterproof plastic bags designed to protect smartphones. I used different housing units for my study because several AudioMoths that I housed in disposable plastic bags and plastic containers were destroyed by water damage during the 2020 non-breeding period. I calibrated the AudioMoths to record at a 256 kHz sample rate with medium gain and, in accordance with firmware settings for V. 1.2.2. and previous research using AudioMoths to record bat activity (Katunzi et al. 2021), I programmed the AudioMoths to record cyclically at 10-second intervals followed by five-second pauses during which the AudioMoths stored data files to micro-SD cards located inside each device. I programmed my SM4BAT monitors with the default settings, which include triggering at a minimum detected frequency of 16 kHz for a minimum recording length of 1.5 ms and recording at a 256 kHz sample rate with a 12 dB gain. I programmed all monitors to turn on 30 min before sunset and to turn off 30 min after sunrise.

For both AudioMoth and SM4BAT data, I used the SonoBat (V. 4.4.1) batch file scrubber to exclude files without bat calls (e.g., noise), then used the SonoBat batch file classifier with the Southeastern southeast classifier and default values (acceptable call quality of 0.60 and decision threshold of 0.90) to automatically identify files to species. Species in the Southeastern southeast classifier include southeastern myotis (*Myotis austroriparius*), tricolored bats (*Perimyotis subflavus*), evening bats (*Nycticeius humeralis*), eastern red bats (*Lasiurus borealis*), big brown bats (*Eptesicus fuscus*), silver-haired bats (*Lasionycteris noctivagans*), Rafinesque’s big-eared bats (*Corynorhinus rafinesquii*), northern yellow bats (*Dasypterus intermedius*), Mexican free-tailed bats (*Tadarida brasiliensis*), and hoary bats (*Aeorestes cinereus*). SonoBat does not distinguish between the echolocation calls of eastern red bats and Seminole bats (*Lasiurus seminolus*) due to their similar call structures, so I designated identifications of eastern red bats as eastern red/Seminole bats for my project. SonoBat additionally classified files as high-frequency (typically $\geq$35 kHz) or low-frequency (typically <35 kHz) bat activity, which is helpful in estimating the activity of narrow and edge space aerial foragers (i.e., high-frequency calls) and open space aerial foragers (i.e., low-frequency calls; Denzinger and Schnitzler 2013). Hereafter I refer to files with bat echolocation as call files because I assumed that each file contained one full bat echolocation sequence. Last, SonoBat identified the number of echolocation pulses in each call file, and I defined an echolocation pulse as a single echolocation wave.

I conducted my statistical analyses in Program R (V. 4.1.1; R Core Team 2021). Because I had low sample sizes for some treatments within sampling periods and activity patterns were similar between sampling periods during the same time of year, I combined data for both non-breeding sampling periods into the non-breeding period and both breeding sampling periods into the breeding period. I then used a Kruskal-Wallis test (Kruskal and Wallis 1952) with $\alpha = 0.05$ to determine if the different types of housing units I used on the AudioMoths had an influence on the number of call files identified to species and species richness during the non-breeding period, indicating that I would need to separate my analyses by housing unit during this period. For each treatment within each period, I used Wilcoxon signed-rank tests (Wilcoxon 1945) to compare the median number of call files identified to species, the median total number of call files (the number of high-frequency call files and low-frequency call files), the median number of call files with high-frequency bat calls, the median number of call files with low-frequency bat calls, the median number of echolocation pulses recorded each night, and median species richness between monitors. I analyzed my treatments separately within each period because there were differences in activity at my acoustic sampling points across treatments.

I also compared the median number of call files identified to species or species groups of interest between monitors for each treatment within each period. These included the eastern red/Seminole bat group, tricolored bats, Mexican free-tailed bats, and hoary bats. I chose these species and species group based on their echolocation frequency ranges (eastern red/Seminole bat group and tricolored bats have high-frequency calls; Mexican free-tailed bats and hoary bats have low-frequency calls), because the calls within these groups can overlap, and because these species were consistently active in my study area. Again, I set $\alpha = 0.05$.

I presented the *Z*-scores associated with my Wilcoxon singed-rank tests to investigate the direction of effect for each acoustic metric; a positive *Z*-score indicated that SonoBat identified more call files, echolocation pulses, or species richness from SM4BAT recordings compared to AudioMoth recordings and a negative *Z*-score indicated that SonoBat identified more call files, echolocation pulses, or species richness from AudioMoth recordings compared to SM4BAT recordings. I presented the medians, ranges, means, and standard errors per monitor type for each variable in tables below. I estimated the magnitudes of difference in call metrics as the ratio of mean values between the monitors. Last, I calculated rank-biserial correlations (*r*) to estimate effect sizes for each acoustic metric (i.e., the difference between the proportion of nights when SonoBat identified more call files, echolocation pulses, or species richness from SM4BAT recordings compared to AudioMoth recordings) based on methods described by Kerby (2014). I assessed the strength of these correlations based on Asuero et al. (2006) with $r \geq |0.90|$ representing very high correlation, $r = |0.70–0.89|$ as high correlation, $r = |0.50–0.69|$ as moderate correlation, $r = |0.30–0.49|$ as low correlation, and $r \leq |0.29|$ as little or no correlation.

3.4. Results

I collected data on 237 nights. I found that SonoBat did not detect bat activity from files recorded during 61 nights for AudioMoths and 24 nights for SM4BATs. In addition, I found that SonoBat did not detect bat activity from files recorded during 7 nights from either monitor. Because my goal was to compare recordings from AudioMoths and SM4BAT monitors, I excluded nights without SM4BAT data from my analyses, resulting in 213 total nights of recordings from both monitors. Of these, I analyzed 62 nights during the non-breeding period (20 nights during the 2020 non-breeding period and 42 nights during the 2021 non-breeding period) and 151 nights during the breeding period (70 nights during the 2020 breeding period and 81 nights during the 2021 breeding period). I found no significant differences in the number of identified call files ($H_2 = 0.89$, $P = 0.64$) or species richness ($H_2 = 1.20$, $P = 0.55$) identified by SonoBat across the three housing units I used for my AudioMoths during the non-breeding period, so I included the recordings from all of the housing units in my subsequent analyses.

During the non-breeding period, I found that SonoBat identified an average of 99 times more call files to species, total call files, echolocation pulses, and species richness from SM4BAT recordings than AudioMoth recordings at the control, red-cockaded woodpecker, and thinned treatments, and that these metrics had little to very high positive correlations with monitor type at these treatments (Table 3.1.). Further, I found that SonoBat identified an average of 20 times more call files with high frequency bat calls from SM4BAT recordings than AudioMoth recordings at the bottomland hardwood, clearcut, control, red-cockaded woodpecker, and thinned treatments during the non-breeding period (Table 3.1.). In addition, I found that call files with high frequency bat calls had a high to very high positive correlation with monitor type at these treatments, meaning that there were more nights during which SonoBat identified more call files with high frequency bat calls from SM4BAT recordings compared to AudioMoth recordings (Table 3.1.). Moreover, I found that SonoBat identified 4 and 10 times more call files identified to species and species richness from SM4Bat recordings than AudioMoth recordings at the clearcut treatment, respectively, and that SonoBat identified 3 times more call files with low frequency bat calls from SM4BAT recordings than AudioMoth recordings at the red-cockaded woodpecker treatment (Table 3.1.). Last, I found moderate positive correlations between call files identified to species and species richness and monitor type at the clearcut treatment and little correlation between call files with low frequency bat calls and monitor type at the red-cockaded woodpecker treatment during the non-breeding period (Table 3.1.).

During the breeding period, I found that SonoBat identified an average of 7 times more call files with high frequency bat calls and echolocation pulses from SM4BAT recordings than AudioMoth recordings at the clearcut, control, group selection harvest, red-cockaded woodpecker, and thinned treatments, and that these metrics had moderate to very high positive correlations with monitor type at these treatments (Table 3.2.). In addition, I found that SonoBat identified an average of 2 times more total call files from SM4BAT recordings than AudioMoth recordings at the bottomland hardwood, clearcut, group selection harvest, and red-cockaded woodpecker treatments during the breeding period, but that total call files had little negative correlation to a very high positive correlation with monitor type at these treatments (Table 3.2.). Further, I found that SonoBat identified an average of 4 times more call files identified to species from SM4BAT recordings than AudioMoth recordings at the clearcut, group selection harvest, red-cockaded woodpecker, and thinned treatments during the breeding period, and that call files identified to species had a low to very high positive correlation with monitor type at these treatments (Table 3.2.). I also found that SonoBat identified 6 times more call files with low frequency bat calls from SM4BAT recordings than AudioMoth recordings at the clearcut treatment during the breeding period, and that this metric had a very high correlation with monitor type at this treatment (Table 3.2.). Last, I found that SonoBat identified an average of 3 times more species from SM4BAT recordings than AudioMoth recordings at all treatments during the breeding period, and that species richness had a moderate to very high correlation with monitor type (Table 3.2.).

I found that, during the non-breeding period, SonoBat identified an average of 18 times more call files as the eastern red/Seminole bat group from SM4BAT recordings compared to AudioMoth recordings at the clearcut and red-cockaded woodpecker treatments, but call files identified as eastern red/Seminole bats had little correlation with monitor type at both treatments (Table 3.3.). Similarly, during the non-breeding period, I found that SonoBat identified 18 times more call files as tricolored bats from SM4BAT recordings compared to AudioMoth recordings at the thinned treatment, but that call files identified as tricolored bats had little correlation with monitor type at this treatment (Table 3.4.). During the breeding period, I found that SonoBat identified an average of 53 times more call files as the eastern red/Seminole bat group from SM4BAT recordings compared to AudioMoth recordings at all treatments, and that call files identified as the eastern red/Seminole bat group had little to very high positive correlations with monitor type (Table 3.3.). Moreover, I found that SonoBat identified an average of 20 times more call files as tricolored bats from SM4BAT recordings compared to AudioMoth recordings at all treatments during the breeding period, and that call files identified as tricolored bats had little to very high positive correlations with monitor type (Table 3.4.).

I found that, during the breeding period, SonoBat identified an average of 5 times more call files as Mexican free-tailed bats from SM4BAT recordings than AudioMoth recordings at the clearcut and thinned treatments, and that call files identified as Mexican free-tailed bats had moderate to high positive correlations with monitor type at these treatments (Table 3.5.). In contrast, I found that Sonobat identified an average of 47 times more call files as hoary bats from AudioMoth recordings than SM4BAT recordings at the bottomland hardwood, control, group selection harvest, red-cockaded woodpecker, and thinned treatments during the breeding period (Table 3.6.). Further, I found that call files identified as hoary bats had low to very high negative correlations with monitor type at these treatments during the breeding period, meaning that there were more nights during which SonoBat identified more or the same number of call files as hoary bats from AudioMoth recordings compared to SM4BAT recordings (Table 3.6.).

Table 3.1. Summary statistics, Wilcoxon signed-rank test results (α = 0.05), and correlation coefficients for bat call file metrics recorded by AudioMoth and SM4BAT monitors in forest management practices of central Louisiana and identified by SonoBat during the non-breeding period (January–February 2020 and December 2020–February 2021). Asterisks denote significant differences between monitors.

Variable	Value	AudioMoth	SM4BAT	*Z*	*P*	*r*
Bottomland hardwood (n = 12)						
Call files identified to species	Median (range)	0 (0–55)	1 (0–26)	0.84	0.40	0
	Mean ± SE	4.9 ± 4.6	3.8 ± 2.1			
Total call files	Median (range)	0 (0–2734)	7.5 (1–111)	1.14	0.26	0.50
	Mean ± SE	409.3 ± 277.6	18.5 ± 8.9			
Call files with high frequency bat calls	Median (range)	0 (0–1)	7.5 (0–52)	2.94	0.003*	0.83
	Mean ± SE	0.2 ± 0.1	11.7 ± 4.3			
Call files with low frequency bat calls	Median (range)	0 (0–2734)	1 (0–59)	-0.06	0.95	-0.17
	Mean ± SE	409.1 ± 277.6	6.8 ± 4.8			
Echolocation pulses	Median (range)	0 (0–1255)	11 (0–530)	0.89	0.37	0.33
	Mean ± SE	181.9 ± 123.7	74.4 ± 43.9			
Species richness	Median (range)	0 (0–2)	1 (0–5)	1.71	0.09	0
	Mean ± SE	0.3 ± 0.2	1.3 ± 0.5			
Clearcut (n = 13)						
Call files identified to species	Median (range)	0 (0–17)	2 (0–25)	2.05	0.04*	0.54
	Mean ± SE	1.3 ± 1.3	5.2 ± 2.0			
Total call files	Median (range)	2 (0–169)	3 (1–42)	0.77	0.44	0.38
	Mean ± SE	22.7 ± 14.0	11.2 ± 3.9			
Call files with high frequency bat calls	Median (range)	0 (0–4)	3 (1–42)	3.20	0.001*	1.00
	Mean ± SE	0.9 ± 0.3	9.1 ± 3.4			
Call files with low frequency bat calls	Median (range)	0 (0–168)	1 (0–13)	-0.65	0.51	-0.23
	Mean ± SE	21.8 ± 14.0	2.1 ± 1.0			
Echolocation pulses	Median (range)	0 (0–1315)	27 (2–258)	1.50	0.13	0.54
	Mean ± SE	121.9 ± 101.0	76.3 ± 24.6			
Species richness	Median (range)	0 (0–2)	1 (0–6)	2.84	0.004*	0.54
	Mean ± SE	0.2 ± 0.2	1.9 ± 0.5			

Table cont'd.

Variable	Value	AudioMoth	SM4BAT	Z	P	r
Control (n = 8)						
Call files identified to species	Median (range)	0 (0–1)	2.5 (0–47)	2.21	0.03*	0.50
	Mean ± SE	0.1 ± 0.1	12.1 ± 6.4			
Total call files	Median (range)	0 (0–1)	10.5 (1–188)	2.53	0.01*	1.00
	Mean ± SE	0.1 ± 0.1	52.5 ± 25.1			
Call files with high frequency bat calls	Median (range)	0 (0)	9.5 (1–174)	2.53	0.01*	1.00
	Mean ± SE	0	49.9 ± 23.6			
Call files with low frequency bat calls	Median (range)	0 (0–1)	0 (0–14)	1.60	0.11	-0.25
	Mean ± SE	0.1 ± 0.1	2.6 ± 1.7			
Echolocation pulses	Median (range)	0 (0–6)	24 (0–531)	2.37	0.02*	0.75
	Mean ± SE	1.0 ± 0.7	175.1 ± 84.4			
Species richness	Median (range)	0 (0–1)	1.5 (0–6)	2.23	0.03*	0.50
	Mean ± SE	0.1 ± 0.1	2.0 ± 0.7			
Group selection harvest (n = 10)						
Call files identified to species	Median (range)	0 (0–7)	1.5 (0–11)	1.41	0.16	0.40
	Mean ± SE	2.2 ± 1.0	3.8 ± 1.4			
Total call files	Median (range)	9 (0–190)	6.5 (1–52)	0.24	0.81	0.20
	Mean ± SE	28.7 ± 18.3	13.9 ± 5.3			
Call files with high frequency bat calls	Median (range)	2 (0–29)	6 (1–23)	1.01	0.31	0.40
	Mean ± SE	6.5 ± 3.2	9.2 ± 2.6			
Call files with low frequency bat calls	Median (range)	2 (0–190)	0 (0–34)	-0.54	0.59	-0.60
	Mean ± SE	22.2 ± 18.7	4.7 ± 3.8			
Echolocation pulses	Median (range)	18 (0–244)	43.5 (0–204)	1.24	0.21	0.40
	Mean ± SE	42.7 ± 23.5	65.7 ± 24.3			
Species richness	Median (range)	0 (0–4)	1 (0–4)	1.46	0.14	0.40
	Mean ± SE	1.1 ± 0.5	1.6 ± 0.4			
Red-cockaded woodpecker (n = 11)						
Call files identified to species	Median (range)	0 (0–15)	1 (0–63)	2.54	0.01*	0.45
	Mean ± SE	1.8 ± 1.4	12.7 ± 5.7			
Total call files	Median (range)	0 (0–51)	8 (1–101)	2.81	0.01*	0.82
	Mean ± SE	6.4 ± 4.6	29.5 ± 10.2			

Table cont'd.

Variable	Value	AudioMoth	SM4BAT	*Z*	*P*	*r*
Call files with high frequency bat calls	Median (range)	0 (0–19)	7 (1–58)	2.94	0.003*	1.00
	Mean ± SE	2.3 ± 1.8	18.0 ± 6.6			
Call files with low frequency bat calls	Median (range)	0 (0–32)	1 (0–58)	1.99	0.047*	0.27
	Mean ± SE	4.1 ± 2.9	11.5 ± 5.7			
Echolocation pulses	Median (range)	0 (0–129)	20 (0–1193)	2.40	0.02*	0.82
	Mean ± SE	19.6 ± 11.8	196.8 ± 105.3			
Species richness	Median (range)	0 (0–3)	1 (0–5)	2.56	0.01*	0.45
	Mean ± SE	0.5 ± 0.3	1.8 ± 0.6			
Thinned (n = 8)						
Call files identified to species	Median (range)	0 (0–1)	2 (0–22)	2.26	0.02*	0.50
	Mean ± SE	0.3 ± 0.2	5.9 ± 2.9			
Total call files	Median (range)	6 (0–141)	15.5 (1–314)	2.38	0.02*	0.75
	Mean ± SE	22.6 ± 17.0	52.0 ± 37.8			
Call files with high frequency bat calls	Median (range)	5.5 (0–141)	14.5 (1–310)	2.39	0.02*	0.75
	Mean ± SE	21.9 ± 17.1	49.6 ± 37.4			
Call files with low frequency bat calls	Median (range)	0 (0–4)	0 (0–13)	0.92	0.36	-0.25
	Mean ± SE	0.8 ± 0.5	2.4 ± 1.6			
Echolocation pulses	Median (range)	5.5 (0–21)	42.5 (4–450)	2.52	0.01*	1.00
	Mean ± SE	7.4 ± 2.7	104.3 ± 52.6			
Species richness	Median (range)	0 (0–1)	2 (0–6)	2.06	0.04*	0.25
	Mean ± SE	0.3 ± 0.2	2.1 ± 0.7			

Table 3.2. Summary statistics, Wilcoxon signed-rank test results (α = 0.05), and correlation coefficients for bat call file metrics recorded by AudioMoth and SM4BAT monitors in forest management practices of central Louisiana and identified by SonoBat during the breeding period (June–August 2020 and 2021).

Variable	Value	AudioMoth	SM4BAT	*Z*	*P*	*r*
Bottomland hardwood (n = 16)						
Call files identified to species	Median (range)	8 (0–260)	19 (0–45)	-0.93	0.35	0
	Mean ± SE	45.2 ± 17.5	21.3 ± 3.5			
Total call files	Median (range)	343.5 (0–1689)	94.5 (2–380)	-1.96	0.049*	-0.25
	Mean ± SE	387.3 ± 119.6	109.3 ± 24.4			
Call files with high frequency bat calls	Median (range)	3 (0–1624)	52 (0–261)	1.81	0.07	0.63
	Mean ± SE	158.7 ± 11.5	71.8 ± 17.8			
Call files with low frequency bat calls	Median (range)	59.5 (0–1045)	23.5 (0–119)	-1.81	0.07	-0.25
	Mean ± SE	228.6 ± 72.8	37.4 ± 9.0			
Echolocation pulses	Median (range)	99.5 (0–1451)	282 (3–932)	0.31	0.76	0.13
	Mean ± SE	297.4 ± 98.1	325.3 ± 72.2			
Species richness	Median (range)	2 (0–2)	4.5 (0–9)	3.09	0.002*	0.63
	Mean ± SE	1.4 ± 0.2	4.3 ± 0.6			
Clearcut (n = 16)						
Call files identified to species	Median (range)	6 (0–84)	165 (3–472)	3.52	<0.001*	1.00
	Mean ± SE	23.2 ± 7.9	168.8 ± 30.3			
Total call files	Median (range)	33 (0–238)	397 (16–1174)	3.52	<0.001*	1.00
	Mean ± SE	83.7 ± 24.2	458.9 ± 80.0			
Call files with high frequency bat calls	Median (range)	21 (0–162)	187.5 (11–478)	3.52	<0.001*	1.00
	Mean ± SE	41.6 ± 12.5	192.3 ± 32.6			
Call files with low frequency bat calls	Median (range)	12 (0–174)	208.5 (5–861)	3.52	<0.001*	1.00
	Mean ± SE	42.1 ± 15.4	266.7 ± 54.0			
Echolocation pulses	Median (range)	63 (0–529)	2986.5 (73–9601)	3.52	<0.001*	1.00
	Mean ± SE	185.3 ± 54.8	3339.9 ± 637.4			
Species richness	Median (range)	3 (0–7)	8 (2–9)	3.52	<0.001*	1.00
	Mean ± SE	3.2 ± 0.6	7.3 ± 0.4			

Table cont'd.

Variable	Value	AudioMoth	SM4BAT	*Z*	*P*	*r*
Control (n = 28)						
Call files identified to species	Median (range)	54 (0–431)	69 (3–617)	0.93	0.35	0.36
	Mean ± SE	107.2 ± 22.9	123.5 ± 28.7			
Total call files	Median (range)	287 (0–2553)	210 (18–1401)	-0.16	0.87	0.14
	Mean ± SE	447.3 ± 105.8	324.5 ± 60.2			
Call files with high frequency bat calls	Median (range)	3 (0–1217)	52.5 (0–200)	3.94	<0.001*	0.86
	Mean ± SE	47.9 ± 43.3	74.8 ± 10.6			
Call files with low frequency bat calls	Median (range)	218.5 (0–2553)	148.5 (2–1374)	-0.93	0.35	-0.21
	Mean ± SE	399.4 ± 103.01	249.7 ± 57.0			
Echolocation pulses	Median (range)	334.5 (0–2553)	602 (45–3790)	2.25	0.02*	0.57
	Mean ± SE	658.9 ± 138.4	1064.8 ± 183.8			
Species richness	Median (range)	2 (0–4)	5.5 (1–8)	4.52	<0.001*	0.93
	Mean ± SE	1.8 ± 0.2	5.3 ± 0.3			
Group selection harvest (n = 35)						
Call files identified to species	Median (range)	6 (0–442)	117 (0–688)	2.27	0.02*	0.37
	Mean ± SE	73.1 ± 19.9	158.6 ± 26.5			
Total call files	Median (range)	61 (0–1419)	412 (2–1699)	2.56	0.01*	0.43
	Mean ± SE	268.0 ± 64.7	576.8 ± 84.8			
Call files with high frequency bat calls	Median (range)	14 (0–99)	260 (0–1380)	4.93	<0.001*	0.83
	Mean ± SE	25.1 ± 4.9	357.0 ± 56.7			
Call files with low frequency bat calls	Median (range)	46 (0–1366)	141 (0–979)	-0.01	0.99	0.20
	Mean ± SE	242.9 ± 61.5	219.8 ± 41.9			
Echolocation pulses	Median (range)	94 (0–3026)	1968 (6–8374)	4.13	<0.001*	0.66
	Mean ± SE	546.7 ± 139.2	2662.3 ± 385.3			
Species richness	Median (range)	2 (0–5)	7 (0–9)	4.68	<0.001*	0.77
	Mean ± SE	1.9 ± 0.3	6.3 ± 0.4			
Red-cockaded woodpecker (n = 27)						
Call files identified to species	Median (range)	23 (0–423)	116 (1–605)	3.44	<0.001*	0.63
	Mean ± SE	57.5 ± 17.8	162.8 ± 29.0			
Total call files	Median (range)	131 (0–891)	423 (2–1237)	3.77	<0.001*	0.70
	Mean ± SE	199 ± 43.9	462.9 ± 64.0			

Table cont’d.

Variable	Value	AudioMoth	SM4BAT	*Z*	*P*	*r*
Call files with high frequency bat calls	Median (range)	30 (0–283)	234 (1–537)	4.35	<0.001*	0.78
	Mean ± SE	57.1 ± 13.5	248.9 ± 30.5			
Call files with low frequency bat calls	Median (range)	46 (0–842)	133 (1–888)	1.63	0.10	0.26
	Mean ± SE	141.9 ± 37.2	214.0 ± 43.2			
Echolocation pulses	Median (range)	243 (0–3420)	1824 (19–13936)	4.42	<0.001*	0.85
	Mean ± SE	560.9 ± 168.7	3222.3 ± 632.3			
Species richness	Median (range)	2 (0–8)	8 (1–9)	3.78	<0.001*	0.70
	Mean ± SE	3.0 ± 0.5	6.9 ± 0.4			
Thinned (n = 29)						
Call files identified to species	Median (range)	8 (0–252)	82 (0–324)	3.28	0.001*	0.59
	Mean ± SE	53.3 ± 14.0	111.7 ± 15.4			
Total call files	Median (range)	66 (0–929)	249 (1–655)	1.30	0.19	0.24
	Mean ± SE	219.3 ± 51.0	285.3 ± 32.9			
Call files with high frequency bat calls	Median (range)	5 (0–38)	110 (0–242)	4.64	<0.001*	0.86
	Mean ± SE	9.7 ± 2.0	117.1 ± 12.1			
Call files with low frequency bat calls	Median (range)	61 (0–923)	139 (0–450)	-0.35	0.72	0.17
	Mean ± SE	209.7 ± 49.9	168.2 ± 25.1			
Echolocation pulses	Median (range)	84 (0–1818)	1238 (1–3693)	4.29	<0.001*	0.79
	Mean ± SE	351.9 ± 87.8	1396.0 ± 172.6			
Species richness	Median (range)	2 (0–5)	7 (0–8)	4.62	<0.001*	0.86
	Mean ± SE	1.6 ± 0.3	6.3 ± 0.4			

Table 3.3. Summary statistics, Wilcoxon signed-rank test results (α = 0.05), and correlation coefficients for the number of call files identified by SonoBat as eastern red/Seminole bats (*Lasiurus borealis/L. seminolus*) from recordings collected by AudioMoth and SM4BAT monitors in forest management practices of central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

Treatment	Value	AudioMoth	SM4BAT	*Z*	*P*	*r*
Non-breeding period						
Bottomland hardwood	Median (range)	0 (0–1)	0 (0–2)	1.34	0.18	-0.67
	Mean ± SE	0.1 ± 0.1	0.3 ± 0.2			
Clearcut	Median (range)	0 (0–1)	0 (0–25)	2.26	0.02*	-0.08
	Mean ± SE	0.1 ± 0.1	2.8 ± 1.9			
Control	Median (range)	0 (0)	1.5 (0–45)	1.83	0.07	0
	Mean ± SE	0	10.1 ± 5.8			
Group selection harvest	Median (range)	0 (0–3)	0 (0–4)	0.92	0.36	-0.40
	Mean ± SE	0.3 ± 0.3	0.8 ± 0.5			
Red-cockaded woodpecker	Median (range)	0 (0–7)	1 (0–17)	2.21	0.03*	0.09
	Mean ± SE	0.6 ± 0.6	5.1 ± 2.1			
Thinned	Median (range)	0 (0)	0 (0–7)	1.34	0.18	-0.50
	Mean ± SE	0	1.1 ± 0.9			
Breeding period						
Bottomland hardwood	Median (range)	0 (0)	4 (0–11)	2.81	0.005*	0.25
	Mean ± SE	0	4.2 ± 1.0			
Clearcut	Median (range)	0.5 (0–33)	34.5 (0–101)	3.41	<0.001*	0.88
	Mean ± SE	4.4 ± 2.3	36.8 ± 7.5			
Control	Median (range)	0 (0)	8 (0–44)	4.46	<0.001*	0.86
	Mean ± SE	0	10.9 ± 1.9			
Group selection harvest	Median (range)	0 (0–5)	29 (0–151)	4.86	<0.001*	0.71
	Mean ± SE	0.7 ± 0.3	42.4 ± 6.9			
Red-cockaded woodpecker	Median (range)	0 (0–20)	29 (0–81)	4.33	<0.001*	0.78
	Mean ± SE	3.0 ± 1.0	34.2 ± 5.2			
Thinned	Median (range)	0 (0–2)	11 (0–44)	4.63	<0.001*	0.93
	Mean ± SE	0.1 ± 0.1	13.2 ± 1.7			

Table 3.4. Summary statistics, Wilcoxon signed-rank test results (α = 0.05), and correlation coefficients for the number of call files identified by SonoBat as tricolored bats (*Perimyotis subflavus*) from recordings collected by AudioMoth and SM4BAT monitors in forest management practices of central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

Treatment	Value	AudioMoth	SM4BAT	*Z*	*P*	*r*
Non-breeding period						
Bottomland hardwood	Median (range)	0 (0)	0 (0–3)	1.34	0.18	-0.67
	Mean ± SE	0	0.4 ± 0.3			
Clearcut	Median (range)	0 (0)	0 (0–2)	1.63	0.10	-0.54
	Mean ± SE	0	0.3 ± 0.2			
Control	Median (range)	0 (0)	0 (0)	-	-	-
	Mean ± SE	0	0			
Group selection harvest	Median (range)	0 (0–1)	0 (0–1)	0.58	0.56	-0.60
	Mean ± SE	0.2 ± 0.1	0.3 ± 0.2			
Red-cockaded woodpecker	Median (range)	0 (0)	0 (0–1)	1.41	0.16	-0.64
	Mean ± SE	0	0.2 ± 0.1			
Thinned	Median (range)	0 (0–1)	1 (0–8)	2.06	0.04*	0.25
	Mean ± SE	0.1 ± 0.1	1.8 ± 1.0			
Breeding period						
Bottomland hardwood	Median (range)	0 (0)	2.5 (0–11)	3.08	0.002*	0.50
	Mean ± SE	0	3.2 ± 0.8			
Clearcut	Median (range)	1.5 (0–8)	22 (0–70)	3.41	<0.001*	0.88
	Mean ± SE	2.6 ± 0.6	23.4 ± 4.2			
Control	Median (range)	0 (0)	1 (0–8)	3.42	<0.001*	0.07
	Mean ± SE	0	1.8 ± 0.4			
Group selection harvest	Median (range)	0 (0–3)	17 (0–118)	4.83	<0.001*	0.71
	Mean ± SE	0.5 ± 0.1	23.4 ± 4.4			
Red-cockaded woodpecker	Median (range)	0 (0–73)	10 (0–159)	4.35	<0.001*	0.78
	Mean ± SE	4.6 ± 2.7	21.3 ± 6.0			
Thinned	Median (range)	0 (0–1)	13 (0–29)	4.46	<0.001*	0.79
	Mean ± SE	0.03 ± 0.03	11.0 ± 1.4			

Table 3.5. Summary statistics, Wilcoxon signed-rank test results (α = 0.05), and correlation coefficients for the number of call files identified by SonoBat as Mexican free-tailed bats (*Tadarida brasiliensis*) from recordings collected by AudioMoth and SM4BAT monitors in forest management practices of central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

Treatment	Value	AudioMoth	SM4BAT	*Z*	*P*	*r*
Non-breeding period						
Bottomland hardwood	Median (range)	0 (0–1)	0 (0–15)	1.29	0.20	-0.50
	Mean ± SE	0.1 ± 0.1	1.6 ± 1.2			
Clearcut	Median (range)	0 (0)	0 (0–2)	1.34	0.18	-0.69
	Mean ± SE	0	0.2 ± 0.2			
Control	Median (range)	0 (0)	0 (0–1)	1.41	0.16	-0.50
	Mean ± SE	0	0.3 ± 0.2			
Group selection harvest	Median (range)	0 (0–5)	0 (0–2)	-1.34	0.18	-1.00
	Mean ± SE	0.9 ± 0.5	0.3 ± 0.2			
Red-cockaded woodpecker	Median (range)	0 (0–4)	0 (0–11)	1.60	0.11	-0.45
	Mean ± SE	0.5 ± 0.4	2.1 ± 1.2			
Thinned	Median (range)	0 (0–1)	0 (0–7)	1.34	0.18	-0.50
	Mean ± SE	0.1 ± 0.1	1.1 ± 0.9			
Breeding period						
Bottomland hardwood	Median (range)	6.5 (0–108)	5 (0–17)	-1.76	0.08	-0.38
	Mean ± SE	22.4 ± 7.8	6.4 ± 1.4			
Clearcut	Median (range)	2 (0–51)	24 (0–233)	3.41	<0.001*	0.88
	Mean ± SE	10.6 ± 4.4	49.3 ± 14.7			
Control	Median (range)	28.5 (0–239)	42.5 (0–599)	1.37	0.17	0.21
	Mean ± SE	56.1 ± 13.7	94.5 ± 27.9			
Group selection harvest	Median (range)	2 (0–280)	33 (0–557)	0.99	0.32	0.14
	Mean ± SE	38.1 ± 10.8	63.7 ± 18.4			
Red-cockaded woodpecker	Median (range)	6 (0–328)	15 (0–331)	0.81	0.42	0.26
	Mean ± SE	30.0 ± 12.6	31.4 ± 12.0			
Thinned	Median (range)	2 (0–139)	41 (0–288)	2.76	0.006*	0.52
	Mean ± SE	27.1 ± 7.5	59.4 ± 13.9			

Table 3.6. Summary statistics, Wilcoxon signed-rank test results (α = 0.05), and correlation coefficients for the number of call files identified by SonoBat as hoary bats (*Aeorestes cinereus*) from recordings collected by AudioMoth and SM4BAT monitors in forest management practices of central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods.

Treatment	Value	AudioMoth	SM4BAT	*Z*	*P*	*r*
Non-breeding period						
Bottomland hardwood	Median (range)	0 (0–55)	0 (0–4)	0.31	0.75	-0.33
	Mean ± SE	4.8 ± 4.8	0.8 ± 0.4			
Clearcut	Median (range)	0 (0–16)	0 (0–1)	0	1.00	-0.69
	Mean ± SE	1.2 ± 1.2	0.2 ± 0.1			
Control	Median (range)	0 (0–1)	0 (0–5)	1.34	0.18	-0.50
	Mean ± SE	0.1 ± 0.1	0.9 ± 0.6			
Group selection harvest	Median (range)	0 (0–3)	0 (0–7)	0.55	0.58	-0.60
	Mean ± SE	0.8 ± 0.4	1.2 ± 0.8			
Red-cockaded woodpecker	Median (range)	0 (0–1)	0 (0–4)	1.63	0.10	-0.45
	Mean ± SE	0.1 ± 0.1	0.5 ± 0.4			
Thinned	Median (range)	0 (0)	0 (0–1)	1.00	0.32	-0.75
	Mean ± SE	0	0.1 ± 0.1			
Breeding period						
Bottomland hardwood	Median (range)	1.5 (0–152)	0 (0–2)	-2.49	0.01*	-0.75
	Mean ± SE	22.8 ± 9.9	0.4 ± 0.2			
Clearcut	Median (range)	1 (0–37)	3 (0–19)	1.80	0.07	0.25
	Mean ± SE	4.6 ± 2.4	4.1 ± 1.2			
Control	Median (range)	16.5 (0–222)	0 (0–3)	-4.11	<0.001*	-1.00
	Mean ± SE	50.9 ± 12.6	0.4 ± 0.2			
Group selection harvest	Median (range)	4 (0–287)	0 (0–25)	-3.32	<0.001*	-0.71
	Mean ± SE	33.7 ± 11.1	1.5 ± 0.8			
Red-cockaded woodpecker	Median (range)	6 (0–94)	1 (0–19)	-2.72	0.006*	-0.48
	Mean ± SE	14.0 ± 4.1	2.3 ± 0.8			
Thinned	Median (range)	1 (0–133)	0 (0–9)	-3.20	0.001*	-0.86
	Mean ± SE	25.5 ± 6.8	1.1 ± 0.4			

3.5. Discussion

I found that SonoBat detected bat activity on 16% fewer nights from files recorded by AudioMoths compared to SM4BATs. Further, I found that SonoBat classified a greater number of identified call files, call files with high frequency bat calls, echolocation pulses, and species from SM4BAT recordings compared to AudioMoth recordings during both periods; however, I found that these results varied by treatment. In addition, I largely found that SonoBat identified a similar number of call files with low-frequency bat calls between monitors, but that SonoBat identified more call files as hoary bats from AudioMoth recordings compared to SM4BAT recordings during the breeding period. My results likely reflected differences in microphone sensitivities (i.e., the distance at which a microphone can record a subject; Britzke et al. 2013), recording specifications, and housing units between the monitors and suggest that SM4BATs provide more comprehensive data that can be used with automated classification software than the version of AudioMoths I used.

Many factors can influence detectability of bat calls by ARUs, including the acoustic properties of the species' calls (e.g., frequency, intensity), the foraging behavior of the species or individuals, habitat conditions, and microphone sensitivities (Britzke et al. 2013). In particular, ARUs with less sensitive microphones record fewer detections of bats with high- or low-frequency calls (Downes 1982, Fenton et al. 2001, Adams et al. 2012) because they detect sounds at shorter distances from the microphone (Britkze et al. 2013). In previous studies, researchers used AudioMoths to document bat species with high frequency calls (Revilla-Martín et al. 2020, Montauban et al. 2021). In my study, SonoBat did identify some high frequency calls from the AudioMoth recordings, but also missed many high frequency calls that were recorded by the SM4BATs likely due to differences in microphone sensitivity between the two monitors.

My results indicate that researchers should be particularly cautious when using AudioMoths to study high-frequency bat species of interest.

Furthermore, the distance that bat calls travel can vary greatly due to environmental conditions (Britzke et al. 2013). For example, high-frequency sound travels shorter distances in hot and humid conditions, meaning that microphones must be hypersensitive to detect high-frequency bat calls in these environments (Petterson 2004). I conducted my study in Louisiana, which has high temperatures and humidity year-round (NOAA 2022). Studying bats in the hot and humid conditions of Louisiana requires that we use highly sensitive monitors to detect high-frequency bat species at far distances from the monitor (Petterson 2004). Thus, it is again likely that AudioMoths were less sensitive than SM4BAT microphones because Sonobat identified more high-frequency bat call files and high-frequency species from SM4BATs compared to AudioMoths. Further, due to these environmental conditions, I may have observed more pronounced differences in the number of species and the number of high-frequency call files recorded by the two monitors than I would have found if I did the same study outside the southeastern U.S.

Understory vegetation structure also likely influenced my results by affecting both bat activity and detectability. Bat activity is often lower in forests with complex understory vegetation structure (Titchenell et al. 2011, Armitage and Ober 2012, Brooks et al. 2017), and bat detectability can be lower in forests with a dense understory compared to forests with more open understory conditions (Britzke et al. 2013). I often observed lower activity (and, thus, a smaller number of recordings from both monitors) in the bottomland hardwood and control treatments, which had complex understory vegetation structures. Conversely, I found that treatments characterized by an open habitat structure (e.g., clearcut and red-cockaded woodpecker treatments) often showed differences between monitors in the number of call files, echolocation pulses, and species recorded, with SM4BATs recording more of each metric. Combined, these findings suggest that the vegetation structures of my treatments influenced the number of recordings that I collected for each monitor, and, thus, my overall results.

Monitor firmware could also contribute to the variation in the number of call files I identified from the SM4BAT recordings compared to the AudioMoth recordings. SM4BATs are designed to increase the signal-to-noise ratio, meaning they adjust to record clearer files as background noise increases or decreases throughout the night (Wildlife Acoustics, Inc. 2018). Similar to researchers who purchased and deployed AudioMoths prior to the firmware update in 2020, the AudioMoths I used did not have signal-to-noise ratio adjustments, amplitude thresholds, or frequency thresholds, and I did not adopt the firmware update after its release to avoid introducing additional variation to the current study. The firmware that I used likely influenced my results because the monitor did not adjust to filter out noise, which decreased the clarity of recordings and inhibited automatic classifications of these recordings to species. Further, it is possible that the newer firmware could increase the clarity of recordings because it includes amplitude and frequency thresholds (Open Acoustic Devices 2020*b*). Thus, it is important for researchers comparing the results of AudioMoth studies across time to consider the AudioMoth's firmware, as future firmware updates could improve on the clarity of recordings and allow for better classifications of call files.

In addition to firmware, it is possible that the housing units I used for AudioMoths influenced the results of my study. SM4BATs are built with weatherproof housing units and external microphones, but AudioMoths do not come with pre-made enclosures for the circuit board and internal microphone. I ultimately housed my AudioMoths in waterproof smartphone bags because I experienced equipment failure due to water damage. I found no differences in the number of identified call files or species richness recorded between disposable plastic bags, which Open Acoustic Devices recommends for deployment (Open Acoustic Devices 2021), and the other plastic housing units I used. However, any enclosure could decrease the amount of sound that reaches the AudioMoth's internal microphone. Indeed, previous studies concerning the use of weatherproofed housing units on a variety of monitor types show that housing units influence the detectability of bats depending on the housing unit design (Britzke et al. 2010, Kaiser and O'Keefe 2015). Further, in a separate study exploring bat occurrence across a rural-to-urban gradient, I deployed AudioMoths in 3D printed housing units with an acoustic membrane over the microphone (Prusa Printers Beta 2020) for fourteen nights and found more congruence between files recorded by the AudioMoths and other commercially available bat detectors than I found in the current study (A. M. Long, *unpublished data*). Therefore, though the AudioMoth housing units likely had an effect on my findings, the small differences among housing units in my study and my use of a housing unit recommended by Open Acoustic Devices suggest that housing unit type had minimal effects on my results.

Though I found that SM4BATs detected more bat activity and species than AudioMoths, no ARU will record 100% of bat calls present in the environment and AudioMoths may represent a useful research tool for some applications. For example, AudioMoths deployed in areas with less humidity and lower temperatures may record similar numbers of both high- and low-frequency bat calls compared to other ARUs. Also, improvements in housing unit designs may increase the amount of sound that can reach the AudioMoth microphone, further increasing the number of recordings that can be identified by automated classification software. In addition, SonoBat identified a similar number of call files with low-frequency bat calls from AudioMoth recordings and SM4BAT recordings, suggesting that AudioMoths may be useful for surveying low-frequency bat species occurrence and activity (e.g., hoary bats, Florida bonneted bats [*Eumops floridanus*]) or bats that primarily echolocate in the audible spectrum (e.g., spotted bats [*Euderma maculatum*]). Last, AudioMoths may record more files that are identifiable to species with automated classification software compared to zero-crossing ARUs (Dailey 2020).

There are many ARUs that record ultrasound on the market today, but there is limited research comparing ARU recording quality across units. Differences in recording quality across ARUs can have a direct influence on the results of monitoring studies, particularly when researchers use automated classifiers to identify species presence in their study area. With declining bat populations worldwide, it is crucial that estimates of species occurrence and activity are accurate to best manage for species of conservation concern. I found that SM4BATs recorded more call files that I could identify with SonoBat than AudioMoths in my study area, suggesting that researchers should use caution when deploying AudioMoths to study high-frequency bats in climates that are hot and humid. However, I have no doubt that the open-sourced technology used to produce AudioMoths will improve rapidly in the coming years, and I urge researchers using AudioMoths to report the firmware and all the settings they used, details on their housing units, typical environmental conditions in their study area, and any issues they had during the recording or classification process. Such transparency could help researchers compare AudioMoth results across studies and identify the limitations of deploying AudioMoths in certain study areas, resulting in improved estimates of bat species occurrence and activity that can inform management for species of conservation concern.

Chapter 4. Conclusions

I found that bat species composition was similar between the non-breeding and breeding periods in the Kisatchie National Forest, and that foraging group composition and species richness did not differ among treatments. In addition, I found that group selection harvest, red-cockaded woodpecker habitat, and clearcut treatments had high bat activity and foraging activity for multiple species compared to bottomland hardwood, thinned, and control treatments. I also found that different habitat variables affected the predicted probability of big brown bats and *Myotis* species occupancy between the non-breeding and breeding periods, and my data demonstrated that patterns of habitat use changed throughout the year for these species. Further, I found that variables associated with riparian habitat (e.g., distance to the nearest perennial water source, proportion of forest aged >30 years in a 1-km radius) negatively influenced the predicted probability of occupancy for evening bats, eastern red/Seminole bats, *Myotis* species, and big brown bats. Last, I found that both Rafinesque's big-eared bats and tricolored bats roosted in double-T bridges during the non-breeding period, but that tricolored bats did not roost in the types of infrastructure I surveyed during the breeding period. My results suggested that heterogenous landscapes with edge and open habitat structures promote bat diversity and activity in Louisiana, and that treatments like group selection harvests, red-cockaded woodpecker habitat, and clearcuts provide navigational and foraging habitat for bats throughout the year. Further, my research emphasized the importance of studying bats during both non-breeding and breeding periods.

I found that SonoBat identified a greater number of call files, echolocation pulses, and species richness from SM4BAT recordings compared to AudioMoth recordings. More specifically, I found that SonoBat identified a greater number of call files with high-frequency bat calls from SM4BAT recordings compared to AudioMoth recordings at almost all treatments during both the non-breeding and breeding periods, but that SonoBat identified a similar number of call files with low-frequency bat calls from both monitors. My results suggested that SM4BATs are more sensitive than AudioMoths, meaning that SM4BAT microphones can detect sounds at further distances than AudioMoth microphones. Further, I likely observed more pronounced difference in recordings between monitors than other researchers might find because hot and humid environmental conditions can affect the distance that high-frequency sound travels in air. In addition, the complex understory vegetation structures of my control and bottomland hardwood treatments may have influenced bat activity and detectability at my study sites, which could have affected the patterns I observed at these treatments. Further, factors like monitor firmware and AudioMoth housing units may have affected the number of recordings that AudioMoths collected. My results suggested that researchers should use caution when deploying AudioMoths in hot and humid climates to study high-frequency bats. Last, my results emphasized that researchers using AudioMoths to study bats should report the climate of their study area and all AudioMoth firmware, settings, and housing units that they used, which could help researchers more accurately compare acoustic survey results across studies.

Appendix. Supplementary Data for Chapter 2.

Figure A.1. The three most common structures within 2 km of my study sites, which I found during my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. These structures include box culverts (top-left), circular culverts (top-right), and double-T bridges (bottom).

Table A.1. Results from Kruskal-Wallis rank sum tests for significant differences among treatments for each species for my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. Asterisks denote significant differences at $\alpha = 0.05$.

Sampling period	Species	DF	*H*-value	*P*-value
2020 non-breeding	Eastern red/Seminole	3	0.56	0.91
	Hoary	3	1.47	0.69
	Myotis species	3	4.68	0.20
	Tricolored	3	4.71	0.19
	Mexican free-tailed	3	0.55	0.91
	Big brown	3	2.09	0.55
2020 breeding	Eastern red/Seminole	3	9.53	0.02*
	Hoary	3	2.09	0.55
	Myotis species	3	3.24	0.36
	Tricolored	3	14.06	0.003*
	Mexican free-tailed	3	3.56	0.31
	Big brown	3	3.18	0.36
	Evening	3	6.67	0.08
2021 non-breeding	Eastern red/Seminole	5	9.26	0.10
	Hoary	5	2.79	0.73
	Myotis species	5	8.00	0.16
	Tricolored	5	3.65	0.61
	Mexican free-tailed	5	1.77	0.88
	Big brown	5	11.45	0.04*
2021 breeding	Eastern red/Seminole	5	22.64	<0.001*
	Hoary	5	5.00	0.42
	Myotis species	5	13.38	0.02*
	Tricolored	5	18.93	0.002*
	Mexican free-tailed	5	10.14	0.07
	Big brown	5	9.09	0.11
	Evening	5	2.71	0.75

Figure A.2. A tricolored bat roosting in a circular culvert during the 2021 non-breeding period, which I detected during my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. This was my only case of a bat detection in infrastructure other than a double-T bridge.

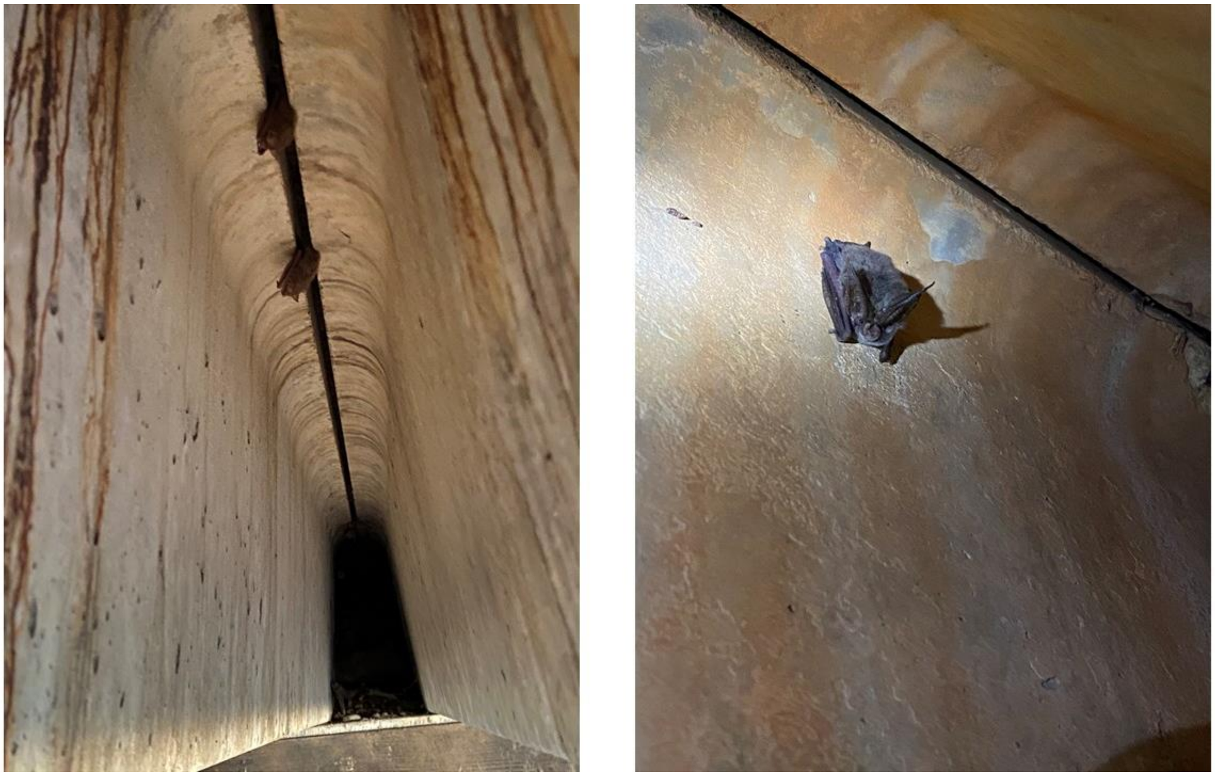

Figure A.3. Examples of tricolored bats (left) and a Rafinesque’s big eared bat (right) roosting in double-T bridges, which I detected during my study examining the influence of forest management practices on seasonal bat species occurrence and activity in central Louisiana during the non-breeding (January–February 2020 and December 2020–February 2021) and breeding (June–August 2020 and 2021) periods. Crevices and cracks are visible in the cement, which the bats used to roost.

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Vita

Jane Marie Kunberger grew up in Franklin, Tennessee where she volunteered to help with habitat restoration projects as a teenager. She received her bachelor’s degree from the University of Alabama and decided to enter the School of Renewable Natural Resources at Louisiana State University to further her education in wildlife ecology. Upon completion of her master’s degree, Jane will continue working on research aimed at wildlife habitat conservation.