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Spatial Use by Mammals within Two State Parks in the Ozarks National Scenic Riverways

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SPATIAL USE BY MAMMALS WITHIN TWO STATE PARKS IN THE OZARKS NATIONAL SCENIC RIVERWAYS

A Master's Thesis

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In Partial Fulfillment

Of the Requirements for the Degree

Master of Science, Biology

By

Benjamin Aaron Smith

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ABSTRACT

A mammalian species inventory with comparisons between sampled sites was conducted via multiple methodologies to document presence of mammals at two Missouri state parks within the Ozark National Scenic Riverways. Camera traps, small mammal traps, acoustic detectors, and mist nets were used to detect species at the parks, and species similarity indices and occupancy analyses were used to discern use of space. A mammalian inventory was compiled for each area of inquiry. Greater diversity was found at the park with more variable habitat types. Bat activity was more in the park with a known hibernaculum, though species specific activity differed between sites. Small mammal captures and diversity were skewed toward the park with more overall ecological diversity, with more captures of generalists at the other park.

KEYWORDS: species richness, accumulation curves, passive sampling, camera traps, live traps, bat acoustics

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OVERVIEW

Biogeography and Animal Spatial Use

Describing the distribution of species in geographic space, the communities they form in those spaces, and the causes of the variation among the spaces/communities constitute the goal of the study of biogeography. Alexander von Humboldt, known as the father of biogeography, was the pioneer of quantitative botanical geography and documented various plant morphologies on his extensive travels around the world (von Humboldt and Bonpland 1805). Charles Darwin, with influence from Alfred Russel Wallace, was the first to relate the locality of species with their evolutionarily driven traits and coupled the two concepts thereafter (Darwin and Wallace 1858; Darwin 1872). A multitude of factors can influence the dispersal and eventual distribution pattern of a species depending upon the scale of inquiry. These can range from large scale climatic influences (Ackerly et al. 2010) down to the top predator of a locality (Mcpeek 1998). Influencing factors can be abiotic (a physical or climatic barrier to the dispersal of species to other potentially favorable habitats; Taulman and Robbins 1996), or biotic (a biological limitation on the species fitness in an area preventing dispersal; Pigot and Tobias 2013). These two factors can often limit range and distribution in concert with one another making it difficult to discern which is the main driving influence.

Species distribution can also be influenced on much smaller and more local scales. These smaller scale influences are generally referred to as habitat preferences (the habitat preferred by a species given a choice). The main areas of study on habitat preference center around how a species makes use of its local environment, with emphasis on the types of habitats that it

occupies and the foods consumed within them (Johnson 1980). Habitat preferences can range from rather vague in more generalist, to specifically defined in specialized species.

Habitat preferences and the variation in use within them drive multiple scales of movement within and among habitats. The most localized scale of movement is an individual's home range; the area in which an individual lives and carries out its daily activities (Burt 1943). Home range is often associated with the concept of an individual's territory but the main difference is that a territory is a space made up of the entirety or only a highly valued portion of the home range, which is defended against conspecifics through an olfactory (Eisenberg and Kleiman 1972), audible (Nowicki et al. 2014), or visual marker (Kilshaw et al. 2009). Territorial markers often have to be regularly maintained and therefore pass on an energetic cost to the defender. Home ranges and territories are two examples of local scale areas through which an individual moves, but with very different usages that cause variation in movement in each.

In contrast to the localized movements of an individual, there are large annual patterns of species movements within a region referred to as migration. Migration can vary in distance and purpose depending upon the species. In general, species migrate to improve their fitness by moving to an area that provides better resource availability, survival opportunity, and/or reproductive chances (Shaw and Couzin 2013; Barten et al. 2014). Long distance travel is energetically expensive so the fitness payoff for such expenditure needs to be worthwhile. Most migration is tied to a seasonal change in resource abundance that in turn leads to increased survival and more food availability for reproduction, offspring development, and continued activity. Migration is one of the largest and most variable movement types, but it is equally as driven by habitat preferences and usage variation as smaller scale movements.

In addition to habitat preferences and usage influencing movement, these varying movement scales can in turn influence population and species density, which can have large impacts on survival and fitness. Migration movements among many species can lead to increases in species densities (Kopachena 1991). Living in high density groups can help reduce predation rates (Nass et al. 1984) and increase the likelihood of offspring survival (Savoca et al. 2011). Large groups also create competition for limited resources as well as increase the possibility of catching a disease or parasite from a group member or other nearby species. The spreading of infectious agents can happen more easily in high density nesting and roosting species which spend large amounts of time in their nesting area, such as species that perform hibernation (Kleindorfer and Dudaniec 2009). This can be detrimental to the species and any other susceptible species in the same area (Coggins et al. 1982). An example of the spread of disease via increased density caused by migration is white-nose syndrome in North American bat species, which in recent years has greatly impacted the populations of many densely roosting hibernating species.

Area Description

The Current River State Park is located on 831 acres along the Current River north of Eminence in Shannon County, Missouri (McCarty 2008) (Figure 1). The park was developed in the 1930's and 1940's as a corporate retreat for the Alton Box Board Company. In 2008 the complex and surrounding acreage was transferred to the Department of Natural Resources and currently the park is only open to the public for recreational day use. A Natural Resource Management Plan on Current River State Park was generated in 2008 which outlined the natural

history of the area, identified natural threats to the park, and set objectives for the park's future (McCarty 2008).

The park landscape spans four tributary streams through Dugan, Jones, Broad Shoal, and Slick Shoal hollows with three intervening ridges, and has approximately 1.6 km of Current River frontage (McCarty 2008). The terrain varies in elevation, from the high point at 336 meters above sea level on Bat Cave Ridge, to just over 200 meters where the Current River exits the park boundary (McCarty 2008). A broad spectrum of the Ozark's geology is exposed in the bluffs, rocky outcrop layers, and rocky residuum. The hillsides are mainly oak-lined, with many small dolomite glades, fens, seeps, creeks, rocky outcrops, bluffs, and bottomland forests. There are also regenerating old fields, lands formerly cultivated but later abandoned leading to an intermediate stage in ecological succession, in sections of the hollows (Core 1949; McCarty 2008). Additionally, two natural patches of shortleaf pine are present in the park, while other shortleaf pines have been cultivated near the lodge on an open lawn (McCarty 2008).

The Echo Bluff State Park property is composed of 476 acres along lower Sinking Creek, just above its confluence with Current River (McCarty et al. 2013) (Figure 1). The area was originally developed as a youth camp for girls in 1929, and more recently as a site for music festivals. The Department of Natural Resources acquired the area in late 2013 to develop it into a modern campground and connected this site with Current River State Park (Suntrup 2015). In 2013, before construction could begin on the park, a natural resources assessment was conducted by McCarty et al. 2013, to describe the current state of the park and to make planning considerations for future construction. The park is now a highly developed campground with rentable cabins and 50 km of wilderness trails.

The landscape of Echo Bluff, much like Current River, contains vast vertical relief with its highest elevation of 1690meters above sea level along the southeastern boundary and it is lowest in Sinking Creek at approximately 1100 m (McCarty et al. 2013). Sinking Creek winds through the property in a southwesterly direction, but meanders in all directions at some point throughout the property. The floodplain and adjacent bottoms are made up of mesic forests and riverfront forests typical of the Ozarks, containing mature sycamores, basswoods, walnuts, and thickets of eastern witch hazel, buttonbush, and willow along the stream edge (McCarty et al. 2013). In contrast, the uplands contain dry-mesic to dry natural communities of oak, hickory, scattered portions of short-leaf pine and cedar (McCarty et al. 2013). High vertical cliffs border the creek on its east and west sides and numerous rock escarpments and outcrops are found on steep slopes and ridges. These areas form a variety of natural community types depending upon their placement in the landscape, underlying geology, solar exposure, and moisture availability. The mesic communities thrive in areas with seepage through the rock formations, relatively dense shading, and wet cliffs (McCarty et al. 2013). Glades and dry woodlands occupy hillsides with greater solar exposure and less moisture (McCarty et al. 2013). There are two moderate springs with other smaller seeps and five known caves are located on the property (McCarty et al. 2013).

Sampling Methods

The methodologies used in this research can be separated into passive and active sampling techniques. Passive sampling includes using any evidence of species presence with direct contact or observation of individuals (e.g. using hair snares (Gardner et al. 2010), tracking footfall impressions, and collecting fecal deposits) to identify presence in the area of interest

(Bider 1968; Aulak and Babinska-Werka 1990). Passive sampling also can involve the use of a collecting medium, sometimes in the form of a man-made device that accumulates and stores the desired data without the investigator being present, such as acoustic recorders (Luczkovich et al. 2008), hair snares (Gardner et al. 2010), and camera traps (Silver et al. 2004). Camera traps allow for estimation of species presence and/or abundance by documenting occurrences over long periods of time and through identification of unique individuals through markings (Wemmer et al. 1996; Cutler and Swann 1999). Passive sampling has the benefit of being less obtrusive to the study species, and therefore is less likely to affect their natural behaviors and life cycles. A detriment of passive sampling can be the level of detail in the data. For example, if a study species is detected acoustically at two different times, it may be hard to discern if the data are from one or more individuals.

Camera traps have been used to survey wildlife since the early 20th century (Chapman 1927). And though camera traps have been used for nearly a century (Kucera and Barrett 1993), recent advances in technology, particularly since 2006, there has been an increase in their use in field studies (Rowcliffe and Carbone 2008). The use of camera traps has become more prevalent with the additions of infra-red movement sensors, digital storage, and infra-red flashes for nocturnal pictures.

Common uses for camera traps include, though are not limited to:

(1) faunal surveys/checklists (Rovero and De Luca 2007; Kelly 2008),

(2) detection of elusive or rare species (Carbone et al. 2002; Sanderson and Trolle 2005; Kelly 2008),

(3) generating relative abundance indices via photographic rate (Carbone et al. 2002; Rovero and Marshall 2009; O'Brien et al. 2010),

(4) estimating abundance through Mark-Recapture studies via individually marked or recognizable individuals (Silver et al. 2004; Trolle and Kery 2016),

(5) using Random Encounter Models to estimate density of unmarked and non-recognizable species (Rowcliffe and Carbone 2008; Bunty 2015; Cusack et al. 2015),

(6) generating detection and occupancy models (Linkie et al. 2007; Thorn et al. 2009; Connell et al. 2014),

(7) long term monitoring of communities and populations (O'Brien et al. 2010; Ahumada et al. 2011; Kays et al. 2011),

(8) measuring habitat associations (Linkie et al. 2007; Bater et al. 2011).

Acoustic monitors are actively being developed and promoted for the passive study of bat activity. The advancement in audio recording and implementation of machine learning methods now allow investigators to record high frequency bat echolocation calls and identify them to the species or species group level with high accuracy (up to 94% (Britzke et al. 2011)). Detectors are set up in areas thought to have high bat activity, and they use a microphone sensitive to high frequency sounds (either unidirectional or omnidirectional) to record and store calls in a digital format.

For bat acoustic data, the digital information can be recorded and stored in two different ways, zero crossing or full spectrum. Full spectrum data captures the entire spectrum of sound including intensity, harmonics, and faint sounds during times of large ambient noise (Frick 2013). In contrast, zero crossing files are a smaller file type that represents data as a series of points on a graph with time on the x axis and frequency on the y axis and count the delay between successive zero-crossings of the signal (Agranat 2013). While zero crossing has reduced information, this allows for smaller file sizes and reduces the amount of storage space needed (Frick 2013). Regardless of recording type, they can be identified using a program with computer algorithms generated by machine learning of known call libraries (Agranat 2013). These

algorithms are used to identify the calls to the species level with a corresponding confidence level per call sequence.

In contrast, active sampling involves data collection via direct observations of a subject of interest by an investigator, and can include line transects (Yapp 1956) and trapping (Flowerdew et al. 2004), which can be conducted using live or lethal methods (Mengak and Guynn Jr 1987). Trapping allows the investigator to have the subject in hand and more easily access diagnostic indicators for identification, take anatomical measurements, and/or attach some form of identification or tracking device. Traps can take a variety of forms such as terrestrial box-type live traps (Donald S. Kisiel 1972), pitfall traps (Mengak and Guynn Jr 1987), aquatic traps (Leigh et al. 2016), and mist nets or harp traps for volant/ gliding creatures (Francis 1989).

Active sampling allows the investigators to collect a variety of detailed anatomical data on the study species. Active sampling can have the detrimental effect of changing or inaccurately representing the study species' behavior, movements or life cycle. For example, if an investigator is conducting a line transect and regularly startles the study species along that transect, it may cause individuals to avoid that area even though they would, under other circumstances, be regularly found there. Active sampling is also subject to observer bias (Pollock and Kendall 1987). If observers are measuring the distance from the transect to the observed study species, there could be a difference in distance estimation between two different observers, leading to inaccurate data. Often the use of multiple observation methods or multiple observers can reduce the amount of variance between different individual measurements (Pollock and Kendall 1987).

Mist netting involves closing off a likely flyway with a fine polyester net. As volant animals travel between foraging or roost locations through the flyways they come into contact with the net and become entangled. They can be removed by the investigators, who can take

anatomical measurements along with other relevant data and identify each capture to species using identifiable characteristics.

Methods for Analyses

When accounting for the number of species at a study site, one way to visually represent the acquisition of new species to the dataset over time is via a species accumulation curve. Accumulation curves are used to estimate the total number of species in a given study area (species richness) by evaluating where the curve asymptotes. As the curve of species acquisition flattens, to the point the line begins to asymptote, it indicates that further sampling is unlikely to yield more previously unaccounted for species.

Rarefaction cures allow for the comparison of species richness by visualizing the total number of species found over the process of collecting data. Rarefaction curves estimate the number of species that would be expected in a sample of a smaller size, helping to facilitate dataset comparisons when sampling may be incomplete due to extraneous factors. Rarefaction curves are created by repeatedly sampling from the pool of data multiple times and plotting the average number of species found in each re-sampling creating a plot of the number of species as a function of the number of samples (Gotelli and Colwell 2001). Rarefaction curves are generally created without replacement within each re-sample in order to generate the expected number of species in a collection of samples drawn at random from a large pool of samples (Simberloff 1977). Gotelli and Colwell (2001) noted that as a rarefaction curve progressed, the curve becomes flatter as increasingly rare species are added and the curve will eventually asymptote at the observed species richness. Species accumulation curves represent the number of new species

detected with additional effort. These will asymptote when all species occurring at a site have been detected.

Species richness estimators treat data collected as a sample of the true whole and extrapolate to provide estimates of the true richness of the community. These estimators incorporate the number of infrequently detected species as a measure of uncertainty in the completeness of the sample. If all species are detected frequently, then the estimator will yield the observed number of species. If a subset of species is detected infrequently, then the estimator will yield a value that exceeds the observed number of species. Estimate S is a software package that calculates asymptotic species richness using seven different estimators for the input dataset (Colwell 2013). The software offers estimators such as Chao 1, Chao 2, Abundance-based Coverage Estimator, Incidence-based Coverage Estimator, Jackknife 1, Jackknife 2, and Bootstrap, each with different approaches to account for the frequency of detection.

When documenting species occurrences on a landscape, it is not always possible to detect them even if they are present. Occupancy models were developed to account for uncertainty in detection and occurrence simultaneous using a hierarchical approach (MacKenzie et al. 2017). These models allow for quantifying the effect of the environment on species while accounting for the process of observation. In these models, the study sites are sampled multiple times over a study window when the occupancy state of that site (occupied or unoccupied) is unchanging, and coefficients are estimated in relation to site-specific (habitat, elevation, etc.) and event-specific covariates (temperature, precipitation, etc.) (Gerber et al. 2020).

Occupancy modeling has four basic assumptions:

(1)Sites are closed to changes in occupancy between sampling events.

(2)Occupancy and detection probability are constant across sites and if not, they are properly modeled using site-specific covariates.

(3)Detection is independent at each site (sites are far enough apart to prevent double detection of an individual).

(4)Detections only occur at occupied sites and there are no false positives such as species misidentification (MacKenzie et al. 2017).

LARGE TERRESTRIAL MAMMALS

Introduction

The goal of this study was to produce a mammalian species inventory for two state parks in the Ozark National Scenic Riverways (ONSR): Current River State Park (CRSP) and Echo Bluff State Park (EBSP) and compare them to past surveys, with the expectation that all common species will be captured using camera trap technology (with the exception of those too small to be captured) or observed visually. Both parks are in Shannon County, Missouri and encompass portions of the Current River. Both parks have stark vertical relief, and a variety of Ozark habitat types. CRSP is a former corporate retreat that is only open for day use, while EBSP has undergone a large amount of development in the recent past to become a modernized recreational area. Past surveys on mammalian species in the ONSR has included archaeological studies (Klippel et al. 1987), invasive species studies (Houston and Schreiner 1995), cave surveys (Meyers 1964; Colatskie 2017),a preliminary ecological reconnaissance of terrestrial vertebrates (Hariowicz 1969), a baseline historical and anecdotal mammalian faunal survey (Murray 1991), and several books and guides published on the Ozarks area (Schwartz et al. 2016).

Second, this study describes the mammal species richness of the study area through species accumulation curves and richness estimators, with the expectation that richness will be larger at EBSP due to the increased habitat heterogeneity and less frequent human disturbance. Finally, this study uses camera trap data to discern what abiotic and biotic influences affect species occupancy, detectability, and overall community ecology, with the belief that human-caused disturbances will have the largest influence on occupancy and detectability across all species, while generalist species will be the least affected.

Methods

Camera Traps. In each park, one Moultrie 880 (Moultrie Feeders LLC, Alabaster, AL) and three Reconyx Hyperfire (Reconyx Inc., Holmen, WI) camera traps were deployed to collect data (Figures 2 & 3). All cameras were set to high motion sensitivity to ensure the capture of smaller mammals. Both camera types were equipped with low-glow infrared flashes for capturing low light images that are less startling for nocturnal mammals. Cameras were set to take three images, each one second apart when triggered, so multiple images of each animal could be obtained for accurate identification. Each camera was set for a cool down time of one minute after being triggered to prevent running down the batteries with images of moving foliage or animals that stayed in the camera location.

Cameras were placed on trees approximately 1–1.5m above the ground angled downward (Figure 4) and secured to the trees via a flexible steel cable. Data were downloaded and checked approximately once a month. Cameras were powered with AA batteries which were replaced during each check when total battery percentage was below 40%. Each camera used a32GB SDHC storage card (SanDisk, Milpitas, CA), which was downloaded and cleared at each camera check. Cameras had sardine bait placed within the triggering area at each check to entice carnivorous and omnivorous mammals to the sites.

Cameras were moved only if it was required to avoid losing data or when found to be in areas with high amounts of human use. Three cameras were moved over the course of the study. The initial locations for each of these cameras were not used in the analysis due to small sampling windows (Table 1). At EBSP, the camera labeled Zoe Prairie had to be relocated due to impending bulldozing of the site. Also, at EBSP the camera labeled Behind Barn had an electronic failure and had to be removed to be repaired. At CRSP, the camera labeled Dump

Road was found to be placed along a path highly traveled by visitors and employees of the park and was subsequently moved. Otherwise, cameras were left in place for the duration of the study (Table 1).

Accumulation Curves. Camera data were formatted into a matrix by each day a camera trap was active, and the total number of each mammalian species identified at that camera site, on that camera day/sample. Abundance matrices are commonly used to organize abundance data, by categorizing the number of times an event happened by another relevant metric or combination of metrics (i.e., species of animal, day of the year, etc.).When multiple detections of the same species occurred in a single day/sample, I recorded the total number of observations. Non-native/invasive species were excluded from this analysis (i.e., domestic dogs and feral horses).

Accumulation curves were created in Estimate S software version 9.1.0 (Colwell 2013). I followed Colwell's user guide to choose the parameters to create the resulting accumulation curves. Each camera's data set was run with 100 randomizations, without replacement, for the resulting estimators. Accumulation curves were not extrapolated past the sampling time window of this study with estimation points occurring at each sample/day. Three camera locations that had to be removed due to disturbance or technical failure were not used to create accumulation curves due to their shorter sampling periods (Table 1).

I used the formula for Chao 2 when calculating the asymptotic richness (Colwell 2013) due to multiple samples having varying numbers of species with exactly one occurrence in the sample (*uniques* according to Chao 2), and species with exactly two occurrences in the sample (*duplicates* according to Chao 2) on eight out of the nine camera sites. Chao 2 is calculated as: $S_{2=} {}_{S_{obs}} + \frac{Q_1^2}{2Q_2}$ where S_{obs} is the total number of species in the sample, Q_1 is the number of

uniques and Q_2 represents the number of duplicates in the sample. Due to the similarities among diversity estimators for each camera location I will only be referring to Chao 2. Of these asymptotic richness estimators, I interpreted the data using Chao 2becausethe dataset is made up of occurrence data.

Similarity Indices. Estimate S software also was used to evaluate the similarity of the community detected at each camera location. Total counts of each species at each camera location were coalesced into a matrix. A coverage-based estimator of 10 was used for the upper limit of rare and infrequent species via the suggestion in the Estimate S Users Guide (Colwell 2013).For the camera trap dataset (Table 2), I used the Chao-Jaccard-Raw Abundance-Based Index since it takes abundance into its estimate of similarity with no loss of generality (Chao et al. 2005). The values range from 0.0 to 1.0, with 1.0 signifying locations that are the most similar.

Occupancy Modeling. An occupancy analysis was conducted for all medium/large mammals detected that were seen at more than one site and had more than five total sightings. I created these occupancy analysis models in the Program R (Version 3.2.4) (R Core Team 2013)using the *unmarked* package (Version 0.11-0) (Fiske et al. 2015).There were 10 covariates used to create the models; seven site specific covariates and three sample specific covariates. The seven site specific covariates for each camera trap included the distance to nearest permanent water source, shortest distance to Current River/Sinking Creek, shortest distance to nearest road, and elevation; all of these were measured in Google Earth v 7.1.2.2041 (Google Inc. 2013). All of the previously mentioned continuous variables were standardized and centered using Program R for model optimization. The three other qualitative site-specific covariates were the state park, the brand of camera, and the habitat type (Table 3). Habitat types were determined using the

Camp Zoe Initial Natural Resource Assessment (McCarty et al. 2013) and the Current River State Park Natural Resources Management Plan (McCarty 2008). Due to the high diversity of habitat types within each park, habitat diversity was reduced to 3 categories in order to make them comparable between parks within the models: Habitat Class 1=Dry Mesic Bottomland, Habitat Class 2=Dry Mesic Forest, and Habitat Class 3=Dry Chert. These classifications were chosen to highlight differences in terrestrial environments based on elevation, plant communities, and association with moisture.

I created three sample covariates: mean daily temperature, daily amount of precipitation, and the Ordinal day (origin = January 1, 2014). Weather associated covariates: daily precipitation and daily mean temperature, were downloaded from PRISM Climate Group (Oregon State University 2004) using interpolated data at a 4km resolution. All continuous weather data were normalized for model optimization using R. Each species data set was run through the 12 detection models created (Table 4) and sorted via the resulting AIC scores, with the lowest scores denoting the most supported models. Each of the top supported detection models, (p), characterized by a ΔAIC value within two of the top supported model, were placed in six occupancy (Ψ) models (Table 4). All model AIC values were compared; the top supported model and those with a ΔAIC score within two were reported. Differing methods for the Markov Chain Monte Carlo (MCMC) random walk were used between species, due to some models returning a singular Hessian value under some MCMC methods. The MCMC method used with each species' top supported models can be seen in Table 5. Two species, white-tailed deer (*Odocoileus virginianus*) and coyote, (*Canis latrans*) were detected at all sites, making an occupancy analysis unnecessary. For these two species, generalized linear models were created

in R and used to discern factors affecting detection of these species using the same covariates as the other species (Table 5).

Results

Mammalian Inventory. A total of 11 indigenous mammalian species were captured on camera traps across both parks: coyote, *Canis latrans*; nine-banded armadillo, *Dasypus novemcinctus*; Virginia opossum, *Didelphis virginiana*; bobcat, *Lynx rufus*; striped skunk, *Mephitis mephitis*; white-tailed deer, *Odocoileus virginianus*; raccoon, *Procyon lotor*; eastern gray squirrel, *Sciurus carolinensis*; fox squirrel, *Sciurus niger*; eastern cottontail, *Sylvilagus floridanus*; and gray fox, *Urocyon cinereoargenteus*. All species were captured in both parks with the exception of eastern chipmunks (Table 2). Eastern chipmunks (*Tamias striatus*) were absent from camera trap data at EBSP though they were documented at both parks via observation during the length of this study. Other mammalian species were visually documented but absent from camera data including groundhogs (*Marmota monax*) which were observed at both parks, and beaver (*Castor canadensis*), which were observed at CRSP. These three species were omitted from analyses because they were not captured in the camera trap data. Other observations of interest include a melanistic coyote seen at multiple locations within CRSP and a herd of seven to nine feral horses moving between parks.

Accumulation Curves. Accumulation curves were generated for a total of nine camera locations, four in CRSP and five in EBSP. The CRSP cameras had sampling windows ranging from 539 days to 563 days, with the smaller value being due to a 24-day power failure on the Current River Moultrie camera. The EBSP cameras had a broader range of sampling day values, 281-561 days, due to cameras being moved or lost due to construction. The Moultrie camera at

EBSP was present for approximately 280 days before it was lost in a large bulldozing effort in the area. The camera was destroyed, and the data were lost along with the camera. The data for this camera location still were used in the analysis because the sampling time was large enough to yield a robust dataset.

Seven of the nine accumulation curves show signs of reaching an asymptote and have tight 95% confidence intervals (Figure 5). Two cameras, Cheyenne and South Road, in Current River showed broadening confidence intervals at the end of their sampling period. All cameras have an associated estimate of species richness (Table 6). The estimated richness values for each camera match the observed species count, with the exception of two cameras whose resulting estimators both were larger than the observed species by a value of two.

Similarity Indices. The table of observed shared species (Table 7) shows that all camera locations share at least two species: white-tailed deer and coyote. These two species were the most numerous and were seen across all sites. The largest number of species shared between two sites was eight (Table 2). The largest number of total animal sightings was 285, and this site shared eight out of the 10 species it captured with the next most abundant site, which had 121 sightings (Table 3). Chao-Jaccard-Raw Abundance-based estimator values range from 0.349 (N Zoe Glade-N Zoe Pond) to 0.976 (N Zoe Glade- Moultrie Zoe). Most comparisons resulted in high similarity estimators with only three of the 36 comparisons being less than 0.6, and 10 estimators having a value greater than 0.9 (Table 8).

Occupancy Modeling. . Occupancy models were created from seven species' data sets (Table 5), resulting in 21 qualifying models (ΔAIC ≤ 2). Two of the species, armadillo and gray fox, had a singular qualifying model, while the five other species had at least two models meet this criteria. All reported models have a corresponding model weight which quantifies the

explanatory value of the model on the variance within each species data set. The corresponding model weights ranged in value from 0.05 to 0.9, with high weights explaining more of the variance within each species. Most of the resulting models are rather weak, with only three out of the 21 having model weights above 0.4. As stated previously, white-tailed deer and coyotes were found at all camera locations and thus were both analyzed using a generalized linear model using the same covariates as the occupancy models. This yielded just a single supported model for each of these two species, with no corresponding model weights for either (Table 5).

Discussion

Mammalian Inventory. The camera trap data captured the majority of expected species in the area, supporting the hypothesis that most common mammals would be captured, but with a few exceptions. There are seven medium to large mammals that are known to occur in Shannon county but were absent from the camera data either due to having highly dispersed populations or cameras not being in ideal habitats to capture them. These include black bear, *Ursus americanus*; wapiti, *Cervis canadensis*; long-tailed weasel, *Mustela frenata*; mink, *Neovison vison*; muskrat, *Ondatra zibethicus*; North American river otter, *Lontra canadensis*; spotted skunk, *Spilogale putorius* and the swamp rabbit, *Sylvilagus aquaticus*. The swamp rabbit is historically rare in the county along with American badgers, *Taxidea taxus*; and mountain lions, *Puma concolor*; (Schwartz et al. 2016). These species also were absent from the camera data.

Many of the unaccounted for species have strong habitat associations with water (mink, muskrat, and river otter) (Schwartz et al. 2016). Many of the cameras were near large bodies of permanent water (Table 3), but these species were still absent from the camera data. Most of the cameras that are near permanent water (<100m) are near ponds and other standing water whereas

only one camera (Mid Zoe) was within 100m of flowing water (Table 3). The lack of proximity of the cameras to an important habitat criterion for these species could have led to their absence in the dataset.

Wapiti have recently been reintroduced to Shannon County starting in 2011-12 with 108 having been brought from Kentucky to Peck Ranch Conservation Area (MDC 2018). There is currently a large enough population for the conservation department to allow a limited number of elk to be harvested per season. Due to the species being recently reintroduced and highly monitored, it is unlikely one would have been captured on the trail cameras. As the population grows and disperses, they will potentially become more common in these parks.

The remaining species that are known to occur but were not captured on camera data (black bear, long-tailed weasel, and spotted skunk) are rarely encountered species in Missouri (Schwartz et al. 2016). Camera-trap success rates for spotted skunks are generally low (<0.5%) (Lombardi et al. 2016; Wilson et al. 2016; Higdon and Gompper 2020). The spotted skunk is listed as endangered and a species of conservation concern within Missouri by the Missouri Department of Conservation.

Black bears have large home ranges with females having ranges over 52 sq km and males over 259 sq km that often overlap with multiple female home ranges (Schwartz et al. 2016). With such large ranges it is possible that multiple bears use the study area as part of their home range but were never captured on camera, or that they spend more time in different areas of their home range. Black bears were nearly extirpated from the state but are currently becoming more abundant after recovery efforts began in Arkansas in 1959. A recent project estimated the current population in Missouri to be between 540–840 bears with a growth rate of around 9% annually (Missouri Dept. of Conservation 2019). The large home ranges and the low population size likely

explain the lack of black bear images in the data set. As populations continue to increase within the state the likelihood of observing bears in these parks will increase.

Accumulation Curves. Looking at Figure 5, as time passes all plots rise and then show signs of beginning to asymptote. As each camera's accumulation curve begins to flatten, varying levels of species richness become apparent. Richness values range from three species (By Creek and N of Pond) up to10 species (Cheyenne). These results did not support the prediction that EBSP would have more richness than CRSP as the average species richness for CRSP was 6.25 species, while the average for EBSP was 5.6 species. It is possible that CRSP had greater diversity due to having habitats that are preferred by more species. I think the more likely reason was the encroachment of construction into the park that pushed larger mammals away from EBSP, possibly even towards CRSP.

The cameras whose plots did not asymptote either have a shorter sampling window than the other camera locations (Moultrie-Zoe) or still have wide 95% confidence intervals (Cheyenne and S Road). Additionally, two sites had estimated species richness values larger than the observed species. This implies that rare species may have been missed during the survey period. Because the total detected species richness seen across both parks is 13 species, and no single camera observed more than 10 species, it is entirely possible that the true species richness at both sites was larger than observed. It is possible that species were not detected because the survey period was not long enough or the cameras' failed to capture images of them because of position or random chance or a combination of both.

The wide confidence intervals for Cheyenne and S Road were the result of two incidences of uniques and a single duplicate. The idea behind Chao 2 is that as a community of species is being sampled and rare species are continually being discovered, so the values suggest

rare species have yet to be documented. Once all rare species have been recovered at least twice (duplicates) then it is unlikely that a new species will be uncovered. The aforementioned data would yield the value of the observed species richness increased by a value of two (Table 6). When the two single encounters of a species (uniques) are hypothetically increased to two encounters per species (duplicates) the confidence intervals close tightly around the ends of the accumulation curves, and species diversity estimators are the same as the observed values (Figure 6).

Similarity Indices. Only one species (chipmunk) was found at a single site (Cheyenne), all other species were found at more than one site. Since all of the camera sites shared at least two species and most shared three or more while also having relatively high abundance for many of the more common species (deer, coyote and gray squirrel) it is not surprising that overall similarity estimates are high, with only two site comparisons being below 0.5 (Table 8). The disparity in abundance values per species (Table 2) is likely the explanation for the difference in estimated similarity values when the observed species richness is the same since the Chao-Jaccard Raw Abundance-based Index takes abundance into account without loss of generality. The high species similarity indices do not support the prediction that there would be low similarity due to the high human disturbance at EBSP compared to CRSP.

Occupancy Modeling. Occupancy models were created for seven species, and generalized linear models were created for two species that were found at all sites. Qualifying models are those with a ΔAIC of 2.0 or less and have the most explanative value. Armadillos had a single qualifying model, p(Habitat Class) Ψ(Distance To Water) (Table 5). The occupancy covariate distance to water (Estimate= 9.0) is not significant (P=0.459) and has a large standard error (SE= 12.16) indicating little explanatory value. The detection covariate for habitat class has

all negative estimates for each of the three habitat types (mesic bottomland= -3.43, dry mesic forest = -4.5, and dry chert= -2.37). Mesic bottomland and dry chert have significant *P*-values ($P < 0.001$) while dry mesic forest has a nonsignificant *P*-value ($P=0.3$) and larger standard error ($SE=5.12$) due to lack of armadillo detections in this habitat class, making comparisons difficult. This results in all 3 habitat classes, even mesic bottomland and dry chert, having very low detectability overall, though it could be said that armadillos have the largest detectability in mesic bottomland, followed by dry chert.

Bobcats had two qualifying models, with the top model being: p(Distance To River) Ψ(Distance To River). The detection coefficient in the model for distance to river (Estimate= 5.33) has a significant *P*-value ($P=0.0495$) indicating that there is a positive association with distance to river and detectability. The occupancy coefficient in the model had a large estimate (Estimate= -18.83) and standard error ($SE= 69.6$) resulting in a nonsignificant *P*-value ($P=0.787$). The second model: p(Distance To River) Ψ(Distance To Road) has little information to add since the occupancy coefficient (Estimate= -30.154) is not significant ($P=0.425$) and the detection coefficient is the same as the top model (Estimate= 5.72, $P=0.71$). Overall bobcats are more likely to be detected closer to rivers, though we cannot say what factors are influencing occupancy. This likely is due to low occurrence rates (Table 2) and the general reclusive nature of bobcats.

Six models were within the Δ2 AIC units for the eastern cottontail which cumulatively account for around 55% of the total variance of the data set (calculated via summation of the model weights) (Table 5). This is likely due to the few images captured of cottontails, only 12 in total (Table 2).It could be that all of the covariate conditions are equally favorable for cottontails or they are responding to conditions within or independent of the chosen covariate categories.

None of our covariates give suggestions as to what areas cottontails are occupying or give suggestions for increasing detections.

The gray fox had a single supported model: p(Mean Temp.) Ψ(Elevation), with a high model weight (wt=0.90) (Table 5). Similar to the bobcat top model, our occupancy covariate (elevation) (Estimate= 19.6) was not significant (P=0.497) and has a high standard error making it non-explanatory. The detection covariate for mean daily temperature (Estimate= -0.92) had a significant P-value (P<0.001) indicating that gray foxes are more detectable at lower temperatures. Gray foxes are mostly nocturnal (Schwartz et al. 2016) which may have led to them being active during lower temperatures. It also possible that gray foxes were more detectable during seasonally lower temperatures due to less foliage presence preventing camera detection, though this seems unlikely since this trend has not been seen in any other species.

The gray squirrel had two qualifying models both of which only took detection into account, meaning factors affecting occupancy could not be determined by this dataset (Table 5). Looking at the top qualifying model's detection covariates, both mean daily temperature and Ordinal date have very low estimates (Mean Temperature = -0.64424, Ordinal Day = 0.00415) and standard errors (Mean Temperature SE= 0.08526, Ordinal Day SE= 0.00068), and both have significant P-values (P<0.0001).There is likely a correlation between the time of year and mean temperature so it is not surprising that they would both be significant. This implies that overall detection was easier during some parts of the year than others. While mean daily temperature and Ordinal day do influence detectability, their effect is so little that it is no different from the null model: p(.) Ψ(.), which also had a significant detection value (Estimate= -2.92, P<0.0001). The second qualifying model, p(Distance To Water) Ψ(.), had a significant P-value (P<0.0001) and a

small estimate (Estimate= 0.712), indicating that gray squirrels are more likely to be detected closer to water, but once again this is not much more informative than the null model.

The top model for opossum was: p(.) Ψ(Elevation), followed closely by the null model: p(.) Ψ(.) with a ΔAIC of 0.91. Even with the top model including elevation as a covariate for occupancy (Estimate= 11.73) it is nonsignificant (P=0.526).The third qualifying model had Ordinal day as a detectability covariate, p(Ordinal) Ψ(.), also had a nonsignificant P-value (P=0.366) and a small estimate for detectability (Estimate=0.00268). The two qualifying models with covariates have low model weight (Table 5). Neither of these models were different (within Δ2 AIC) from the null model. Therefore, none of the covariates used for the opossum data were explanatory, suggesting that some other unmeasured factor or combination of factors is likely affecting their occupancy and detectability.

Raccoon had five qualifying models all with parkas the covariate for detection (Table 5). The top model, p(Park) Ψ(Distance to River), was uninformative when it comes to occupancy (Estimate=-1.25, P=0.212), but has a significant detectability factor (Estimate=0.599, P=0.00017).The model with park as the only factor in detectability, p(Park) Ψ(.), had a low estimate (Estimate= 0.58) and was significant (P=0.00013). Occupancy factors were uninformative overall, but detectability in park was significant in all models. This is likely due to two cameras at Current River never detecting raccoons (Table 2). Once again, occupancy is uninformative in these models, but raccoons were more easily detected at EBSP.

Two species, white-tailed deer and coyote, were detected at all sites and therefore they cannot use the occupancy modeling framework. I used a generalized linear model to discern factors affecting detection, because they both occupy all sites. The coyote model with the lowest AIC score (*d.f.*=6) had significant results for detectability between parks (P<0.001,

Estimate=1.6128, z-value= 126.2) with detection being larger at CRSP, suggesting that coyotes avoided the construction at EBSP or at least were more secretive if they were present at EBSP. There also were significant results for detection in the dry chert habitat type (P<0.001, Estimate= 0.0799, z-value= 3.33), suggesting coyotes were detected more often in that habitat type. The interaction between the dry chert habitat and park was also significant (P<0.001, Estimate= -2.115, z-value= -5.56) indicating that their ability to be detected in the dry chert habitat cannot be interpreted independently from the parks.

The white-tailed deer detection model (*d.f.*=6) had significant values for detectability between habitat types with detection being lower in the dry mesic forest habitat class (P=<0.001, Estimate= -1.8, z-value= -0.711). There was also a significant value for the interaction between park and the dry mesic forest habitat class (P=<0.001, Estimate= 1.753, z-value= 3.71). Similar to the coyote detection model, habitat class and the interaction between habitat class and park have small estimates making these factors significant but with a small effect on detection. This implies that white-tailed deer may not be deterred by the construction at EBSP.

Most of the occupancy models were not informative, making comparisons difficult, but many of the detectability models were informative. The predictions that generalist species would be less affected by the anthropogenic impacts of EBSP were mostly supported with white-tailed deer and raccoons both showing increased detectability at EBSP than CRSP. Coyotes, despite being a generalist species, were less likely to be detected at EBSP which was not expected. The other species in this study did not have park as a factor in their top models.

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BAT ACTIVITY AND CAPTURES

Introduction

Bats (order Chiroptera) are the only volant mammals and are the second most diverse mammalian order, with more than 1,386 extant species (Burgin et al. 2018). They have evolved to take advantage of a diverse array of food sources, including insects, nectar, fruit, small fish, amphibians, and blood (Kunz et al. 2011). Bats can echolocate for navigation and foraging via the bounce back and interpretation of ultrasonic calls. This is especially effective because most species of bats are nocturnal and roost in a variety of complex structures during the day such as caves, foliage, dead trees, and man-made structures (Jones et al. 2009).

Monitoring bat populations over time is essential for making land management decisions that do not negatively impact various species. Due to their seasonal movements, from winter hibernacula to summer roosts, human impacts at either of these high density habitats could have large consequences for the disturbed bats (Johnson et al. 1998). One example of detrimental land management practices is the increased use of wind turbines. As wind-energy has become a more viable source of renewable energy there is increasing evidence of negative impacts to bat populations related to the mortality caused by flying near the rotating blades (Jones et al. 2009; Frick et al. 2017).

Bats in North America are particularly in need of additional monitoring due to the spread of white-nose syndrome across the continent (USFWS 2017). White-nose syndrome (WNS) is a potentially lethal fungal infection caused by the fungus *Pseudogymnoascus destructans* (*Pd*) (Minnis and Lindner 2013) and was first documented in the United States in a photograph taken in Howe's Cave in New York in early 2006 (Blehert et al. 2009). As of December 2019, WNS

has been confirmed in 34 U.S. states and seven Canadian provinces, with five more U.S. states having confirmed the presence of the causal fungus, but not having confirmed any bat mortality (Whitenosesyndrome.org 2019a). In Missouri, Shannon County was one of the first two counties to have occurrences of WNS in 2009, with over 45 counties now reporting the presence of the fungus (Whitenosesyndrome.org 2019b). The fungal growth, which occurs while bats are in a torpid state, during the winter results in the spread of fuzzy white fungal mats across the muzzles and wings. Bats afflicted with the syndrome arouse more often than normal during winter months, and exhibit odd behaviors such as flying outside during daylight hours and clustering near the entrances of caves (USFWS 2017). The increased activity during a usually dormant period causes depletion of energy reserves needed to survive winter torpor and contributes to increased mortality rates. WNS has killed an estimated 5.7- 6.7 million bats in the affected regions, with some sites seeing die-offs of 90-100% (USFWS 2017). It is likely that millions more have been killed as a result of the disease in more recent years. *Pd* is continuing to spread, both from bat to bat, from bat to cave, and from cave to bat, likely using bats as the principal vector (USFWS 2010; Maher et al. 2012). Tissues damaged by the fungus show scaring that can be identified in the following spring and summer. It presents as pale spots on the wings when they are backlit or as obvious holes in the membrane. A wing damage score has been established to categorize the damage by severity ranging from 0 (no visible damage) to 3 (severe wing damage) (Reichard and Kunz 2009).

The intent of this study was to create a bat species inventory for two state parks in the Ozark National Scenic Riverways (ONSR): Current River State Park (CRSP) and Echo Bluff State Park (EBSP). Both parks are in Shannon County, Missouri and encompass portions of the ONSR. Both parks have stark vertical relief and a variety of Ozark habitat types. CRSP is a

former corporate retreat that is only open for day use, while EBSP has undergone development in the recent past to become a modernized recreational area. As part of this study I have included a measure of bat activity in the area, both generally and species specific, using two long-term passive acoustic detectors at each park. Acoustic data were collected for 20 months and analyzed using identification software. Mist netting was conducted at both parks, outside of the winter months, to confirm species presence in the area. Two primary predictions for this study include; the expectation that due to the seasonal variation in bat activity in the area, acoustic activity will vary throughout the year with a clear minimum occurring during the winter months, and that gray bats (*Myotis grisescens*) will represent the majority of identifiable species activity in the area due to the proximity of a hibernaculum within CRSP and their high resistance to WNS.

Methods

Acoustic Calls. Two Anabat (Titley Electronics, Ballina, New South Wales, Australia) echolocation detectors were placed at each park, with one detector in each park placed near a pond and the other near a river/creek (Figures 7 & 8). One or two solar panels (SunWize Technologies, Inc., San Jose, California) were attached to provide additional power to the units, which were powered at night by one or two 12 volt batteries (Batteries Plus LLC, Hartland, WI).

Acoustic detectors were set to record bat echolocation calls from 20:00 each night until 06:00 the next morning, with the intention of recording during the most active times for bats. Each detector was placed on a tree stand at least four feet off the ground with a directional cone on the microphone aimed upward at a 45°angle. Hunting seats were used to support the weather proof cases used to protect the detectors (Pelican Products Inc. Torrance, CA) (Figure 9).The sensitivity setting used for each acoustic detector was site and seasonally specific, based on the

amount of ambient noise, and ranged from five (in warmer months, when insect and other ambient noise is more present) to seven (in colder months, when insect and other ambient noise was less prevalent). The division ratio was set to 16, a commonly used setting in the Midwest region due to bat species present having higher frequency calls (Messian 2002).

All detectors were equipped with a two or four gigabyte compound flash (CF) card for storing recorded calls. All detector data were downloaded from the CF cards during each visit to a site, totaling 16 download events. Call data were converted into individual time stamped call files using CFread software (Titley Scientific, Ballina, New South Wales, Australia). Call sequences were identified to species level by comparing call structure to known species calls using Bat Call Identification (BCID) software (BCID Kansas City, Missouri). Calls identified to a species contained a minimum of five echolocation pulses to ensure identification accuracy by providing the software with enough data to ensure an accurate identification (Allen et al. 2008). When estimating general bat activity, a minimum of two pulses were used, to help discern between ambient noise and bat echolocation calls (Allen et al. 2008).

Multiple issues led to instances of detectors malfunctioning, resulting in some loss of data. The main issue encountered was loss of power to the units due to foliage blocking the solar panels, causing the batteries to fully drain before they could be replaced during the next site visit. The other main contributor to data loss came from corrupted CF cards, which were not discovered until downloaded. One of the detector set-ups (by the large pond in southern EBSP) was removed on February 14, 2015 due to encroaching construction near the site. This detector was not in use for two months before it was reestablished on May 1, 2015, in another upland area near a small ephemeral pond on the southwestern side of the property. Another cause of data loss was ants building a nest inside a detector and its weatherproof box (Figure 10). Detectors were

nonfunctional for a total of 1,313 nights over the study period. A total of 1,109 detector nights were completed with 488 at CRSP and 621 detector nights at EBSP (Table 9). Species with total call numbers >22,000 over the course of the study were graphed individually for comparison.

Mist Netting. There is some debate surrounding the level of certainty in identification to the species level of bat echolocation calls due to variation in the quality of calls being identified and the type of algorithm used in the identification software (Mirzaei et al. 2011). To validate identifications made via echolocation identification software, I used mist nets to physically identify bat species found at the study site. A minimum of two mist nets were established per netting night, though at some locations an additional net was set when suitable locations were available (Figures 7 & 8). Mist net locations included water sources and suspected flyways with canopy cover, as these areas are known for successful captures (Carroll et al. 2002). A range of mist net lengths (4m, 6m, 8m, or 12m) were used to cover as much of each potential flyway as possible. Mist nets were 2.6 meters tall and were used in set-ups ranging from a single net to three nets stacked one on another for a maximum height of 7.8m.If the ambient temperature dropped too low for bat activity (<10°C) or if it began to precipitate, nets were closed as recommended by the United States Fish and Wildlife Service Summer Survey Guidelines (USFWS 2016a). Mist nets were set-up during the day and opened approximately 30 minutes before sunset each night and remained open for five hours or as long as weather and temperature permitted (Figure 11). A total of 82 net nights were conducted over the course of the study (Table 10).

During mist netting nights, researchers remained within a five minute walk of each mist net. Each net was checked for captured individuals at approximately 10 minute intervals, and any captured bats were immediately removed for measurements and identification. Bats were

transported to processing tables in individual cloth bags to reduce the stress of transport. All bats were released at the capture site immediately after handling and attaining anatomical measurements for identification. The location of capture, net of capture, and time of capture were recorded for each bat. Recorded anatomical data included species, sex, age, reproductive status, mass, forearm length, and white-nose wing damage scores (0-3: Reichard and Kunz 2009).

All researchers practiced WNS disinfection protocols in accordance with the USFWS National Decontamination Protocol (USFWS 2016b). All captured individuals in the genus *Myotis* were banded on their forearm with 2.6mm aluminum clamp band with the identifying number beginning with *BRR* (placed on *Myotis grisescens*) or *INB* (placed on *Myotis sodalis*),for future study of migratory patterns. All other bat species were marked on their forearm with a sharpie, alternating between left and right on sequential nights, in order to identify any recaptured bats from the same evening. This project was approved by the MSU IACUC (14-036) and all bats captured were reported to USFWS and Missouri Department of Conservation and were handled following guidelines from the American Society of Mammalogists (Sikes and Gannon 2011).

Results

Mammalian Inventory. A total of 11 species were documented via the acoustic detectors at both parks (big brown bat, *Eptesicus fuscus*; eastern red bat, *Lasiurus borealis*; hoary bat, *Lasiurus cinereus*; silver-haired bat, *Lasionycteris noctivagans*; gray bat, *Myotis grisescens*; eastern small-footed bat, *Myotis leibii*; little brown bat, *Myotis lucifugus*; Indiana bat, *Myotis sodalis*; northern long-eared bat, *Myotis septentrionalis*; evening bat, *Nycticeius humeralis*; and tri-colored bat, *Perimyotis subflavus*). During the netting portion of this study there were no

physical captures of the little brown bat (*Myotis lucifugus*) or eastern small-footed (*Myotis leibii*) bat at either park. There were no physical captures of Indiana bats at CRSP, and no northern long-eared bats, hoary bats, or silver-haired bats were physically captured at EBSP. Southern flying squirrels (*Glaucomys volans*) were netted at EBSP during the pre-construction survey.

Bat Activity. There were 428,938 acoustic files collected over the course of the study. After the data were scrubbed of ambient noise and unidentifiable calls, 105,222 identifiable bat calls remained (Table 11). Both detectors at CRSP had more identifiable calls than the EBSP detectors, with the CRSP Pond detector recording the most calls (44,901) despite having half as many sampling nights as the other detectors (Table 10).

Eastern red bats, gray bats, and tri-colored bat calls were the most commonly identified bats over the course of the study, with all others being nearly an order of magnitude less frequent. Big brown bat calls were detected more often at creek sites than pond sites. Gray bat calls were much more abundant at CRSP along with most other species call numbers. Notable exceptions are northern long-eared bat calls being most abundant at EBSP Pond, and tri-colored bat calls being most abundant at EBSP Creek.

Examining raw bat call abundance over time shows a stark decline in activity during the winter months (Figure 12).The highest peaks of activity were approximately 1,500 calls per night at CRSP Creek in May of 2014 and at EBSP Creek in September of 2014. While CRSP Pond had the fewest sampling days, it had consistently heavy bat activity during the time periods it was functioning. The opposite is true for EBSP Pond which never had a night with more than 500 calls. CRSP Pond also has the largest average number of calls per detector day (1906.04) followed by EBSP Pond and EBSP Creek detectors coming in at similar averages (966.65 and

970.35 respectively) and finally CRSP Creek (937.60). Each detector had varying peaks throughout the year, though various gaps in the data make it difficult to compare these data.

At CRSP Pond, gray bats made up the vast majority of the identified calls over the entire study window, with the sole exception of late March of 2015 where both tri-colored bats and red bats had more activity (though gray bats were still quite active) (Figure 13). While at CRSP Creek, gray bats were the most active during all portions of the study, compared to the red and tri-color activity (Figure 14). EBSP Pond had comparable numbers of red bat and gray bat calls (2349 and 3049 respectively) with both species being active at similar times of year (Figure 15). EBSP Creek had the lowest gray bat activity with tri-colored and red bat calls being nearly equal or more abundant during the entirety of the study.

Discussion

Mammalian Inventory. All of the bat species that have been known to occur in Shannon County were identified during this study within at least one of the sampling methods (Boyles et al. 2009), including all threatened and endangered species in Shannon county (South Central Ozark Counsel of Governments 2018). Eastern small-footed bats were not captured via mist netting and had low acoustic call abundance across all sites (Table 11). South central Missouri is on the northern edge of the range for the species so it is not surprising that they are rarely captured in the area (Schwartz et al. 2016). Southeastern bats (*Myotis austroriparius*), Seminole bats (*Lasiurus seminolus*), Rafinesque's big-eared bats (*Corynorhinus rafinesquii*), Ozark big-eared bats (*Corynorhinus townsendii ingens*), and Brazilian free-tailed bats (*Tadarida brasiliensis*) have all been captured within the state though not in Shannon County and were not documented during this study.

Bat Activity. Issues with data continuity made the overall comparisons of the detector sites difficult, however a few key observations can be made. We can see a decline in activity in the winter months with a resurgence of activity in the spring following the pattern of hibernation with minimal arousal by overwintering bats (Figure 12).This supports the prediction that activity would be at a minimum during the winter months due to migration and torpor. Both of the creek sites had peak call abundance spiking around 1,500 calls, with CRSP Creek peaking in May of 2014 and EBSP Creek in late-August/early-September of 2014. The timing of these peaks was caused by the differing species abundances at each site. The majority of the peak at CRSP Creek is gray bat activity, as they are foraging during maternity season in June, and the majority of the peak at EBSP Creek is attributed to tri-colored bat activity while they are in the middle of their mating season in September (Boyles et al. 2009).

Both detectors at CRSP had larger numbers of calls that were able to be identified than the detectors at EBSP (Table 11), even with CRSP Pond having the shortest sampling time (Table10). Gray bats were more active at both CRSP sites compared to EBSP and were the most active species at all detector sites with the exception of EBSP Creek where tri-colored bats were most active overall. The results showed that the majority of identifiable calls at CRSP were gray bats, while tri-colored bats were only more abundant at EBSP; this does not support the prediction that gray bats would be the most abundant species in both parks. The known gray bat hibernaculum just off of the CRSP property had 15,000 gray bats as of 2016 (Colatskie 2017) and highly contributed to their presence in the park. Another factor that could have influenced the greater abundance of gray bat activity at CRSP is their resistance to impacts of white-nose syndrome. WNS has greatly impacted cave hibernating bat populations in North America since

2001, but seems to have little if any negative effects on gray bats (Powers et al. 2016; USFWS 2017).

The gray bat is a cave obligate species and are highly selective of their cave locations, generally choosing the coldest caves in their range (Tuttle 1976a; Decher and Choate 1995). Gray bats often forage in riparian areas, where they will often fly over bodies of water below the forest canopy (Tuttle 1976b). This describes all of the detector locations at both EBSP and CRSP; however, the detectors at CRSP were much closer to the hibernaculum and therefore captured more gray bat calls. Of the two CRSP detectors, the CRSP Pond detector had more gray bat calls than the CRSP Creek detector, even though the CRSP Creek detector was closer to Current River, the hibernaculum, and had more sampling days. This result was surprising but may be due to more insect activity at the CRSP Pond location, which may have attracted more individuals to that location. This was not measured during the study, but we can potentially infer insect activity by the number of noise files filtered out at the location. Call files filtered out as noise are any recorded files that do not fit the acoustic signature of a bat call. They are often high frequency interference from babbling water, nearby insect calls or any other non-bat source of high frequency sound. CRSP Pond had the largest number of files categorized as noise despite having no moving water in the vicinity, and therefore can be inferred to be due to more insect activity in the area, thereby attracting more bats to the area to forage. This could be the driving force behind overall activity at CRSP Pond despite such a short sampling period. Gray bats made up the majority of calls while the detector was functioning with a large peak in October of 2014 likely tied to swarming behavior prior to hibernation (Boyles et al. 2009) (Figure 13). There were occasional days where eastern red bat or tri-colored bat call abundance exceeded gray bat call abundance in the spring of 2015, but it is difficult to discern a larger trend from the data.

Indiana bats and big brown bats had the largest abundance at CRSP Creek (Table 11). The Indiana call abundance was greater at both detectors at CRSP than those at EBSP (Figure 14). Indiana bats are known to associate with gray bats while in hibernacula in Missouri and often compete for similar foraging areas (LaVal et al. 1977; Decher and Choate 1995). As of 2017, approximately 300 Indiana bats were counted in the hibernaculum close to CRSP (Colatskie 2017). Big brown bat calls were more abundant at the creek sites of both parks compared to the pond sites. This was surprising since big brown bats are considered generalists and show no real preference for over-water versus over-land or edge versus non-edge habitats for foraging (Kurta and Baker 1990). This result could be due to more big brown bats using the creeks as travel corridors too but is hard to discern without a more robust dataset.

The lowest number of calls for a species overall was the eastern small-footed bat with just over 100 calls and no more than 64 calls at any single location (Table 11). Shannon County is also close to the northern edge of the range for the species, likely contributing to their low representation in the dataset. This species has since been captured during the fall swarming period at the hibernaculum near CRSP.

EBSP Creek had the largest abundance of tri-colored bat calls by nearly twice that of any other site (Table 11). Tri-colored bats are known to primarily forage over water, but this does not explain why they were more active at EBSP Creek than at the other detectors. Looking at the calls over time we can see that the majority of tri-colored bat calls came during September of both 2014 and 2015 (Figure 15). This is the timing of swarming behavior prior to hibernation, which occurs typically from late August through mid-October (Boyles et al. 2009). There are five small caves on the EBSP property along the river bluff, which were potential roosting areas for these individuals (McCarty et al. 2013). Tri-colored bats are less choosy than other bat

species and overwinter in many caves across the state, but prefer smaller caves (Boyles et al. 2009). I believe that these small caves may be overwintering sites or even just short term stopover sites during the swarming season for tri-colored bats. This would lead to the increased activity at this detector due to its proximity to the caves and the time of year.

EBSP Pond had most of its documented calls in 2014, with comparable numbers of red bats and gray bats over the course of the study (Figure 16). EBSP Pond had both the least amount of activity overall and the lowest counts of each species except northern long-eared bats (Table 11). As mentioned previously, this detector was removed due to construction, and relocated to a different pond on site. Prior to moving the detector (March 9, 2014 – February 14, 2015) there were 64 calls identified as northern long-eared bats. After the detector was moved to an upland pond (May 1, 2015 – November 15, 2015) there were 255 calls identified as northern long-eared bats. The two ponds are quite different with the first being a large open pond with very few trees on its shore and open fields around it, while the second pond is much smaller with trees surrounding it. While it is possible that northern long-eared bats were much more active around the second pond as opposed to the first, I believe the explanation lies in the acoustic plasticity of the bat species in the area, and the potential misidentification of calls. Bats have the ability to shift their echolocation calls to higher frequencies in areas of high clutter. These higher frequencies do not travel as far but provide the bat with greater detail about its surroundings. Other species in the genus *Myotis* have been recorded echolocating in cluttered habitats at a greater frequency than is typical for them, which can often lead to them being misidentified as northern long-eared bats which typically have high frequency calls. I believe this is what occurred at the second pond; the increased clutter caused other species of bats to alter their

typical call frequencies to better move within the cluttered habitat, which in turn caused their calls to be mistakenly identified as northern long-eared calls by the identification software.

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SMALL MAMMAL CAPTURES

Introduction

Small non-volant mammals are generally defined as those that are smaller than 1kg (Lim and Pacheco 2016). They are nocturnal, found on every continent except Antarctica and represent more than half of overall mammalian diversity (Lim and Pacheco 2016). The small mammal species of Missouri are encompassed within the orders Eulipotyphla and Rodentia (Schwartz et al. 2016). These orders each have many species with differing habitat preferences, ecosystem roles, and diets.

Small mammals in the order Eulipotyphla are generally insectivorous, using their elongated snout and enlarged front incisors to capture and to crunch their insect prey, although they are also known to eat berries, nuts, roots, and meat (Schwartz et al. 2016). Species in the order Rodentia have large gnawing incisors to chisel into foods such as seeds, nuts, wood, roots and berries. Small mammals are prey for many other species, and their availability as prey can impact the populations of the predators causing predator and prey populations to cyclically rise and fall (Therrien et al. 2014).

The intent of this study is to document as many small mammal species as possible at two state parks in Shannon County, Missouri, Echo Bluff State Park (EBSP) and Current River State Park (CRSP), prior to the impacts of development and to draw comparisons between the different sites. Both parks encompass portions of the Current River, stark vertical relief and a variety of Ozark habitat types. CRSP is a former corporate retreat that is only open for day use, while EBSP has undergone a large amount of development in the recent past to become a modernized recreational area.

A total of 8 sites across both parks were trapped multiple times and were compared/contrasted against each other. Due to the greater diversity of habitats and minimal human disturbance at EBSP, greater species richness is predicted at the sampling sites located in this park. CRSP has a greater number of old buildings and potential anthropogenic food sources, therefore we expect to see a greater number of overall mammal captures at this park, but that the captures will be from more common or invasive species. Additionally, other areas were surveyed to further document known mammal species in the area, which will be discussed in context with the two study areas.

Methods

Trapping. A variety of trap methods were used to capture the small mammals, all of which were different types of live traps. Sherman traps (H.B. Sherman Traps Inc., Tallahassee, FL) were also utilized for capturing small mammals. Sherman traps are small box-style live traps that utilize an internal treadle that will be tripped, once an animal enters enough to be clear of the front door, closing the spring loaded door. Three sizes of Sherman traps were used, one large enough for rats (10.16 x 11.43 x 38.1 cm), one sized for mice (7.62 x 8.89 x 22.86 cm), and one small enough for shrews (5.08 x 6.35 x 16.51 cm). Mixtures of sizes were used at each trapping location. Trap areas were often on ecotones to increase the diversity of captures. Sites were sampled 12 times between August 13, 2014 and November 14, 2015 (Table 12). At each trapping location 70 traps were placed, and four locations being trapped each time, yielding 280 traps set per night for two sequential nights. A total of 6,650 trap nights were conducted across both parks, with 2,840 trap nights at CRSP and 3,810 trap nights at EBSP. The difference in effort between the two parks was due to focusing sampling efforts at EBSP early on in the study, prior

to the start of construction at the site. Construction did hamper efforts to consistently resample certain areas at Echo Bluff leading to uneven sampling effort at some locations due to lack of access or habitat alterations (Table 13).

Sherman traps were baited with a combination of peanut butter, oats, and bird seed. Traps were placed in areas that would provide good cover to rodents, and prior to placement the trapper would make a small path to encourage rodents to run along these paths into the trap, an example of which can be found in Figure 17. Sherman traps were baited and set in the afternoon and checked first thing in the morning to prevent animal overheating during the day. All captured individuals were immediately removed from the traps, were photographed, and had anatomical data (mass, sex, length of right hind foot, head and body, ear, and total length) collected for identification purposes.

Pitfall traps were used to capture mammals that are too small to trip the treadle or to wary to be captured in Sherman traps, such as shrews. Pitfall traps were dug in two sites at CRSP and three sites at EBSP. Pitfalls were placed in areas deemed good habitat such as the edge of glades or in rocky areas. A hole approximately 0.3m deep was dug, and an empty four-liter container was placed in the hole so that the ground surface was flush with the rim of the container, creating a hole in the ground into which a small mammal could fall. Holes were placed in the bottom of the container to allow any water in the container to drain out. Each container also had a lid for closing the traps when they were not in use. If the pitfall was in an open area, the buried containers were placed in groups of five with one container in the middle and four drift fences radiating out from the center container to the other four, acting as a barrier to small mammals to guide them to the containers. If the pitfall was placed in an area with natural structures, such as logs or rocky areas, then the containers were located near these structures and small pieces of

drift fence were used to provide a possible pathway to the containers. Examples of each pitfall set-up can be seen in Figures 18 & 19. The location of the capture was recorded, and all individuals were then released in the area of capture. This project was approved by the MSU IACUC (14-036), all captures were reported to Missouri Dept. of Conservation, and I followed guidelines from the American Society of Mammalogists (Sikes and Gannon 2011).

Similarity Indices. Similarity indices and observed shared species tables were calculated with a subset of the data for sites that were trapped repeatedly to compare the species makeup of the sites. Estimate S software was used to evaluate the similarity of species captured between each trapping location. The total counts of each species at each trap location were coalesced into a matrix, and a coverage-based estimator of 10 was used for the upper limit of rare and infrequent species via the suggestion in the Estimate S Users Guide (Colwell 2013). I have chosen to use the Chao-Jaccard-Raw Abundance-based Index since it takes abundance into account with no loss of generality (Chao et al. 2005). The Chao-Jaccard Raw Abundance-based index values range from 0-1 with 1 being most similar (Chao et al. 2005). The index score is calculated as: $J_{abc}= \frac{UV}{U+V-UV}$ where U denotes the total relative abundance of individuals belonging to shared species within the first sample ($U= p_1 + p_2 \ldots + p_x$) and V denotes the total relative abundance of individuals belonging to shared species within the second sample ($V= \pi_1 + \pi_{2 \ldots} \pi_x$) (Chao et al., 2005).

Results

Mammalian Inventory. A total of 164 individuals comprising nine species was captured over the course of the study (eastern woodrat, *Neotoma floridana*; white-footed mouse, *Peromyscus leucopus*; hispid cotton rat, *Sigmodon hispidus*; golden mouse, *Ochrotomys nuttalli*;

eastern chipmunk, *Tamias striatus*; prairie vole, *Microtus ochrogaster*; southern bog lemming, *Synaptomys cooperi*; northern short-tailed shrew, *Blarina brevicauda*; and least shrew, *Cryptotis parus*). All nine species were captured at EBSP and six species were captured at CRSP; the least shrew, southern bog lemming and prairie vole were not been found at CRSP (Table 14). EBSP had greater numbers of all species common to both parks with the two exceptions, the eastern chipmunk and northern short-tailed shrew.

By far, the most commonly captured species was the white-footed mouse, with 93 total captures, more than triple the next most commonly captured species (northern short-tailed shrew) with 25 captures. White-footed mice made up 56.7% of the total captures across both parks (Table 14). Two species were the least captured; the least shrew and the southern bog lemming were only captured twice each, both at EBSP. The least shrew and southern bog lemming made up the smallest capture percentage each species representing 1.22% of the total captures (Table 14).

Similarity Indices. A subset of the total data, the eight sites that were sampled multiple times, was used to create a similarity index. This subset included 48 captures of the nine species mentioned previously. Only three species were found across more than two sites; northern short-tailed shrew, eastern woodrat, and white-footed mice, with northern short-tailed shrew being found at all sites but one. White-footed mice had the largest total captures (15) but only were captured at half of the sites. The EBSP Z Box Pond site had both the largest number of site captures (17) and the largest species richness (7). EBSP Z Box Pond was the site where all of the hispid cotton rats were captured and one of two sites where golden mice and prairie voles were captured. EBSP Zoe Prairie, EBSP Mid Zoe, and CRSP Behind Trailer sites each had four

captured species, while CRSP by Moultrie and CRSP by Cheyenne both had a single capture of northern short-tailed shrew (Table 15).

The observed shared species values show that two site combinations (EBSP Z box Pond x EBSP Mid Zoe, and EBSP Z Box Pond x CRSP Behind Trailer) shared four species, with all other combinations being three or fewer. The combinations of EBSP Zoe Prairie x EBSP Zoe Riparian, EBSP Zoe Riparian x EBSP Mid Zoe, and EBSP Zoe Riparian x CRSP by Cheyenne had no species in common (Table 16).

The calculated Chao-Jaccard Raw Abundance-based similarity estimator values show a single comparison (CRSP by Moultrie x CRSP by Cheyenne) having a value of 1.0, the largest possible value of similarity. Two site comparisons (CRSP Behind Trailer x EBSP Z Box Pond and CRSP Behind Trailer x EBSP N Zoe Glade) have the next largest similarity estimator value of 0.714, while two others (EBSP N Zoe Glade x EBSP Z Box Pond and EBSP Mid Zoe x EBSP Z Box Pond) have a value of 0.643. All other combinations had an estimated value of less than 0.525, with four of them being zero due to a lack of any shared species (Table 17).

Discussion

Mammalian Inventory. A total of nine species were captured via Sherman and pitfall traps over the course of the study (Table 14). There were six native species that are known to occur in Shannon county that were absent from the trap captures effort: southeastern shrew, *Sorex longirostris*; western harvest mouse, *Reithrodontomys megalotis*; fulvous harvest mouse, *Reithrodontomys fulvescens*; deer mouse, *Permyscus maniculatus*; woodland vole, *Microtus pinetorum*; meadow jumping mouse, *Zapus hudsonius*. These likely were absent due to poor representation of their ideal habitat in the sample area or insufficient sampling effort or a

combination. The house mouse (*Mus musculus*) is known to occur in Shannon County and the brown rat (*Rattus norvegicus*) is likely to occur in the county, though they are both invasive species.

White-footed mice prefer wooded areas with grassy borders with population densities varying from one to 20 per acre (Schwartz et al. 2016). The study areas fit this description well, especially the ecotone/edge areas, which is the likely basis for their abundance in the dataset. The least shrew and southern bog lemming had the fewest captures overall. The least shrew prefers open grass, brush, and old fields; which are represented at both parks (Schwartz et al. 2016). The species is abundant across the state and is known to occur in Shannon County (Schwartz et al. 2016). This leads me to believe that due to their small mass, 2-10 g, they were too small to trigger the treadle in the Sherman traps and only were able to be captured in the pitfall traps. The bog lemming prefers areas with heavy grass growing in low moist places along with damp woods with lots of leaf cover (Schwartz et al. 2016). While these areas are represented within both parks, the steep topography of the area does not leave a lot of low lying area. This type of habitat was more represented at EBSP than CRSP and likely led to the captures.

When comparing CRSP and EBSP, there were more than double the number of captures and four additional species at EBSP than CRSP (Table 14). These results support the prediction that EBSP would have greater richness. Even when comparing trap success rates EBSP is much larger than CRSP (2.99 and 1.76 captures per 100 trap nights, respectively). While the data did not support the prediction that CRSP would have a larger total number of captures, this result possibly was due to a difference in almost 1000 trap nights between the parks (Table 12). The difference in effort was due to the desire to trap more heavily at EBSP while the areas were still

intact prior to the initiation of construction and its impacts on the habitat. Another factor that could have influenced both total capture numbers and the difference in species richness between the two parks is the greater habitat diversity at EBSP than CRSP, especially prior to the development of the park (McCarty 2008; McCarty et al. 2013). It is difficult to discern how large of a factor the habitat diversity had due to the difference in trapping effort between the parks, but the data support the prediction that EBSP would have greater species diversity.

Two species, northern short-tailed shrew and the eastern chipmunk, were captured more at CRSP than EBSP. Northern short-tailed shrew had one more capture at CRSP despite both parks having ample habitat for them with many low-lying, wet localities with weedy growth along the Current River and the small creeks that feed into it (McCarty 2008; McCarty et al. 2013; Schwartz et al. 2016). The eastern chipmunk was captured much more frequently at CRSP than EBSP. CRSP had many more old fields and maintained lawns leading to more consistent timber borderland, likely contributing to their greater capture rate even with lower trapping effort in the park (McCarty 2008; Schwartz et al. 2016). Both of these species are common species that are less affected by human disturbance. Chipmunks often inhabit parks, stone walls, and outbuildings, and the northern short-tailed shrew is often one of the most abundant species in a forested area (Schwartz et al. 2016). These two species were captured more often at CRSP supporting the prediction that generalist species and those that are less affected by anthropogenic impacts would be captured in greater numbers there.

Similarity Indices. Only three species were captured across more than two resampled sites, northern short-tailed shrew, eastern woodrat, and white-footed mice (Table 15). Both northern short-tailed shrew and white-footed mouse can reach high densities (up to 50 and 20 per acre, respectively) and were the most common species in all samples and the resampled areas

(Schwartz et al. 2016). The eastern woodrat prefers rocky wooded areas and most often was captured at sites with higher elevation and therefore the traps set in drier, rocky, forested areas led to their greater capture rates. Eastern woodrats are one of the heavier species in the study, which allowed them to be more easily captured by Sherman traps.

Two comparisons between three different sites had four species in common (Table 16). These three sites had the most captures and were among the most diverse sites. The four species in common between CRSP Behind Trailer and EBSP Z Box Pond were the three most commonly captured species mentioned previously and the eastern chipmunk. While many sites were forest-edge ecotones, the CRSP site had many old, little used outbuildings nearby and stonewalls, both of which are preferred habitat for the species (Schwartz et al. 2016).

The sites with the largest abundance and the most diversity (tied for second most in the case of EBSP Mid Zoe) shared four species between them. These two sites shared some of the rarer species across resampled sites, namely the golden mouse and the prairie vole. The golden mouse is a habitat specialist and highly localized, preferring moist lowland habitats and forest borders. Both of these areas had patchy wooded areas near water sources and lots of vines and shrubs that golden mice use to climb (Schwartz et al. 2016). Prairie voles, on the other hand, prefer fields and grasslands with herbaceous cover in which to make runways. Due to the high amount of forest/field edge and the proximity of water sources at these sites both species were able to be captured. The ecotonal nature of the sites likely contributed to the abundance of captures, diversity, and high site similarity result (Sekgororoane and Dilworth 1995).

The calculated similarity matrix shows one site combination of 1.0 (Table 17), but also had both of the sites with the lowest capture number (Table 15). CRSP Moultrie and CRSP Cheyenne were sampled six and eight times respectively (Table 13), and only had one capture of

a northern short-tailed shrew at each site. After high effort to sample these areas and only having found one example of the same species, it is clear how the Chao-Jaccard-Raw Abundance-based similarity estimate would calculate a perfect similarity (Chao et al. 2005). It is unlikely they are of perfect similarity biologically, but further sampling is needed to determine the true extent of their similarity.

EBSP Z Box Pond had the most captures and the largest species richness (Table 15); leading to high calculated similarity values with other less diverse sites (Table 18). CRSP Behind Trailer also had an estimated similarity score of 0.714 with EBSP Z Box Pond and shared four species, the three most common and the eastern chipmunk. EBSP Z Box Pond is a rocky/brushy upland area next to an open field. CRSP Behind Trailer is at a lower elevation but has a wooded area abutting an open area, in the form of a maintained lawn. This similarity likely led to the species they share. The major difference between the sites is the presence of permanent water in the form of a pond at EBSP Z Box Pond. The habitat diversity and the pond likely contributed to EBSP Z Box Pond having the most captures and largest species richness. EBSP Z Box Pond was one of two sites that captured some of the rarer species, namely the golden mouse and the prairie vole. These two species were also captured at EBSP Mid Zoe, hence the similarity value of 0.643. While EBSP Z Box Pond is at a much higher elevation, these sites are both very moist and lush due to permanent water (pond at EBSP Z Box Pond and Current River at EBSP Mid Zoe). Both sites have grassy areas that appeal to the prairie vole, and abutting wooded areas with vine filled undergrowth ideal for the golden mouse (Schwartz et al. 2016). These two sites had the largest number of captures and some of the largest species richness. This relationship shows that the presence of permanent water and an edge type habitat are likely factors that create species richness and abundance.

CRSP Behind Trailer and EBSP N Zoe Glade only share three species but also have an estimated similarity value of 0.714. These sites had similar capture numbers (Table15), with the only difference being an additional white-footed mouse capture at CRSP Behind Trailer. These three species were the most commonly captured species in the resampled dataset (Table 15). These two trap location habitats were similar in that they were both ecotones of an open to wooded area with CRSP Behind Trailer being a transition from maintained grass to heavily wooded and N Zoe Glade transitioning from a glade to heavily wooded. Both of these areas had large dead logs, rocky areas, and understory brush, which are ideal for eastern woodrats (Schwartz et al. 2016). This habitat is also ideal for the eastern chipmunk, which was the only additional species found at CRSP Behind Trailer that was absent from EBSP Zoe Glade. I think it is possible an eastern chipmunk could have been captured at EBSP Zoe Glade with continued sampling.

While there were similarities between habitats of the same type at both parks, the dataset shows that EBSP had both the larger species richness and larger capture number due to the larger habitat heterogeneity across the park or the difference in the level in sampling effort between the two parks. The larger captures of chipmunks and northern short-tailed shrews make sense because they are generalist species that are not averse to areas with human disturbance such as that found around the various buildings and maintained lawns in CRSP.

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SUMMARY

Over the course of this study I attempted to quantify species richness, habitat types, and spatial use of mammals within two state parks inside the ONSR through the use of camera traps, acoustic detectors, mist netting, Sherman traps, and pitfall traps to identify and count species. These data were analyzed using accumulation curves, shared species estimates, and occupancy analysis to fully describe the specific species parameters for the mammalian communities in the study area. Additionally, I have attempted to describe how the habitat characteristics in each park may have influenced the mammalian community structure observed during this study. Finally, I attempted to add to the species inventory for the state parks and the ONSR.

Almost all large mammal species that were expected in the area were found on the camera traps with the exception of some rare and more difficult to document species. Accumulation curves indicated that further sampling would be unlikely to yield additional species. Similarity indices for the sites show high estimated similarity between many sites across both parks. Occupancy models helped to elucidate landscape and anthropogenic factors that affected my ability to detect those species and their occupancy. The known gray bat hibernaculum near CRSP greatly influenced the abundance of bat activity and which species were most active between the sites. All bat species known from the county, including all threatened and endangered species, were accounted for over the course of the study. Habitat use and general bat activity was seasonally dependent with more activity in warmer months. EBSP had a greater number of small mammal captures and greater overall species richness. Greater numbers of generalist species were captured at CRSP likely due to old infrastructure and their increased tolerance of human presence.

As the heterogeneity of the landscape increases the more habitat features there are for species to choose from. While EBSP had greater overall habitat diversity it had varying degrees of mammalian richness compared to CRSP. EBSP had greater richness of small mammals compared to CRSP, but lower richness of large mammals. This could be due to making choices at different habitat scales between the two groups, with smaller mammals deciding on a much more local scale than larger mammals. With this in mind, the smaller total size of EBSP may have led to the decreased richness of large mammals, as the scale of the park itself was not large enough to provide sufficient habitat for the larger species. Anthropogenic factors also could be influencing this discrepancy in richness. The habitat disturbance at EBSP during the course of the study likely impacted species differently dependent upon scale. In general habitat disturbance and fragmentation have been shown to cause negative effects to population size (Andren 1994) although some small mammals have been known to benefit from disturbance, resulting in greater abundance and larger individual masses (Kaminski et al. 2007). The difference in scale of the impact of disturbance may account for the difference in diversity between the two parks overall.

Habitat heterogeneity and differing species habitat preferences are the driving factors for the movement and survival patterns dictating population trends and eventually community scale influences. The scale and types of habitat features that species were shown to choose differ across the various mammalian groups being investigated, which was then shown to influence the among species composition and counts at each location. The various levels of mobility between species, ranging from local movements to migrations, affected the level of landscape features they chose. Bats and larger mammals have the ability to travel greater distances than small mammals and therefore make wider habitat choices which was seen in the seasonally decreased activity in the study habitats of bats, the high similarity of the study sites for large mammals, and

even the absence of typically common large mammals from the dataset completely due to their extremely large home ranges. With reduced mobility, small mammals must make more localized preferences in habitat, which was seen across different trap sites within the same park. These findings support the idea that movement scales are strongly linked to habitat preferences which in turn both impact species richness and population sizes, increasing total community richness (Burt 1943; Shaw and Couzin 2013; Barten et al. 2014).

This study has provided information on trends in mammalian species, biodiversity, and habitat needs in the ONSR. While this is not the first study of its type in the area, it provides essential data on the current state of mammalian biodiversity, while examining two parks with different histories and modern management practices. With the descriptions of species richness and landscape hotspots, this study can be used to help guide future land management and wildlife conservation practices in the area. This dataset shows the impact of construction and human disturbance in natural areas, both in the short term (recent construction at EBSP), and the long term (many old buildings and human made land features in CRSP). This information is essential for understanding the impact of anthropogenic events on wildlife across a variety of time scales. Additionally, these data help with the understanding of how species and populations fluctuate naturally over time, which is necessary knowledge for their proper management. This study is a starting point for the future monitoring of CRSP and EBSP, the important habitats in them, and the many mammalian species which call the parks their home.

Limitations and Recommendations

Almost all limitations arose from two sources: the desires of the study parameters requested by Missouri DNR and the construction occurring at EBSP.As with most studies,

concessions were made due to limited funding and time. In this case, funding limited the number of possible visits to the study site and the number of study sites that could be sampled. Increased levels of effort across all portions of the study could have yielded additional species and more robust analyses, but this was not possible due to the limited time frame of this project. More frequent visits would have cut down on data loss on camera traps and acoustic detectors.

When doing a long term study, consistency in data collection is essential. On multiple occasions during the study, encroaching construction necessitated moving passive sampling equipment that had long been present in a location. This inconsistency potentially impacted the data collection and the resulting analyses. Another issue was baiting the camera traps, which came from the tradeoff of wanting to attract as many species as possible into each camera's view to encapsulate the species richness of the area, while also knowing that the sardine bait could potentially entice species that do not normally occupy that space and/or repel others that normally do occupy that space. Repelling species that normally occupy the area and enticing species to areas that they would not occupy naturally could lead to false presence in an area and greatly affect the occupancy model outcomes and the relative abundance of certain species, creating an unrepresentative and biased sample.

Recommendations for Future Studies

There are still many questions that could be asked about mammalian species in the EBSP and CRSP as well as the ONSR in general such as the influence of introduced mammal species, or the impact of human disturbance on mammalian abundance. Many of these inquiries have been investigated in areas across Missouri and the United States but have yet to be explored in this area. It would be interesting to see the impact of large introduced mammal species on the

native mammals of the area, such as the feral horse population that roams between parks. The detriments of introduced large mammal species have been studied before and would likely apply in this region (Houston and Schreiner 1995). I also think that it would be interesting to see the impacts of the development of EBSP and other anthropogenic influences on the mammalian species diversity and abundance in the area over time. I would be curious as to how the rapid pace and disturbance of differing habitats would change the species make up.

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Table 1. Level of effort for each camera trap, including dates deployed and removed, and total number of days sampled across both parks.

Camera name	Park	Date deployed	Date removed	Days nonfunctioning	Number of camera days
Moultrie	EBSP	4/18/2014	~1/23/15 *	0	280
By Creek	EBSP	5/4/2014	11/15/2015	0	563
Zoe Prairie	EBSP	12/1/2014	3/6/2015#	0	95^
Mid Zoe	EBSP	10/25/2014	11/15/2015	0	387
Behind Barn	EBSP	8/15/2014	9/1/2014*	0	19^
N Zoe Glade	EBSP	10/25/2014	11/15/2015	0	387
Z Box Pond	EBSP	10/25/2014	11/15/2015	0	386
EBSP total				0	2117
Cheyenne	CRSP	5/2/2014	11/15/2015	0	563
Dump Road	CRSP	5/27/2014	8/14/2014#	0	79^
N of Pond	CRSP	5/2/2015	11/15/2015	0	563
S Road	CRSP	5/2/2014	11/15/2015	0	563
Moultrie	CRSP	5/2/2014	11/15/2015	24	539
CRSP total				24	2307
Study total				24	4424

* Error or loss of camera # Moved due to low observances/construction

^ Considered insufficient for analysis

Table 2. Total observations across all camera sites over the course of the sampling period used to create similarity indices in Estimate S software.

	By Creek	Cheyenne	Mid Zoe	Moultrie CRSP	Moultrie EBSP	N of Pond	N Zoe Glade	S Road	Z Box Pond	Total per species
Coyote	3	54	15	23	14	13	51	60	2	235
Armadillo	0	17	0	0	0	1	0	0	3	21
Opossum	0	1	0	0	0	0	0	2	3	6
Bobcat	0	1	0	0	1	0	3	0	3	8
Striped skunk	0	0	2	0	0	0	0	1	0	3
Deer	38	67	91	31	24	20	9	16	78	374
Raccoon	15	44	22	0	8	0	4	31	23	147
G Squirrel	0	59	3	5	9	0	99	26	3	204
F squirrel	0	9	0	0	0	0	0	0	0	9
E cottontail	0	2	0	0	0	0	3	1	6	12
Chipmunk	0	4	0	0	0	0	0	0	0	4
Gray fox	0	0	0	2	0	0	1	10	0	13
Total per site	56	258	133	61	56	34	170	147	121	
Species per site	3	10	5	4	5	3	7	8	8	

Table 3. All site-specific information for each camera used in occupancy analysis.

Camera Name	Park	Habitat type	Dist. to road(m)	Dist. to river(m)	Dist. to water (m)	Camera brand	Elevation (m above sea level)
By Creek	EBSP	Dry Mesic Bottomland	308	523	2	Reconyx	820
Cheyenne	CRSP	Dry Mesic Bottomland	64	130	130	Reconyx	795
Mid Zoe	EBSP	Dry Mesic Bottomland	172	40	40	Reconyx	732
Moultrie- EBSP	EBSP	Dry Mesic Bottomland	193	188	28	Moultrie	772
Moultrie-CRSP	CRSP	Dry Mesic Forest	216	885	70	Moultrie	855
N of Pond	CRSP	Dry Chert	400	1036	160	Reconyx	811
N Zoe Glade	EBSP	Dry Chert	66	251	251	Reconyx	946
S Road	CRSP	Dry Mesic Bottomland	529	1061	15	Reconyx	865
Z Box Pond	EBSP	Dry Chert	75	205	205	Reconyx	853
Zoe Prairie	EBSP	Dry Mesic Forest	63	178	178	Reconyx	748

Table 4. Detection and occupancy models used for analysis, with open parenthesis for the detection covariate in the occupancy models representing the best detection models per species which were determined before attempting to estimate occupancy.

Detection models	Occupancy models
p(.) Ψ(.)	p(__) Ψ(Dist. To Road)
p(Habitat Class) Ψ(.)	p(__) Ψ(Dist. To River)
p(Park) Ψ(.)	p(__) Ψ(Dist. To Water)
p(Mean Temp.) Ψ(.)	p(__) Ψ(Habitat Class)
p(Elevation) Ψ(.)	p(__) Ψ(Elevation)
p(Precip.) Ψ(.)	p(__) Ψ(Park)
p(Ordinal) Ψ(.)	
p(Dist. To Water) Ψ(.)	
p(Dist. To River) Ψ(.)	
p(Dist. To Road) Ψ(.)	
p(Precip.+Ordinal) Ψ(.)	
p(Mean Temp.+Ordinal) Ψ(.)	

Table 5. Top scoring occupancy and detection models for each species.

Species	Optim method	Top models	Δ AIC	Model weight
Armadillo	SANN	p(Habitat Class) Ψ(Dist. To Water)	0.00	0.87
Bobcat	L-BFGS-B	p(Dist. To River) Ψ(Dist. To River)	0.00	0.37
		p(Dist. To River) Ψ(Dist. To Road)	0.87	0.32
E. cottontail	Nelder-Mead	p(Habitat Class) Ψ(Habitat Class)	0.00	0.13
	SANN	p(Park) Ψ(Habitat Class)	0.25	0.12
	Nelder-Mead	p(Dist. To Water) Ψ(Habitat Class)	0.85	0.09
	SANN	p(Habitat Class) Ψ(.)	0.90	0.08
		p(Dist. To Water) Ψ(.)	1.14	0.08
	L-BFGS-B	p(Habitat Class) Ψ(Dist. To River)	1.77	0.05
	SANN	p(Dist. To Water) Ψ(Dist. To Road)	1.95	0.05
Gray fox	SANN	p(Mean Temp.) Ψ(Elevation)	0.00	0.90
Gray squirrel	Nelder-Mead	p(Ordinal+Mean Temp.) Ψ(.)	0.00	0.67
		p(Dist. To Water) Ψ(.)	1.44	0.33
Opossum	L-BFGS-B	p(.) Ψ(Elevation)	0.00	0.16
		p(.) Ψ(.)	0.91	0.10
		p(Ordinal) Ψ(.)	1.53	0.07
Raccoon	L-BFGS-B	p(Park) Ψ(Dist. To River)	0.00	0.20
		p(Park) Ψ(.)	0.26	0.18
		p(Park) Ψ(Park)	0.64	0.15
		p(Park) Ψ(Habitat Class)	0.65	0.15
		p(Park) Ψ(Dist. To Water)	1.85	0.08
Coyote	GLM	p(Park+Habitat Class) Ψ(.)	0.00	-
Deer	GLM	p(Park+Habitat Class) Ψ(.)	0.00	-

Table 6. The number of species observed at each camera compared to the number of estimated species calculated using the equation for Chao 2.

Park	Camera	Observed	Chao 2
CRSP	Cheyenne	10	12
CRSP	Moultrie-CRSP	4	4
CRSP	North of Pond	3	3
CRSP	South Road	8	10
EBSP	By Creek	3	3
EBSP	Mid Zoe	5	5
EBSP	Moultrie-EBSP	5	5
EBSP	North Zoe Glade	7	7
EBSP	Z Box Pond	8	8

Table 7. A comparison of the number of observed species shared among each camera location.

Cheyenne	Mid Zoe	Moultrie CRSP	Moultrie EBSP	N of Pond	N Zoe Glade	S Road	Z Box Pond	
3	3	2	3	2	3	3	3	By Creek
	4	3	5	3	6	6	8	Cheyenne
		3	4	2	4	5	4	Mid Zoe
			3	2	4	4	3	Moultrie CRSP
				2	5	4	5	Moultrie EBSP
					2	2	3	N of Pond
						6	6	N Zoe Glade
							6	S Road

Table 8. Calculated values of the Chao-Jaccard-Raw Abundance-based estimator for each comparison between camera locations.

Cheyenne	Mid Zoe	Moultrie CRSP	Moultrie EBSP	N of Pond	N Zoe Glade	S Road	Z Box Pond	
0.64	0.962	0.669	0.821	0.716	0.376	0.728	0.851	By Creek
	0.857	0.682	0.872	0.535	0.875	0.821	0.95	Cheyenne
		0.797	0.968	0.778	0.945	0.912	0.864	Mid Zoe
			0.816	0.862	0.941	0.762	0.67	Moultrie CRSP
				0.665	0.976	0.89	0.901	Moultrie EBSP
					0.349	0.509	0.686	N of Pond
						0.963	0.945	N Zoe Glade
							0.883	S Road

Table 9. Effort for mist netting and corresponding species captured. EPFU=*Eptesicus fuscus*, GLVO= *Glaucomys volans*, LABO=*Lasiurus borealis*, LACI=*Lasiurus cinereus*, LANO=*Lasionycteris noctivagans,* MYGR=*Myotis grisescens,* MYSO=*M. sodalis*, MYSE= *M. septentrionalis,* NYHU=*Nycticeius humeralis*, PESU= *Perimyotis subflavus*

Date	Park	Net nights	Net sizes	Species captured
5/24/2014*	EBSP	3	2x12, 2x9, 2x6	PESU, GLVO, MYGR, LABO, EPFU
5/25/2014*	EBSP	4	3x12, 3x9, 2x4, 2x6	LABO, MYSE, MYSO, GLVO, EPFU
5/26/2014*	EBSP	5	1x4, 1x4, 2x9, 2x4, 3x6	MYGR, LABO, EPFU, NYHU, GLVO, MYSE
5/27/2014*	EBSP	3	2x12, 2x9, 2x6	EPFU, NYHU, MYSE, MYGR, PESU
5/28/2014*	EBSP	4	3x12, 3x9, 2x4, 2x6	MYGR, PESU
5/29/2014*	EBSP	5	1x4, 1x4, 2x9, 2x4, 3x6	LABO, MYGR, MYSO, MYSE
5/30/2014	CRSP	2	2x4, 2x9	MYGR
7/10/2014	CRSP	3	2x6, 2x6, 3x9	NYHU
7/11/2014	CRSP	3	3x12, 2x9, 2x9	MYSE
8/13/2014	CRSP	5	2x12, 2x9, 2x9, 2x9, 3x12	LABO, EPFU, PESU
8/14/2014	EBSP	3	2x9, 2x4, 2x12	LABO, PESU, MYSE
8/15/2014	CRSP	3	2x9, 2x6, 2x12	LACI, LABO, EPFU
9/19/2014	CRSP	2	2x6, 2x9	EPFU
9/20/2014	CRSP	2	2x9, 2x12	NYHU, LABO
10/24/2014	EBSP	2	2x4, 3x12	LABO
10/25/2014	CRSP	3	2x12, 2x6, 2x9	LABO, LANO
11/29/2014	CRSP	3	2x9, 2x9, 2x9	LABO
11/30/2014	EBSP	3	2x9, 2x6, 1x4	LABO
1/24/2015	CRSP	2	2x9, 2x6	LABO
3/20/2015	EBSP	2	2x4, 2x6	
3/21/2015	CRSP	2	2x6, 2x6	LABO
5/1/2015	EBSP	2	2x4, 2x4	LABO
5/2/2015	CRSP	2	2x6, 2x6	
6/24/2015	CRSP	2	2x9, 3x12	
6/25/2015	CRSP	2	2x9, 3x12	
8/31/2015	CRSP	2	2x9, 2x12	LABO
9/1/2015	CRSP	2	2x9, 2x12	
10/16/2015	CRSP	2	2x4, 2x12	MYGR, LANO
10/17/2015	EBSP	2	2x6, 2x12	
11/14/2015	CRSP	2	3x9, 2x12	

* Pre-construction survey

Table 10. Level of effort utilized for bat acoustic detectors.

Detector name	Park	Date deployed	Date removed	Days nonfunctioning	Number of det. days
Pond	CRSP	3/9/2014	11/15/2015	455	161
Creek	CRSP	3/9/2014	11/15/2015	289	327
Pond #	EBSP	3/30/2014	11/15/2015	290	305
Creek	EBSP	3/30/2014	11/15/2015	279	316

#-Removed 2/14/15 and replaced 5/1/15.

Table 11. Summary of calls identified to species per location. EPFU=*Eptesicus fuscus*, LABO=*Lasiurus borealis*, LACI=*Lasiurus cinereus*, LANO=*Lasionycteris noctivagans,* MYGR=*Myotis grisescens,* MYLE= *M. leibii*, MYLU- *M. lucifugus,* MYSE= *M. septentrionalis,* MYSO=*M. sodalis,* NYHU=*Nycticeius humeralis*, PESU= *Perimyotis subflavus,* NOID= Not able to be identified

	CRSP Creek	CRSP Pond	EBSP Creek	EBSP Pond	Totals
EPFU	1336	700	1231	196	3463
LABO	9396	17440	4849	2349	34034
LACI	738	767	492	361	2358
LANO	395	2392	1893	430	5110
MYGR	12008	15639	2204	3049	32900
MYLE	30	64	16	7	117
MYLU	592	718	390	77	1777
MYSE	32	93	20	319	464
MYSO	166	153	36	18	373
NYHU	759	923	547	363	2592
PESU	3826	6012	11802	394	22034
NOID	207	139	184	59	589
Noise	35149	145886	123196	18896	323127
Site totals	64634	190926	146860	26518	428938
Identified calls per site	29278	44901	23480	7563	105222

Table 12. Level of effort of small mammal trapping.

Date	Sequential nights	Traps	Total trap nights	CRSP nights	EBSP nights
8/13/2014	2	220	440	200	240
9/20/2014	2	480	960	480	480
10/25/2014	2	175	350	200	150
11/30/2014	2	280	560	280	280
12/19/2014	2	280	560	0	560
1/26/2015	2	280	560	280	280
2/14/2015	2	280	560	0	560
3/21/2015	2	280	560	280	280
5/1/2015	2	280	560	280	280
8/31/2015	2	210	420	280	140
10/16/2015	2	280	560	280	280
11/14/2015	2	280	560	280	280
Total			6650	2840	3810

Table 13. Sherman trap sites that were trapped multiple times and the date of the first day of the two day surveys.

EBSP Zoe Prairie	EBSP Z Box Pond	EBSP Zoe Riparian	EBSP N Zoe Glade	EBSP Mid Zoe	CRSP by Cheyenne	CRSP by Moultrie	CRSP Behind Trailer
12/1/2014	2/14/2015	5/27/2014	8/13/2014	12/19/2014	8/13/2014	7/10/2014	9/20/2014
12/19/2014	3/21/2015	7/10/2015	9/20/2015	1/24/2015	9/20/2014	8/13/2014	10/25/2014
2/15/2015	5/2/2015	11/30/2015	10/26/2015	2/14/2015	8/31/2015	10/25/2014	11/30/2014
	8/31/2015		11/30/2015	10/18/2015	10/18/2015	12/1/2014	11/15/2015
	10/17/2015		12/1/2015			11/30/2015	
						1/26/2015	

Table 14. Summary of small mammal captures by park and each species percent makeup of the total captures.

Species	Total captures	Captures at EBSP	Captures at CRSP	% of total captures
Northern Short-tailed Shrew	25	12	13	15.25
Least Shrew	2	2	0	1.22
Prairie Vole	3	3	0	1.83
E. Woodrat	12	10	2	7.32
Golden Mouse	13	7	6	7.92
White-footed Mouse	93	72	21	56.70
S. Bog Lemming	2	2	0	1.22
Hispid Cotton Rat	4	4	0	2.44
E. Chipmunk	10	2	8	6.10
Totals	164	114	50	

Table 15. Observations from sites sampled multiple times used for similarity index.

	N. Short-tailed Shrew	Least Shrew	Prairie Vole	E. Woodrat	Golden Mouse	White-footed Mouse	S. Bog Lemming	Hispid Cotton Rat	E. Chipmunk	Site totals	Total species
EBSP Zoe Prairie	2	1	0	0	0	1	1	0	0	5	4
EBSP Z Box Pond	2	0	1	3	2	4	0	4	1	17	7
EBSP Zoe Riparian	0	0	0	2	0	0	0	0	0	2	1
EBSP N Zoe Glade	1	0	0	1	0	2	0	0	0	4	3
EBSP Mid Zoe	2	0	1	0	3	5	0	0	0	11	4
CRSP by Cheyenne	1	0	0	0	0	0	0	0	0	1	1
CRSP by Moultrie	1	0	0	0	0	0	0	0	0	1	1
CRSP Behind Trailer	1	0	0	1	0	3	0	0	2	7	4
Total per species	10	1	2	7	5	15	1	4	3		
Number of sites	8	1	2	4	2	4	1	1	2		

Table 16. Observed shared species between resampled small mammal trapping sites created in Estimate S.

EBSP Z Box Pond	EBSP Zoe Riparian	EBSP N Zoe Glade	EBSP Mid Zoe	CRSP by Cheyenne	CRSP by Moultrie	CRSP Behind Trailer	
2	0	2	2	1	1	2	EBSP Zoe Prairie
	1	3	4	1	1	4	EBSP Z Box Pond
		1	0	0	0	1	EBSP Zoe Riparian
			2	1	1	3	EBSP N Zoe Glade
				1	1	2	EBSP Mid Zoe
					1	1	CRSP by Cheyenne
						1	CRSP by Moultrie

Table 17. Estimate S output for the Chao-Jaccard-Raw Abundance-based similarity estimator between small mammal trapping sites with 1 being most similar and 0 being least similar.

EBSP Z Box Pond	EBSP Zoe Riparian	EBSP N Zoe Glade	EBSP Mid Zoe	CRSP by Cheyenne	CRSP by Moultrie	CRSP Behind Trailer	
0.333	0	0.5	0.447	0.4	0.4	0.414	EBSP Zoe Prairie
	0.214	0.643	0.643	0.143	0.143	0.714	EBSP Z Box Pond
		0.25	0	0	0	0.143	EBSP Zoe Riparian
			0.525	0.25	0.25	0.714	EBSP N Zoe Glade
				0.182	0.182	0.431	EBSP Mid Zoe
					1	0.143	CRSP by Cheyenne
						0.143	CRSP by Moultrie

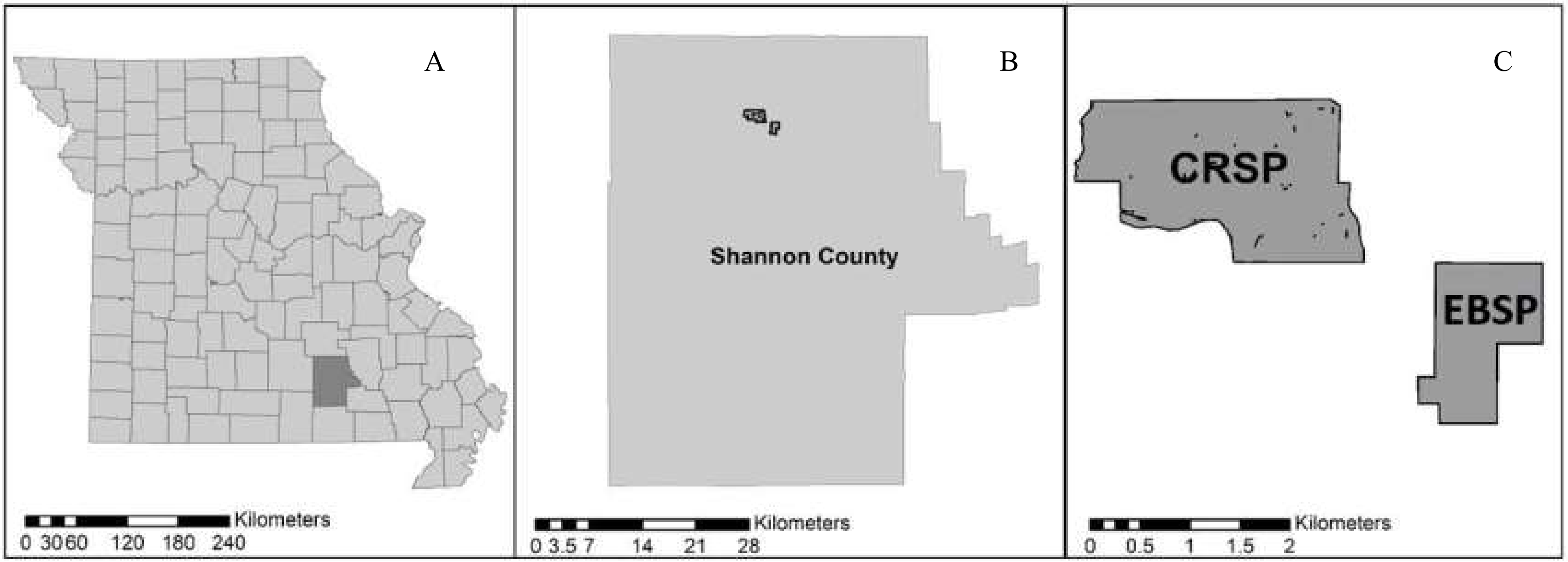

Figure 1. Location of Current River State Park and Echo Bluff State Park within Missouri (A), Shannon County (B), and in relation to one another (C).

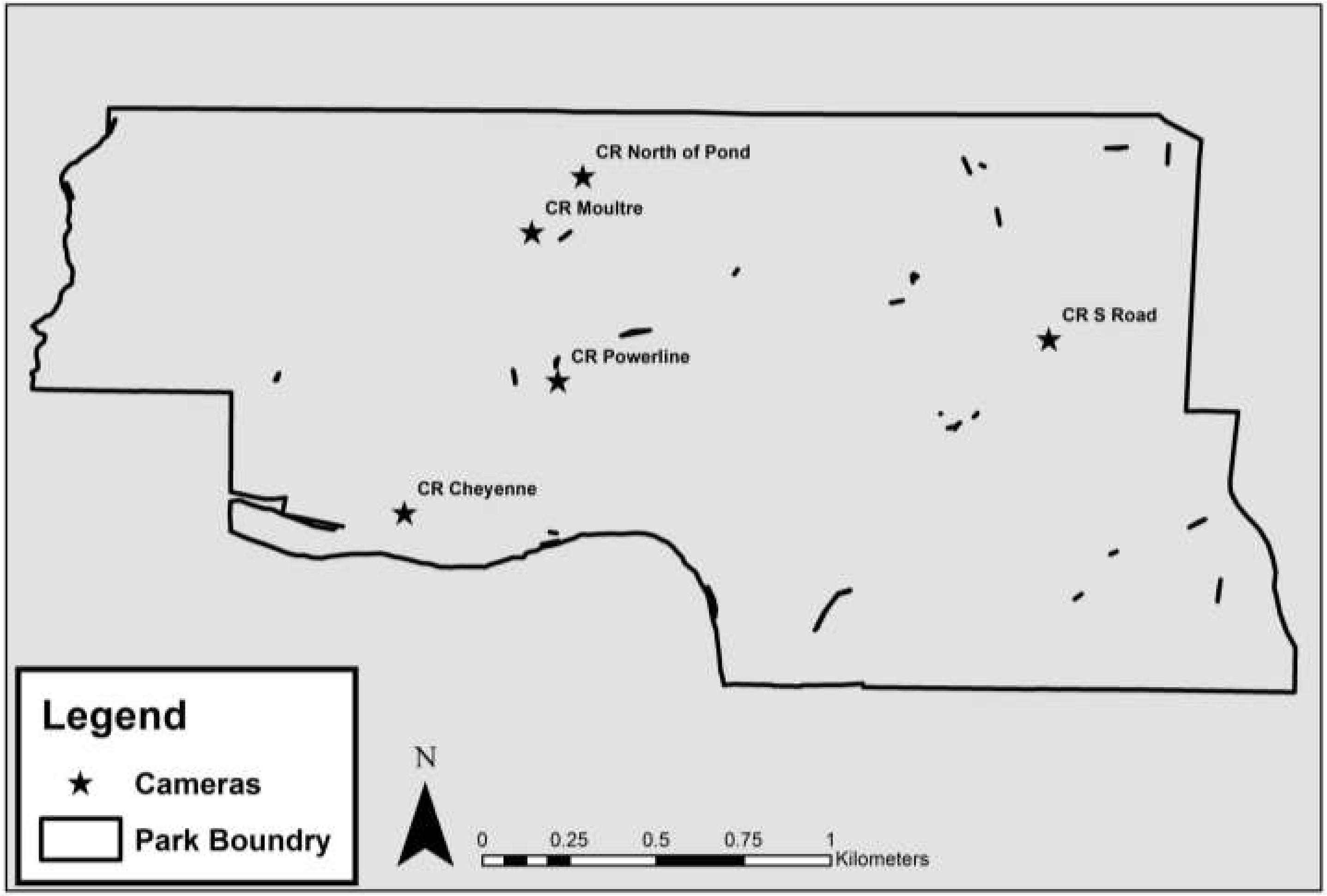

Figure 2. Locations of camera traps and their identifying names within Current River State Park.

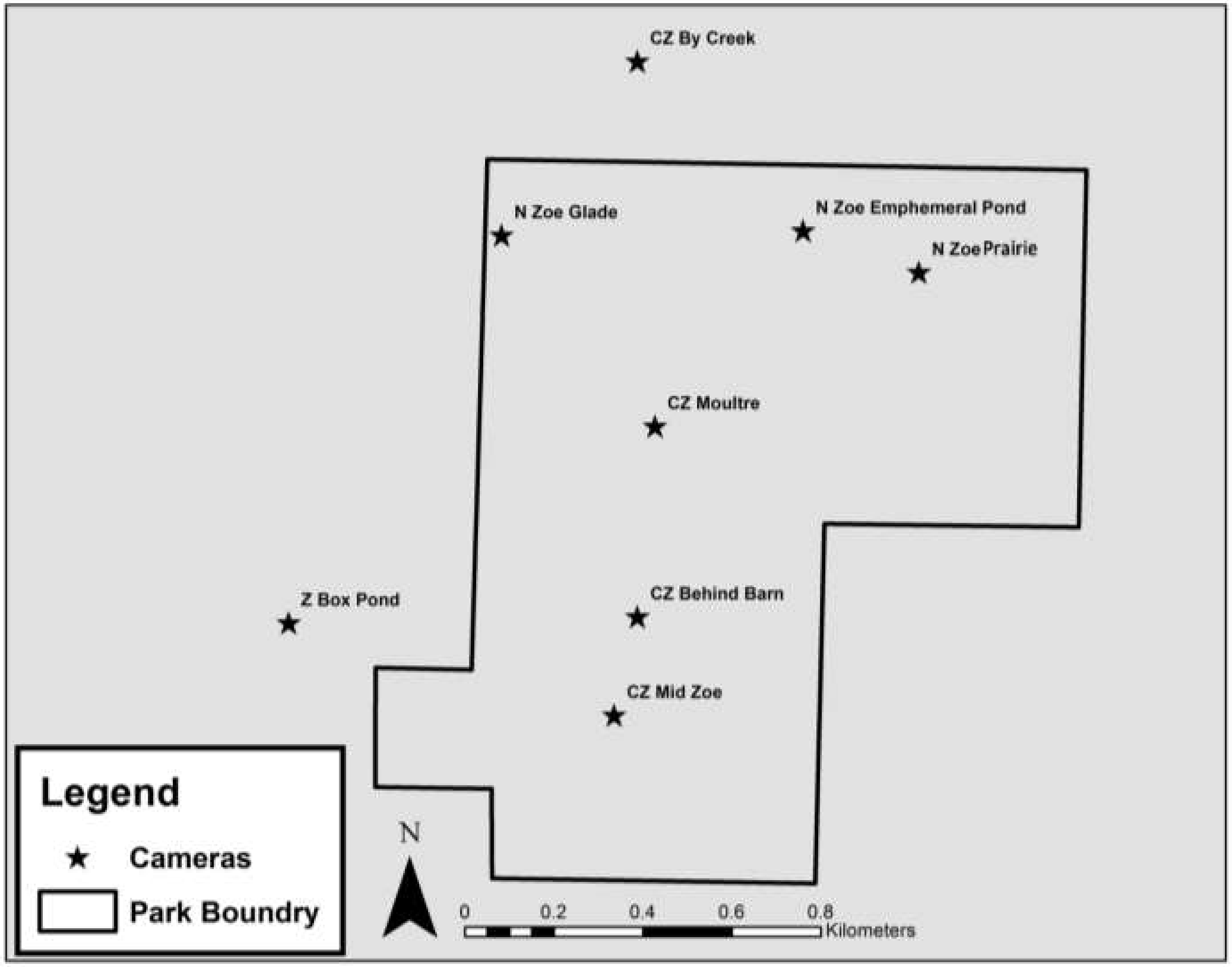

Figure 3. Locations of camera traps and their identifying names within Echo Bluff State Park.

Figure 4. Image of a camera trap set-up.

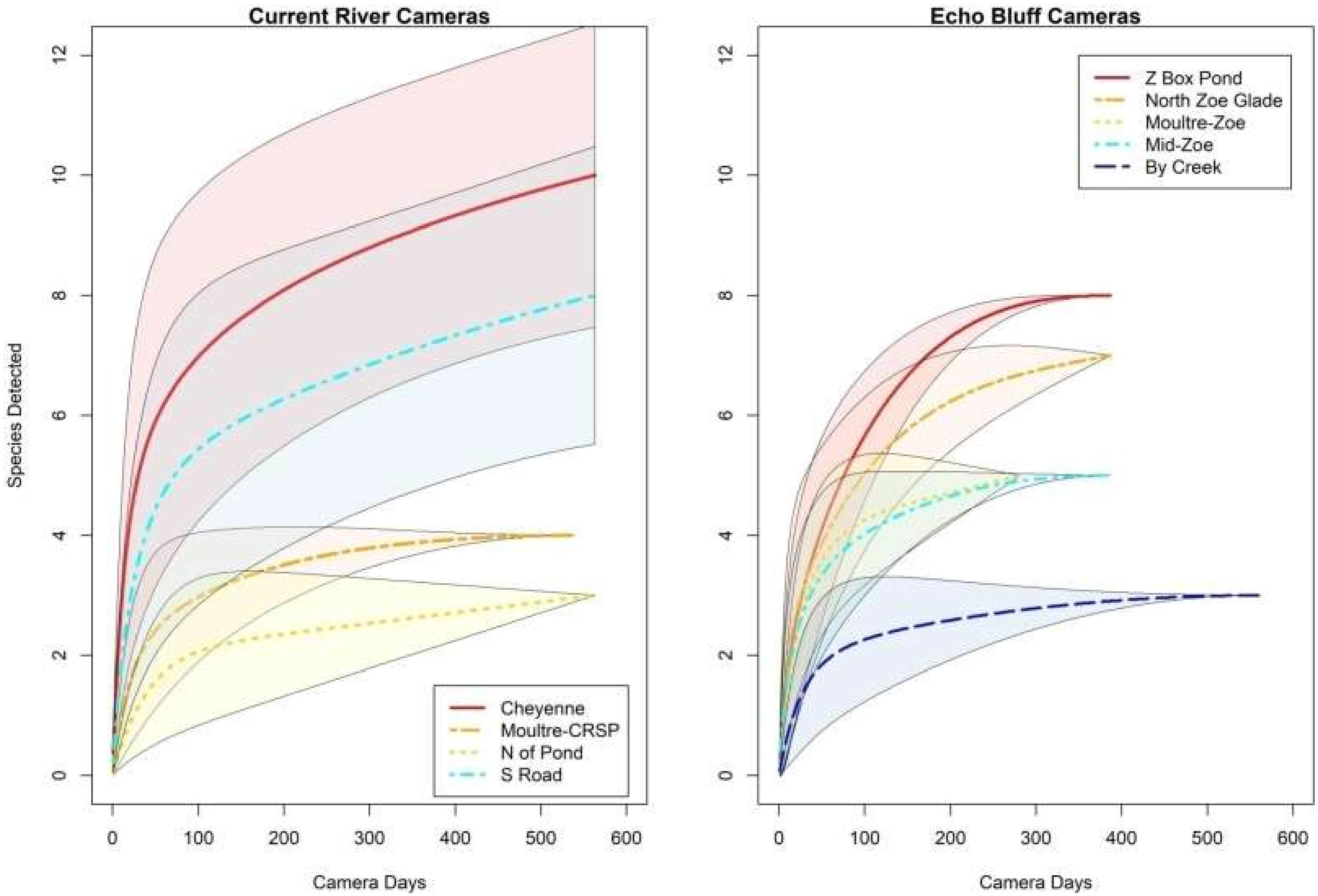

Figure 5. Species rarefaction curves and their associated confidence intervals (shaded areas) for each camera within each park.

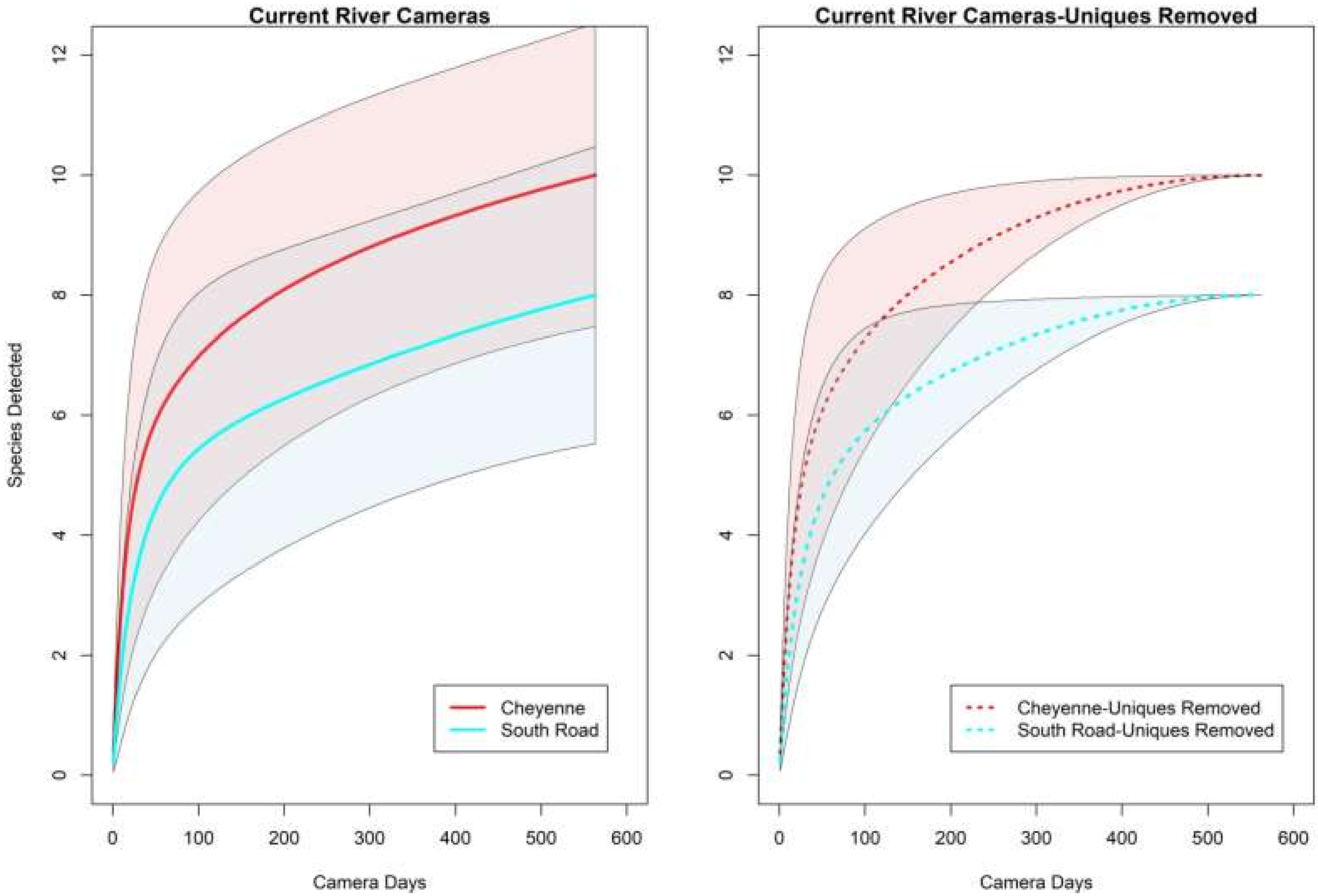

Figure 6. Comparing collected data with replacing uniques with duplicates and their associated confidence intervals (shaded area) on cameras Cheyenne and S Road.

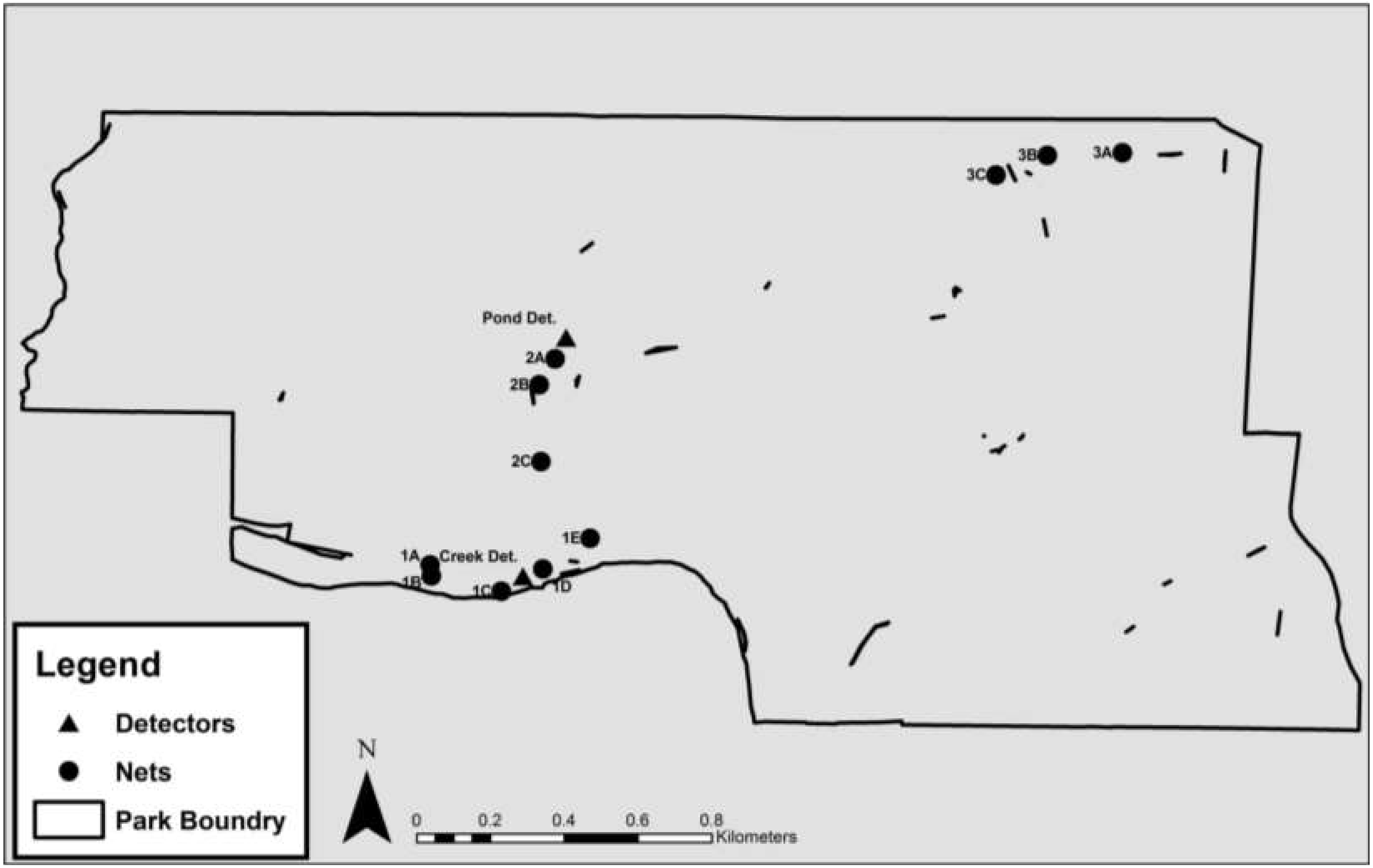

Figure 7. Locations of all bat acoustic detectors and mist netting sites inside Current River State Park and their identifying names.

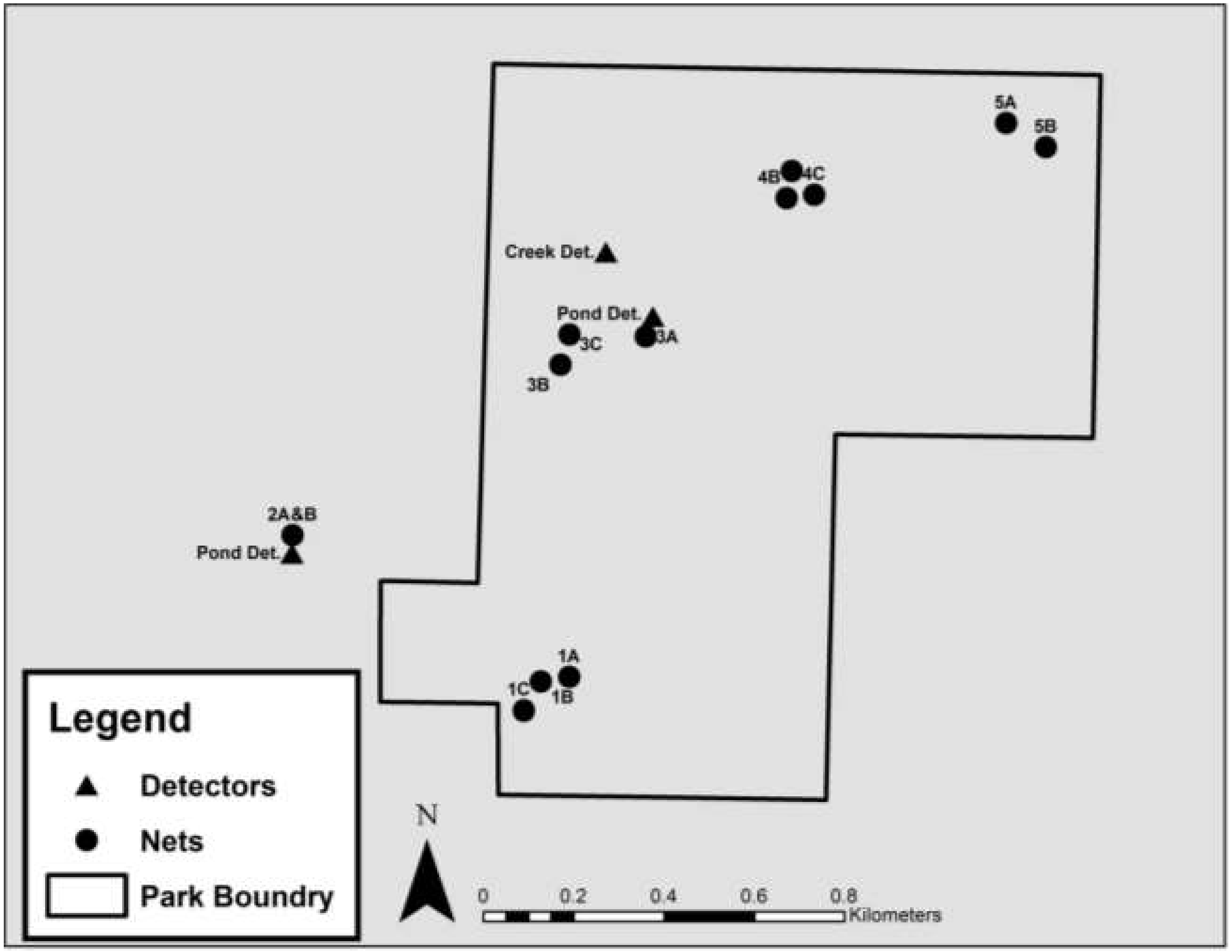

Figure 8. Locations of all bat acoustic detectors and mist netting sites inside Echo Bluff State Park and their identifying names.

Figure 9. An example of an acoustic detector set-up.

Figure 10. Ant colony occupying weatherproof box and detector resulting in corrupted data and detector.

Figure 11. An example of a mist net location and set-up.

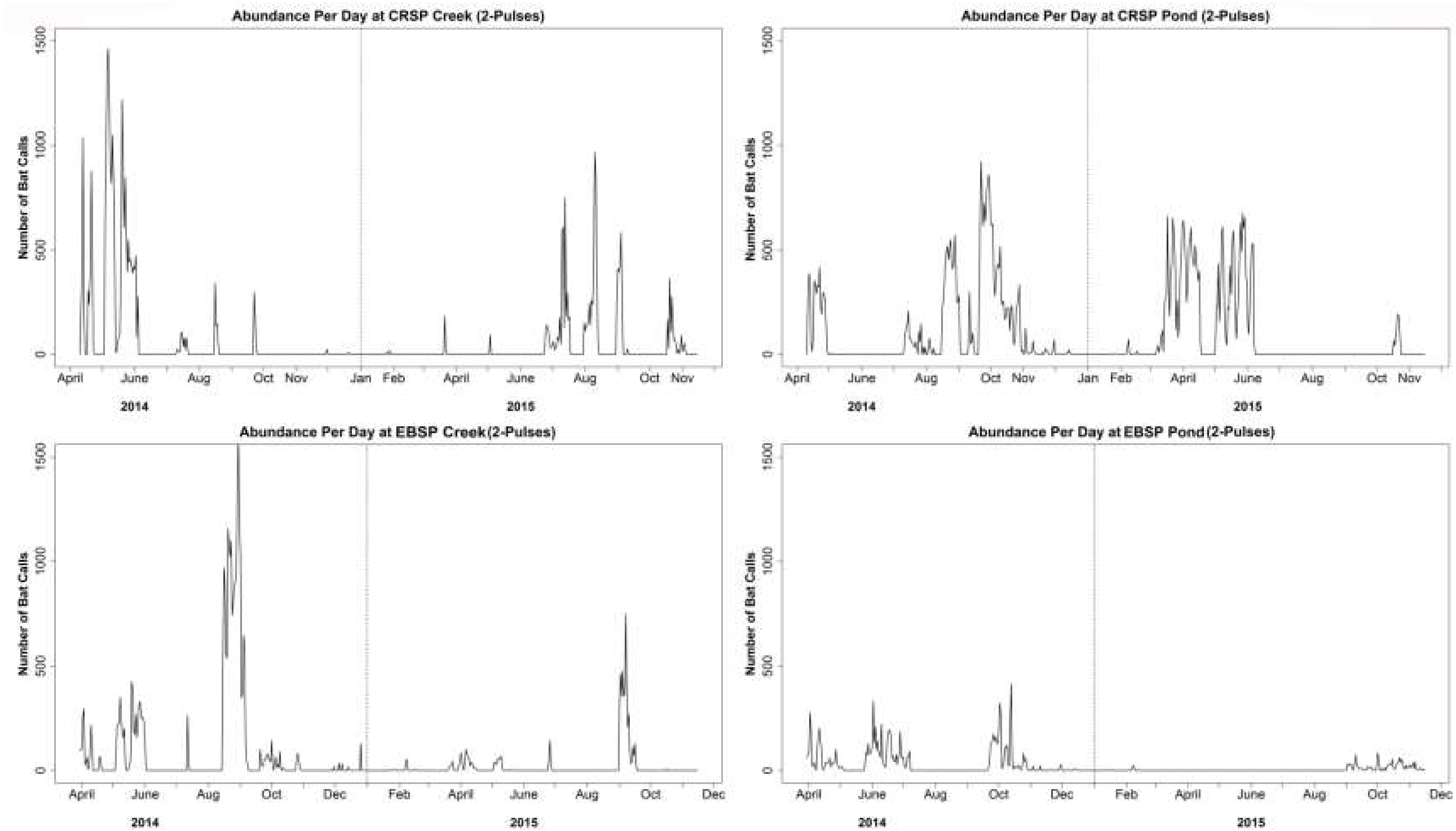

Figure 12. The raw abundance of bat calls for each detector per day, using two echolocation pulses to identify a bat call.

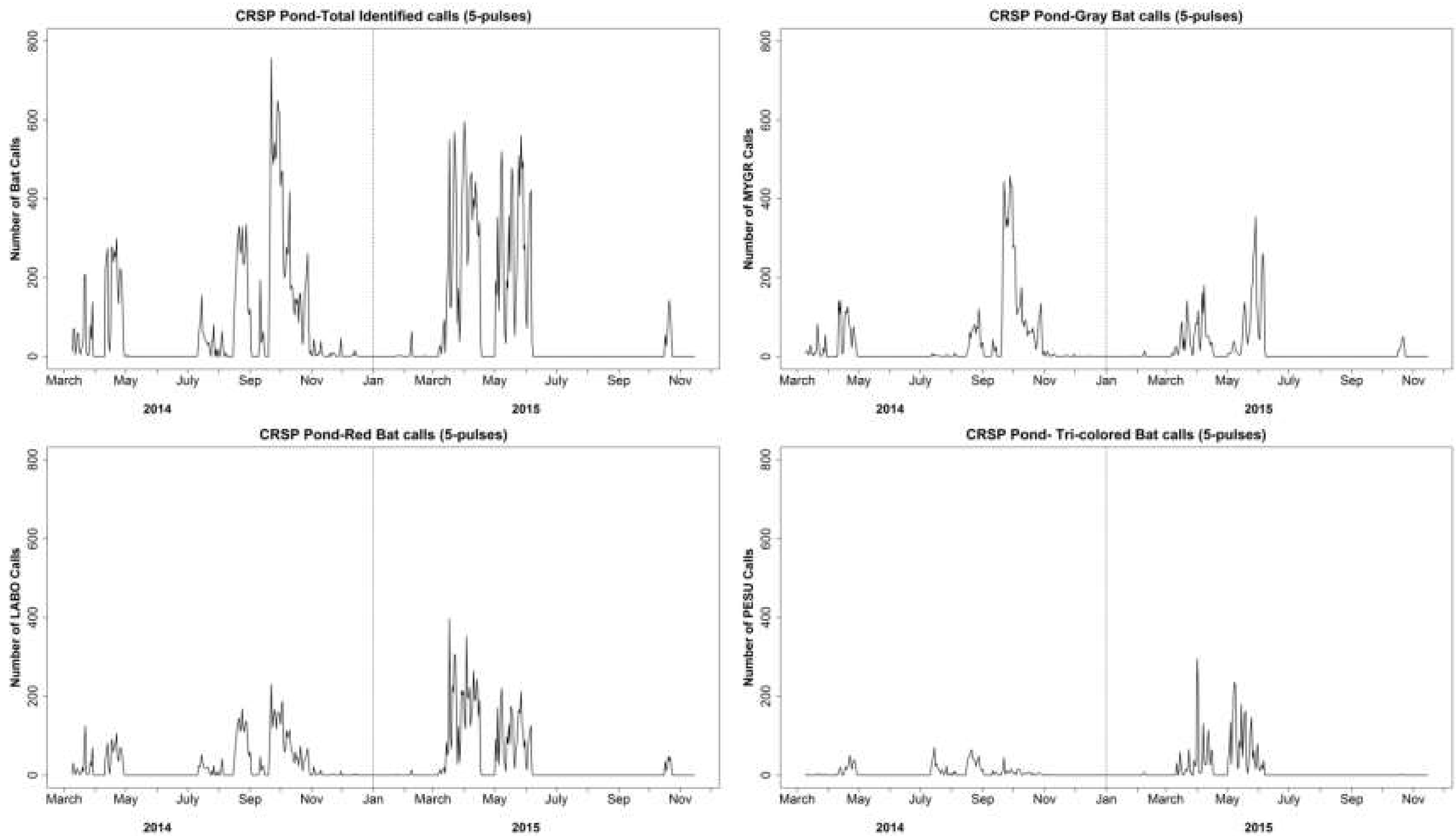

Figure 13. Comparison of three different bat species (gray bats, red bats, and tri-colored bats) compared to the total abundance of calls using 5 echolocation pulses to identify a call at the CRSP Pond site.

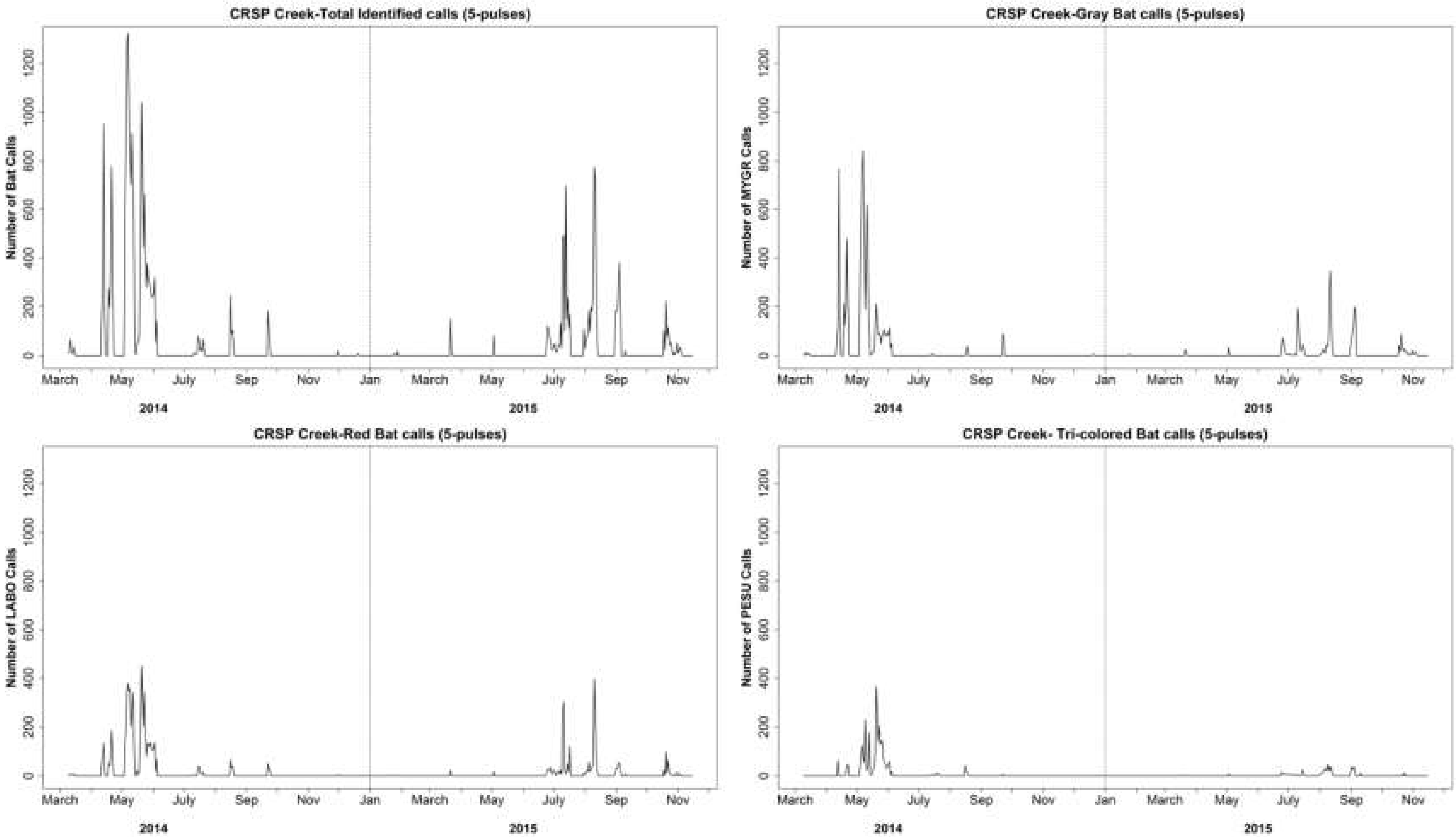

Figure 14. Comparison of three different bat species (gray bats, red bats, and tri-colored bats) compared to the total abundance of calls using 5 echolocation pulses to identify a call at the CRSP Creek site.

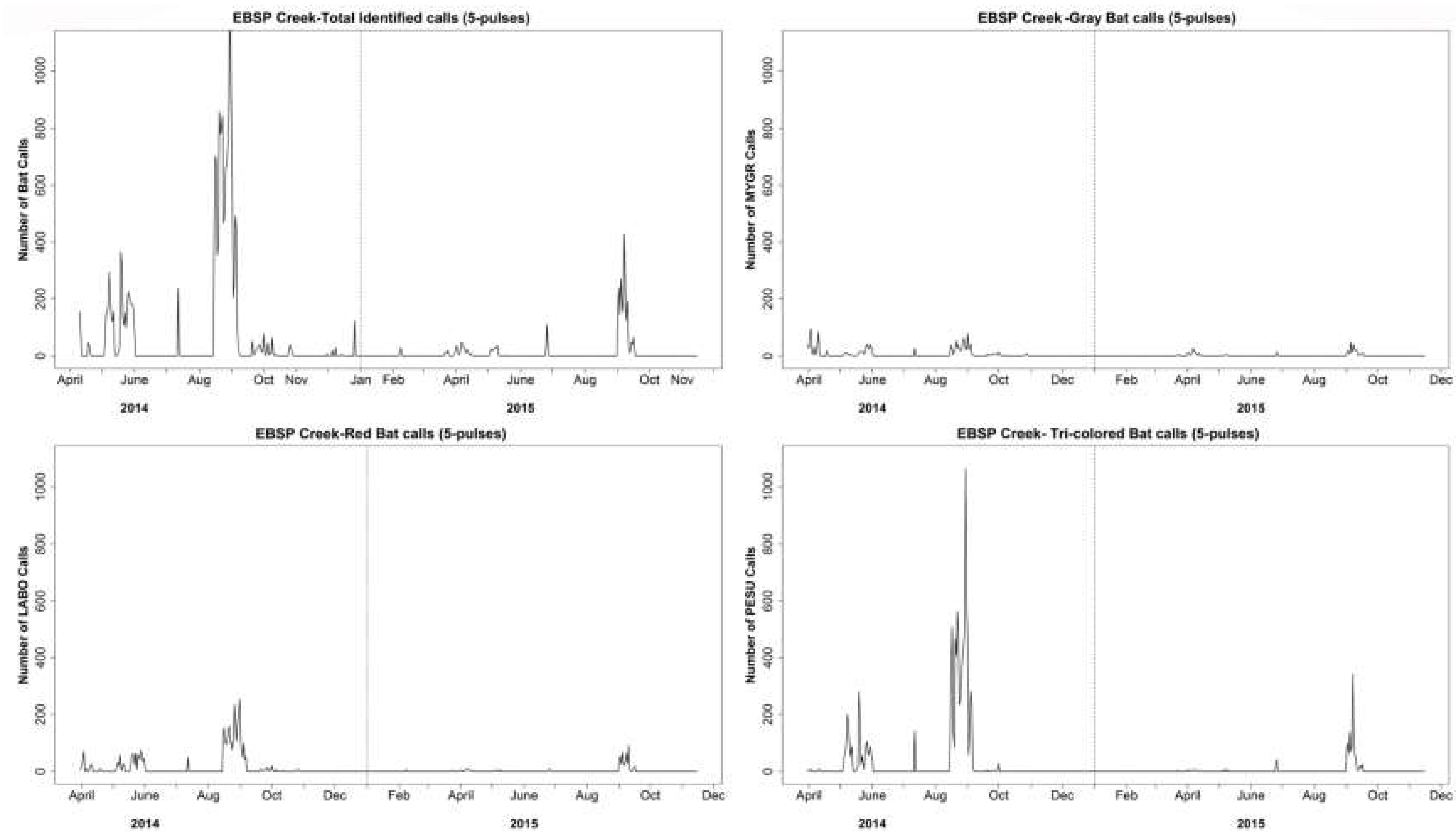

Figure 15. Comparison of three different bat species (gray bats, red bats, and tri-colored bats) compared to the total abundance of calls using 5 echolocation pulses to identify a call at the EBSP Creek site.

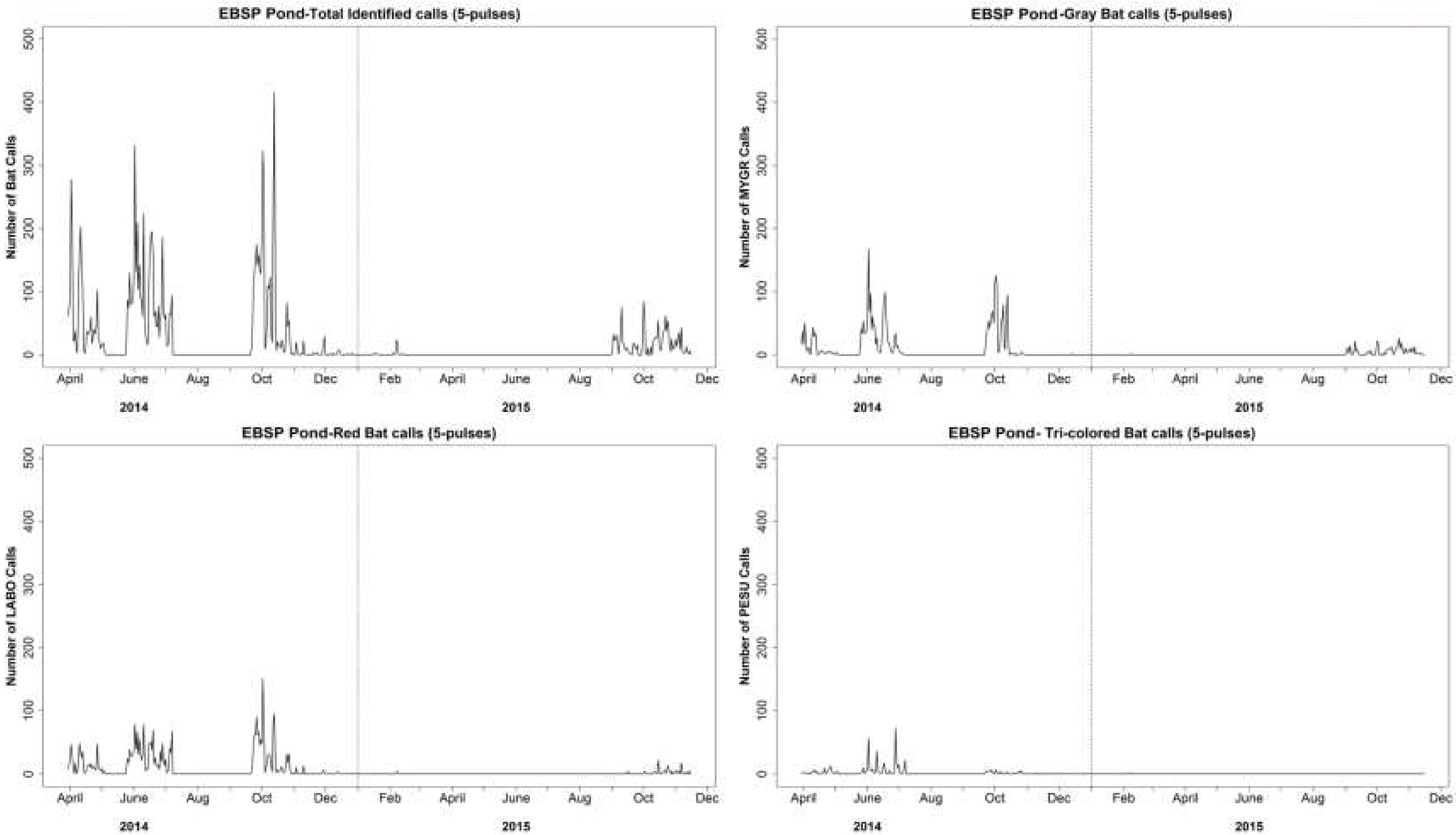

Figure 16. Comparison of three different bat species (gray bats, red bats, and tri-colored bats) compared to the total abundance of calls using 5 echolocation pulses to identify a call at the EBSP Pond site.


Figure 17. Example of a Sherman trap setup location.

Figure 18. Example of a pitfall setup with drift fence in an area with natural structure.


Figure 19. Example of a pitfall set up with drift fence in an open glade area.