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Habitat Associations with Small Mammal Communities at Wilson's Creek National Battlefield

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HABITAT ASSOCIATIONS AND SMALL MAMMAL COMMUNITIES AT WILSON'S CREEK NATIONAL BATTLEFIELD

A Masters Thesis
Presented to
The Graduate College of
Missouri State University

In Partial Fulfillment
Of the Requirements for the Degree
Master of Science, Biology

By
Stephanie A. Ellison
August, 2017

HABITAT ASSOCIATIONS AND SMALL MAMMAL COMMUNITIES AT WILSON'S CREEK NATIONAL BATTLEFIELD

Biology

Missouri State University, August 2017

Master of Science

Stephanie A. Ellison

ABSTRACT

The purpose of this study was to aid the National Park Service at Wilson's Creek National Battlefield with the reported habitat monitoring and management goals through vegetation and wildlife surveys within the park. I provided a description of two major habitat types that are of ecological concern, which included non-native ruderal grasslands and upland deciduous woodlands and forests. I evaluated small mammal communities to determine factors that may affect the detection of individual species and examined habitat associations with occupancy, as small mammals are good indicators of habitat quality. My study highlights the need to manage invasive species such as Japanese honeysuckle (*Lonicera japonica*), hackberry (*Celtis occidentalis*), and Eastern red cedar (*Juniperus virginiana*). Small mammal detectability was influenced by temperature, precipitation, nightly sky illumination, and Julian date, and occupancy was influenced by patch size, vegetation composition and micro-habitat structure. Further monitoring of small mammals could improve habitat associations and account for dynamic population fluctuations. Additionally, the dominance of the white-footed mouse (*Peromyscus leucopus*) in the forests may have implications in disease ecology if stands are not managed within this fragmented landscape.

KEYWORDS: National Park Service, Wilson's Creek National Battlefield, habitat management, monitoring, invasive, small mammal, occupancy, white-footed mouse

This abstract is approved as to form and content

Sean P. Maher, PhD
Chairperson, Advisory Committee
Missouri State University

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Approved:

Sean P. Maher, PhD

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OVERVIEW

I examined 3 geographical scales at Wilson's Creek National Battlefield to describe the overall structure of the landscape and how these characteristics potentially influence small mammal communities within the park. Chapter 1 describes the level of fragmentation by using a standardized land-cover map provided by the Missouri Resource Assessment Program and the Heartland Inventory and Monitoring Network in software program FRAGSTATS. I also collected field measurements to compare the patch characteristics of 4 non-native ruderal grasslands, and 4 upland deciduous woodland and forests within the park.

Chapter 2 describes small mammal communities that were live-trapped at the sampled habitat patches from Chapter 1. I used logistic regression models to test covariates that may influence the detectability of small mammals. I also used presence/absence data to build single-species, single-season occupancy models that tested whether the vegetation metrics from Chapter 1 influence habitat selection. This study highlights the importance of continued vegetation monitoring in sampled habitat patches to identify invasive species as soon as possible, and to determine ecological shifts in vegetation and successional state of these patches. Small mammals are good indicators of the biotic integrity of these patches, as they respond quickly to disturbance, so continued sampling is recommended.

CHAPTER 1: HABITAT STRUCTURE AT WILSON'S CREEK NATIONAL BATTLEFIELD

Introduction

Grasslands in North America have been diminished to highly fragmented patches due to intensive agriculture and urbanization (Mannouris and Byers, 2012), with as little as 1% of the pre-European grasslands remaining in some regions such as Iowa (Dornbush, 2004). The urgent need for conservation and restoration of remaining grasslands has led to ongoing management actions by state and federal agencies, which include seeding (Rossiter, et al., 2016), prescribed burning (Clubine and Davis, 1993), treatment with herbicides (Robertson, et al., 2013), and mowing (DeJaco and Batzli, 2013). However, the restoration of prairies requires several centuries (Samson and Knopf, 1994) and, despite conservation efforts, prairies are still declining (Zylka, et al., 2016). In Missouri, less than 36,500 ha of pre-European grasslands remain, as most have been converted to agricultural use or native grasses have been replaced by imported cool-season fescue (*Festuca*, spp.; DeGolier, et al., 2015; Jacobs et al., 2012; Mulligan, et al., 2013).

European settlement also led to extensive deforestation in North America where by the 18th and 19th centuries the forests on the Atlantic coast were nearly all cleared for anthropogenic development, with rapid destruction spreading westward (Pimm and Askins, 1995). In Missouri, deciduous forests have been significantly impacted by historical burning, grazing, and intensive logging, leaving only a few low-quality stands in the Missouri Ozark region (Missouri Department of Conservation, 2014). Although

Missouri forests are now expanding, oaks (*Quercus* spp.), the typical dominants in this region, have been rapidly declining due to insects, disease, weather, and drought (Piva et al., 2016). Furthermore, fire suppression has facilitated the replacement of oaks by shade-tolerant maples (*Acer* spp.; Cowell, et al., 2010; Nigh, et al., 1985), hackberry (*Celtis occidentalis* L.; Abrams, 1986), and Eastern red cedar (*Juniperus virginiana* L.) that outcompete oak saplings (Piva et al., 2016). Oaks contribute to several ecological functions and provide highly nutritional and widely available food for wildlife (Pierce II et al., 2013), so the high mortality and low regeneration rates are concerning to land managers (Piva et al., 2016).

Wilson's Creek National Battlefield (WICR) is a 910ha (D. Peitz, personal communication, December, 3rd, 2015; wildlife ecologist for the Heartland Inventory and Monitoring Network of the National Park Service at Wilson's Creek National Battlefield (david_peitz@nps.gov)) historical landmark of the 1861 Civil War Battle of Wilson's Creek located in southwest Missouri (Fig. 1; 37.1158° N, 93.4190° W). The historic landscape of the park at the time of the battle consisted of oak woodland, tallgrass prairie, and dense mesic forests, but by 1861 mostly had been converted to overgrazed agricultural land (Annis et al., 2011). WICR now is comprised of about half open and half forested areas that are fragmented with much of the native vegetation replaced due to fire suppression (Diamond et al., 2012). Native vegetative species are rapidly declining in abundance and distribution, and immediate restoration actions have been recommended (National Park Service, 2004). The National Park Service has committed to restoring the historic conditions of WICR and enhancing visitor experiences by assessing the conditions of each area of the park and taking appropriate management actions (Annis et al., 2011; National Park Service, 2003).

To aid the WICR monitoring and management goals, I provided a description of major habitat types within the park that are of ecological concern. I investigated non-native ruderal grasslands and upland deciduous woodland and forests, addressing two dimensions of vegetation 1) structure within the overall landscape and 2) site differences between sampled patches with respect to composition. I addressed the question of patch dynamics by using program FRAGSTATS (v.4.2.1; McGarigal, 2012. Spatial pattern analysis program for categorical and continuous maps. University of Massachusetts, Amherst.), and compared field measurements in 8 sites to determine compositional differences.

Methods

Study area. In 2013, restored tallgrass prairie covered less than 20 ha in WICR, whereas non-native ruderal grasslands spanned approximately 187 ha (Diamond et al., 2012). Within and surrounding the park are large areas of tall fescue (*Festuca arundinacea* Schreb.) and successional woodlands (G. Sullivan, personal communication, September 9, 2016; chief of resource management of the National Park Service at Wilson's Creek National Battlefield (gary_p_sullivan@nps.gov)), which may act as source populations that could lead to replacement and homogenization of current vegetative structures absent preventative measures (Annis et al., 2011). The National Park Service has been making efforts to monitor the grasslands and restore native vegetation through prescribed fire and mowing (Mlekush and James, 2012).

Upland deciduous woodland and forests cover approximately 241 ha of WICR (Diamond et al., 2012). Almost all of the forest stands are impacted by fire suppression, and all stands are characterized by a relatively low oak species basal area, low sapling cover, and dominance of early successional species (Annis et al., 2011). Although native to Missouri, the invasive spread of Eastern red cedar and hackberry in these forests is problematic, as they dominate and homogenize the landscape, reduce ecological functions and increase wildfire susceptibility (Piva et al., 2016), especially in areas with shallow soils (Fig. 2). In 2011, the WICR management goal was to restore oak woodlands by reducing the growth of Eastern red cedar, other shade tolerant mesic tree species, and shrubby understory through regular burns (Annis et al., 2011).

Sample sites. A classified land-cover map of WICR was provided by the Missouri Resource Assessment Partnership (MoRAP) at the University of Missouri, and the Heartland Inventory and Monitoring Network (HTLN) of the National Park Service (Diamond et al., 2012) via the Integrated Resource Management Applications (IRMA) data store (https://irma.nps.gov). I selected 4 non-native ruderal grassland and 4 upland deciduous woodland patches to survey for small mammals and quantify vegetative composition and structure (Fig. 2; Chapter 2). As such, these locations were minimally sized to include centrally located 200m line transects, which were placed in each patch using ArcMap (v.10.3; Environmental Systems Research Institute (ESRI). 2012. ArcGIS Release 10.1. Redlands, CA.), and coordinates were uploaded into a Garmin (GPSMAP® 64s) unit with the Minnesota DNRGPS software package (v.6.1.0.6; http://www.dnr.state.mn.us). I used program 'R' (v.3.3.2; R Core Team, 2016) for all statistical analyses and used the 'ggplot2' (v.2.2.1; Wickham, 2009) package for figures.

I converted the WICR vector dataset to a 1m raster dataset in ArcMap to input into program FRAGSTATS. I set the FRAGSTATS analysis parameters to use an 8-cell neighborhood rule with no sampling strategy selected. I treated water as a background feature to avoid including aquatic features in analysis. I then calculated the number of patches and the area-weighted mean patch size as landscape-level metrics.

To account for patch size and isolation, I calculated area-weighted mean patch size (P_A), patch density (P_D), and Euclidean nearest neighbor distances (C_I) as class-level metrics in FRAGSTATS. These were combined into a single spatial integrity index (SII) based upon Kapos et al. (2012), which varies from low (0) to high (10), with higher values representing a greater capacity for maintaining species biodiversity. To calculate SII, I standardized data using the 'scale' function, and implemented a user-defined function in program R (Equation 1).

$$SII = (0.25 \times P_A) + (0.25 \times P_D) + (0.50 \times C_I) \qquad \text{Equation 1.}$$

I calculated the area and edge of sampled sites as patch-level metrics in FRAGSTATS. I generated a histogram of sampled patch areas within both land-cover types, and identified where my sampled patches fell in the overall distribution. Patch area and edge are generally correlated variables, so to see if sampled patches followed the anticipated trend, I ran Pearson's correlation tests using the 'cor.test' function.

Field sampling. In the grasslands, I sampled vegetation in each patch 3 times at 4-week intervals between May and July 2017. However, one of the patches was classified incorrectly on the land-cover map, as it had since been converted for agricultural use (G.

Sullivan, personal communication, June, 2016) and one patch was initially uploaded into the GPS incorrectly. Therefore, new sites were designated during the field season. Thus, patch C had a shorter, 2-week interval between sampling periods 1 and 2, and patch D had 1-week sampling interval between each period. At 10m intervals, I randomly tossed a square meter quadrant and measured the vegetation height using a modified Robel pole and proportion of bare ground by ocular estimation within the boundaries. To summarize data across the patch, I averaged data from individual sampling points for the entire transect during each of the three sampling periods.

I determined plant composition during the final week of July because herbaceous species are difficult to distinguish when not flowering. I recorded herbaceous vegetation within the boundaries of the square meter quadrant, and any tree or shrub species in contact with the quadrant that were otherwise too large to fit within the square. Plants were keyed to the lowest taxonomic group possible. I summarized vegetation for each patch to provide a general inventory of species composition, and to use as a measure of richness and diversity.

Similar to the grasslands, I sampled each forest patch 3 times at 4-week intervals from May–July 2017. I used the same methods to measure vegetation height and proportion of bare ground in the forests as I did the grasslands. Additionally, I measured canopy closure at each sample point using a spherical densitometer and then averaged the data from individual sampling points for the entire transect during each of the three sampling periods. I counted the number of downed woody materials (DWM) and the number of rocks within 5m of each sampling point that were at least 2.5in in diameter. DWM can provide habitat for several species and can indicate stand age because older forests typically have more fallen logs (Shirley et al., 1997). I classified fallen log decay based on 1) wood color, 2) presence or absence of twigs and branches, and 3) texture of decaying portions, and used a rank order from 0 to 5, where higher values indicate overall structural integrity (Brown, 1974; Shirley et al., 1997; Waddell, 2002).

To inventory trees, I used the point-centered quarter method (PCQ) to sample the closest tree within 5m per quadrant in which quadrants were defined by cardinal directions (Mitchell, 2015). If no tree occurred with 5m of the sample point in a quadrant, it was recorded as vacant. For each tree, I recorded the species, diameter at breast height (DBH), and height. I classified trees using the US Forest Service Forest Inventory and Analysis schema of live trees based upon their trunk diameter: saplings (2.5–12.6cm), poletimber trees (12.7–23cm), and sawtimber trees (<23cm; https://www.nrs.fs.fed.us/fia/).

Data analysis. I used the 1m^2 quadrant data for grassland patches, and the PCQ data for forest patches to calculate Bray-Curtis dissimilarity indices of sampled vegetative species with the 'vegdist' function in the 'vegan' package in program R (v. 2.4-3; Oksanen et al., 2017). I then performed a cluster analysis of these results by using the 'hclust' function to visualize compositional similarities. To account for potential undetected species, I used the 'estimateR' function in the 'vegan' package to find corrected Chao and ACE metrics. To summarize community structure, I calculated Simpson's diversity indices of each patch with the 'diversity' function in the 'vegan' package.

For grasslands, I combined community metrics with the FRAGSTATS patch-level outputs using principal component analyses (PCA) to identify potential patterns within

WICR. Because I had more metrics than observations, I first reduced the dataset by removing highly correlated variables and then principal component analysis was performed using the ‘prcomp’ function. I set the ‘scaling’ and ‘center’ options to true, to adjust for different magnitudes of the measurements that can otherwise can skew the PCA transformation, causing misleading results.

I calculated the absolute density of trees in each forest, and relative importance values for individual tree species at each forest patch following abundance procedures detailed in Mitchell (2015). Importance values are a composite measure of the relative density, relative cover and relative frequency of individual tree species to determine dominant species in a forest patch. I used the Warde correction factor to correct for vacant PCQ quadrants (Warde and Petranka, 1981). I then classified forests into stand-size classes (small, medium, or large) based on the average DBH of the dominant tree species in each stand (Piva et al., 2016).

To make site comparisons, I first wanted to emphasize the ecological contributions of each oak species encountered in the forests, hackberry, and Eastern red cedar, because these are management targets at WICR (Annis et al., 2011). To test for differences between patches for the number of rocks, fallen logs, saplings, poletimber, and sawtimber trees, I used a series of ANOVAs with the ‘aov’ function. If the ANOVA was significant, I used Tukey’s Honestly Significant Difference post hoc tests with the ‘TukeyHSD’ function to determine what groups were significantly different. Finally, I reported the hardwood basal areas, hardwood tree height, and oak basal areas from each site and the WICR 2008 values, and target goals of 2011.

Results

There were 329 calculated terrestrial habitat patches, with an area-weighted mean of 16.12 hectares. Spatial integrity indices for non-native ruderal grasslands and upland deciduous woodland and forest were 7.22 and 5.78, respectively. Patch sizes of the sampled non-native ruderal grassland ranged between 2.62 ha and 9.15 ha, with a positive correlation with edge ($r = 0.99$, $P = 0.01$; Tab. 1). Patch sizes of the sampled upland deciduous woodland and forest patches ranged between 6.65 ha, and 22.87 ha, with a positive correlation between area and edge ($r = 0.98$, $P = 0.02$; Tab. 1). The sampled forest and grassland patches represent a breadth of patch sizes within WICR (Fig. 3). There were 2 large grassland patches (41, and 48 hectares) that were classified as non-native ruderal grasslands, but had since been converted for current agricultural use and therefore, are not included in the histogram for this reason (G. Sullivan, personal communication, June, 2016).

There were 42 total species found in the non-native ruderal grasslands: 5 tree species, 6 shrub species, and 31 herbaceous species (Appendix A.). Average vegetation heights ranged between 89–94cm throughout the course of the summer, and the percentage of bare ground ranged between 0–22% over the summer (Tab. 1). The Chao and ACE estimated species richness for all grasslands was 45 and 50, respectively, and estimates for individual patches were relatively equal (Tab. 1). Simpson's diversity was high for all patches and as a collective site estimate (0.95; Tab. 1). For grassland principal component analyses, I excluded patch edge, percentage of bare ground, ACE richness, and Simpson's diversity indices because of high intercorrelations of variables, and retained patch area, vegetation height, and Chao estimated richness. Principal component

1 (PC1), explained 54.5% variation (Fig. 4), with mixed, relatively high component loadings for the original variables: patch area, and Chao estimated richness (-0.67 and 0.61, respectively). The second principal component (PC2) explained 29.9% variation (Fig. 4). PC2 also had mixed component loadings with a strong positive correlation with vegetation height (0.88), and negative correlation with Chao estimated richness (-0.45). Cluster analysis of Bray-Curtis dissimilarity indices showed a forked pattern, where patches A and B had similar composition, as did C and D (Fig. 5).

There were 22 tree species found collectively in the upland deciduous woodland and forest sites (Appendix B.) Average vegetation heights ranged between 66–75cm throughout the course of the summer, and the percentage of bare ground ranged between 2.2–30% over the summer (Tab. 1). Collective species richness estimates for both Chao and ACE was 24 species, but estimates among individual patches varied (Tab. 1). Simpson's diversity was high for all patches and as a collective site estimate (0.91; Tab. 1). Forest composition showed a nested pattern where patches E and F had similar species composition and H had the least similar composition (Fig. 5).

The estimated absolute tree density ranged from ~850 trees/ha to over 1000 trees/ha; patch E had the greatest density of trees, and patch H had the lowest tree density (Tab. 2). Hackberry, black walnut (*Juglans nigra* L.), slippery elm (*Ulmus rubra* Muhlenberg) and Osage orange (*Maclura pomifera* Rafinesque) were the dominant tree species in patch E (Tab. 3). Patch F was characterized by Eastern red cedar, slippery elm, hackberry, chinkapin oak (*Quercus muehlenbergii* Engelmann), black oak (*Quercus velutina* Lamarck), black walnut, and black hickory (*Carya texana* Buckley; Tab. 3). Patch G was mostly comprised of black oak, chinkapin oak, and red oak (*Quercus rubra*

L.), and to a lesser degree, hackberry, sassafras (*Sassafras albidum* Nuttall), honey locust (*Gleditsia triacanthos* L.) and white ash (*Fraxinus americana* L.; Tab. 3). Patch H was primarily post oak (*Quercus stellata* Wangenheim) dominant, with Eastern red cedar, red oak and black walnut contributing to a much lesser degree (Tab. 3). The patch stand-size classifications ranged between medium to large DBH of dominant trees (Tab. 2).

Hackberry was more dominant in patch E than other sites, while Eastern red cedar was more dominant in patches F and H, and individual oak species importance values were in the low to middle ranges for all patches, expect in patch H (Fig. 6). There were significant differences in the number of saplings ($F_{3,76} = 4.24$; $P = 0.01$) and sawtimber trees ($F_{3,76} = 3.72$; $P = 0.02$) between patches, but not poletimber counts ($F_{3,76} = 1.36$; $P = 0.26$; Fig. 7). Post hoc tests showed there are significant differences in the sapling counts between patches H and F ($P = 0.03$), and H and G ($P = 0.01$; Fig. 7). There were significant differences in sawtimber counts between patches G and E ($P = 0.07$), and between H and G ($P = 0.03$; Fig. 7). There were no significant differences in the number of rocks for any patches ($P = 0.14$; Tab. 2), but there were differences in the number of fallen logs ($F_{3,76} = 3.64$; $P = 0.02$; Tab. 2). Post hoc tests revealed the number of fallen logs was significantly different between patches E and H ($P = 0.03$).

Overall hardwood basal area (m^2/ha) included all species besides Eastern red cedar (Appendix B). Three of 4 patches had higher values than 2008 and were in the goal range of values set in 2011, with patch G the exception in both cases (Tab. 4). Oak basal area (m^2/ha) was higher than WICR recorded values in 2008, but patches E, F, and G had values lower than the management goal set in 2011 (Tab. 4). Average tree height was above 2008 reported values, and the 2011 management goals in E, G, and H.

Discussion

Landscape structure. Habitat heterogeneity is an important component of landscape composition to support wildlife populations, as there is often a positive correlation between resource variation with species diversity and abundance (Heinen and Cross, 1983; Tews et al., 2004). High spatial integrity of patches can also be important for vegetation species richness in a heterogeneous landscape (Holt, et al., 1995). Previously, Annis et al. (2011) characterized the number of patches at WICR, and the geometric mean patch size by using a developmental, broad-scale land-cover classification system, but with a fine-scale vegetation map. From this report, the landscape at WICR was believed to be more fragmented than historic conditions with 1,819 patches at a geometric average size of 0.44 ha, concerning resource managers (Annis et al., 2011). A revised land-cover map was developed in a vegetation-mapping and classification initiative described by Diamond et al. (2012), and the number of patches and geometric mean sizes were again reported, but at a broader map resolution, resulting in 379 polygons with an average size of 2.1 ha (Diamond et al., 2012). My analysis of this map using a finer raster resolution characterized the landscape as 329 polygons with an area-weighted mean of 16.12 ha. This does indicate a fragmented landscape, although not to the great extent as previously reported (Annis et al., 2011). Additionally, evaluating the patch area-weighted mean is a more ecologically relevant metric than the geometric mean, as the continuous area of one large patch has a greater ecological impact by outweighing certain edge effects of several small patches (Kapos, et al., 2002; Leitao, et al., 2006).

Spatial integrity indices can aid WICR park managers in setting reasonable management goals, and deliver meaningful information to a broad audience by combining three metrics important for sustaining biodiversity (Tierney et al., 2009), rather than simply patch count and geometric mean patch size as previous reports (Annis et al., 2011). Providing spatial integrity indices for two of the major vegetative classification types within the park represents a starting point for assessing the biodiversity potential of these habitats (Kapos et al., 2002; Piva et al., 2016). Non-native ruderal grasslands had a higher spatial integrity index than the upland deciduous woodland and forests, indicating the grassland patches may be larger with greater connectivity.

It is essential to monitor patch-level compositional and ecological trends as indicators of successional state and overall health to best manage stands and predict future changes in response to invasive species, potential management, and recreational activity (Piva et al., 2016). The plots I sampled do not coincide well with the long-term vegetation sampling locations at WICR (Diamond et al., 2012; Mlekush and James, 2012). However, future monitoring of these patches in addition to the existing sites would be beneficial, because they represent a wide proportion of the respective sizes within the park.

Habitat characteristics. The principal component analyses in the grassland sites identified that patch area contrasted with vegetation richness along the first axis to explain the majority of the total variation in PC1. Principal component 2 was explained by vegetation height contrasting with Chao richness on the second axis. It appears from the results that PC1 scales from large to small patches, and PC2 scales short to tall vegetation heights, while Chao estimated richness further separates sampled patches on both axes. However, without enough sample patches to incorporate all of the grassland metrics, these are tentative conclusions as there does not seem to be stark differences in vegetation height or Chao estimated richness. It is likely patch area explains most of the variability between patches, but a larger sample size may indicate further differences.

Restoring the native prairie vegetation is a primary management goal within the park, but non-native ruderal grasslands receive little to no attention. My sampled grassland sites have no ongoing or routine management (S. Leis, personal communication, December 12, 2016, fire ecologist for the Heartland Inventory and Monitoring Network (sherry_leis@nps.gov); J. Bell, personal communication, January 3, 2017, project manager of exotic plant management for the Heartland Inventory and Monitoring Network (jordan_bell@nps.gov)), and the prescribed burns are either infrequent or non-existent. Considering the vegetative species composition and the patch metrics, the previous burns appear to be ineffective over several years; the most recently burned site (A, 2010) has a similar structure to a site (B, 1997) burned just previously to a nearby site (D, 1998), which is most similar to site that has not been burned (C).

My sampled forest patches had varied tree richness and diversity estimates, and each were characterized by different dominant tree species. This follows the reported trend of upland forests having inconsistent communities across short distances at WICR (Diamond et al., 2012). The varied species composition and structure found in my forest patches may be beneficial, as diverse forests are often related to disturbance resilience, and can be a positive influence on wildlife communities by providing habitat requirements for different species (Piva et al., 2016).

Although patch E has the smallest area, it had the highest tree density of the sampled patches, and the only patch in the medium stand-size classification group. This indicates that patch E may be in an earlier successional state than the other sites. Without natural or anthropogenic disturbance, the high tree density on such a small patch may have negative implications for tree growth, yield, and persistence at this patch as the trees increase in volume (Piva et al., 2016). The lower tree density and large stand-size classification of patches F, G, and H indicate maturing forests.

When number of saplings and sawtimber trees are viewed together, each patch showed different trends in the proportion of coverage of different tree sizes. The mean number of saplings along transect H was significantly different than in patches F and G, whereas patch G had significantly different sawtimber trees than patches E and H. This indicates that patch H is an older forest where canopy gaps have allowed for tree stratification, whereas patches F and G are more even-aged stands. Monitoring the changes in size-classifications of these patches can help determine successional changes and potential forest sustainability (Piva et al., 2016).

Presently, my sites appear in overall good health when compared to WICR Natural Resource Condition Assessments previously reported. Hardwood basal areas were higher than 2008 records, and within the target goals of 2011 for 3 of 4 patches. Patch G was lower and this is surprising because it is characterized mainly by hardwoods, and had the second highest tree density of all sites. On the other hand, patch F had the softwood dominant, Eastern red cedar, but still surpassed the 2011 target. Oak basal area is higher than the recorded 2008 values for all 4 patches, but only patch H, which is profoundly post oak dominant, had higher values than the management goals of 2011.

Average tree height was greater than 2008 in all patches, but in patch F, average tree height did not meet the management goals of 2011, potentially due to average heights of Eastern red cedar reaching only 17–18m after 50 years (Anderson, 2003).

Non-native and invasive species. Non-native plants were detected along all transects, but perhaps most the problematic grassland discovery was the frequency of the invasive Japanese honeysuckle (*Lonicera japonica* Thunb.) along transect D (Appendix A). Japanese honeysuckle is an aggressive and widespread species that can grow rapidly under several biotic conditions such as acidic or basic soils, or in different habitat types, contributing to reduced species diversity by outcompeting native plants for light gaps and nitrogen-use efficiency (Schierenbeck, 2004). Japanese honeysuckle is on the WICR park-established watch list, with an estimated park-wide frequency of ~38% as of 2006 (Young, et al., 2007). Japanese honeysuckle was found on 1/3 of the sampled non-native ruderal grassland sites in 2011, Once established, Japanese honeysuckle is difficult to remove (Annis et al., 2011), as there are no known native pests or diseases, it is able to recover quickly from herbivory, and mowing encourages growth (Schierenbeck, 2004).

Patch E is mainly characterized by hackberry, which is common in secondary growth forests (Mcbride, 1973) and found frequently in upland deciduous woodland and forest sites at WICR, although not to this extent previously (Diamond et al., 2012). Hackberry may lead to management problems because it is a shade-tolerant, opportunistic, and resilient species capable of replacing young oaks in fire suppressed forests (Abrams, 1986). Hackberry is a soft-mast producing tree, so dominance in an oak-hickory forest may be problematic for wildlife species dependent on oak masts (Missouri Department of Conservation, 2014).

Eastern red cedar was dominant in patch F, but the presence of this species was not previously detected in prior vegetation mapping, as the only prior sampling location in patch F does not coincide with the location of my transect (Diamond et al., 2012). At WICR Eastern red cedar is considered an invasive species that converts the landscape and changes vegetative composition across several habitat types, but is most commonly associated with encroaching upon old fields, glades, and fire suppressed forested areas within the park (Diamond et al., 2012). The discovery of dominant Eastern red cedar should be of management concern if oak regeneration is a priority goal within the park, as Eastern red cedar is able to outcompete oak saplings in Missouri and can quickly become the most abundant tree species (Piva et al., 2016), as demonstrated in patch F.

Management considerations. Fire suppression has allowed non-native and invasive species to replace native vegetation in the grasslands and typical overstory dominants in the forests (Annis et al., 2011). My sampled patches reflect the need for routine prescribed fires, especially in response to the Japanese honeysuckle in patch D, and the Eastern Red cedar in patch F, and surrounding areas. Since these patches are close together, the Eastern red cedar is likely to encroach upon the adjacent grasslands, eroding soils and washing away native seeds while further facilitating the spread of Japanese honeysuckle by shading native vegetation (National Park Service, 2004). In addition, management in patch E is recommended as an early response to the dominance of hackberry.

Frequent high-intensity fires (every 2–5 years) followed by herbicidal treatment is recommended to control dominant Japanese honeysuckle (Schierenbeck, 2004), and other invasive species to facilitate the re-establishment of native grasses that are otherwise inhibited (Annis et al., 2011). My sampled forest sites have never been cut or thinned (J. Bell, personal communication, January 3, 2016) and have not been recently burned, but are target patches for future management (S. Leis, personal communication, September 19, 2016). Forests should not only undergo routine fire regimes, but selective thinning should facilitate oak regeneration and allow forests to reach the later successional states. However, initial stand management to improve spatial integrity of the forests should also be a top management priority.

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Table 1. Patch metrics, observed (OBS.) richness, estimated ACE and CHAO richness, and Simpson's diversity indices for sampled herbaceous vegetation composition at 4 non-native ruderal grassland patches (A-D), and woody composition at 4 upland deciduous woodland and forest patches (E-H), along 200m transects at Wilson's Creek National Battlefield, Missouri; 2016.

Metric	A	B	C	D	E	F	G	H
Area (ha)	5.50	9.15	2.62	3.75	6.55	9.88	22.9	12.2
Edge (km)	2.45	3.22	1.63	2.11	2.40	2.76	10.3	5.38
Veg Height (cm)	91	89	89	94	67	75	66	66
Bare Ground (%)	13	22	0.0	0.0	9.5	2.2	30	8.6
OBS. Richness	18	18	19	15	9	13	19	11
ACE Richness	25	26	26	27	12	14	25	15
CHAO Richness	22	23	24	24	12	13	23	14
Diversity	0.91	0.90	0.93	0.88	0.78	0.89	0.89	0.73

Table 2. Site characteristics of sampled upland deciduous woodland and forest patches (E-H) at Wilson's Creek National Battlefield, Missouri; 2016. Stand-size classification details (small, S; medium, M; and large, L) are published by the U.S. FOREST SERVICE (Piva et al., 2016).

Characteristic	E	F	G	H
Rock Count	149	200	89	190
Fallen Log Count	79	75	51	41
Canopy Closure (%)	94	89	93	89
Tree Density (trees/ha)	1010	867	908	853
Sapling Count	88	82	74	130
Poletimber Count	85	110	82	86
Sawtimber Count	34	20	15	36
Dominant species[1]	CEOC	JUVI	QUVE	QUST
Mean DBH of dominants (cm)	16.36	23.95	39.47	30.23
Stand-size classification	M	L	L	L

[1] Includes (*Celtis occidentalis*; CEOC), Eastern red cedar (*Juniperus virginiana*; JUVI), black oak (*Quercus velutina;* QUVE), and post oak (*Quercus stellata*; QUST).

Table 3. Relative importance values (IMP) for the top 80% contributing tree species in sampled upland deciduous woodland and forest patches (E-H) at Wilson's Creek National Battlefield; 2016.

E	IMP.	F	IMP.	G	IMP.	H	IMP.
Celtis occidentalis	30.85	*Juniperus virginiana*	17.82	*Quercus velutina*	22.30	*Quercus stellata*	53.29
Juglans nigra	26.98	*Ulmus rubra*	16.64	*Quercus muehlenbergii*	20.02	*Juniperus virginiana*	12.44
Ulmus rubra	15.67	*Celtis occidentalis*	12.96	*Quercus rubra*	11.35	*Quercus rubra*	9.37
Maclura pomifera	7.34	*Quercus muehlenbergii*	10.56	*Celtis occidentalis*	6.42	*Juglans nigra*	6.18
		Quercus velutina	9.32	*Sassafras albidum*	5.98		
		Juglans nigra	7.48	*Gleditsia triacanthos*	5.84		
		Carya texana	6.97	*Carya tomentosa*	5.40		
				Fraxinus americana	3.73		

Table 4. Natural Resource Condition Assessments (NCRA's) of key structural and cover attributes in upland deciduous woodland and forest reporting units at Wilson's Creek National Battlefield in 2008, the target management goals of these units as of 2011 (Annis et al., 2011), and the 2016 reported values in 4 sampled patches (E-H).

Attribute	2008	2011	E	F	G	H
Oak basal area (m^2/ha)	4.3	9–18.2	8.1	5.9	8.8	16.6
Hardwood Basal Area (m^2/ha)	15.0	14–23	33.6	18.2	11.5	20.3
Mean Hardwood Tree Height (m)	13.5	13–21.3	24.6	14.9	23.7	24.2

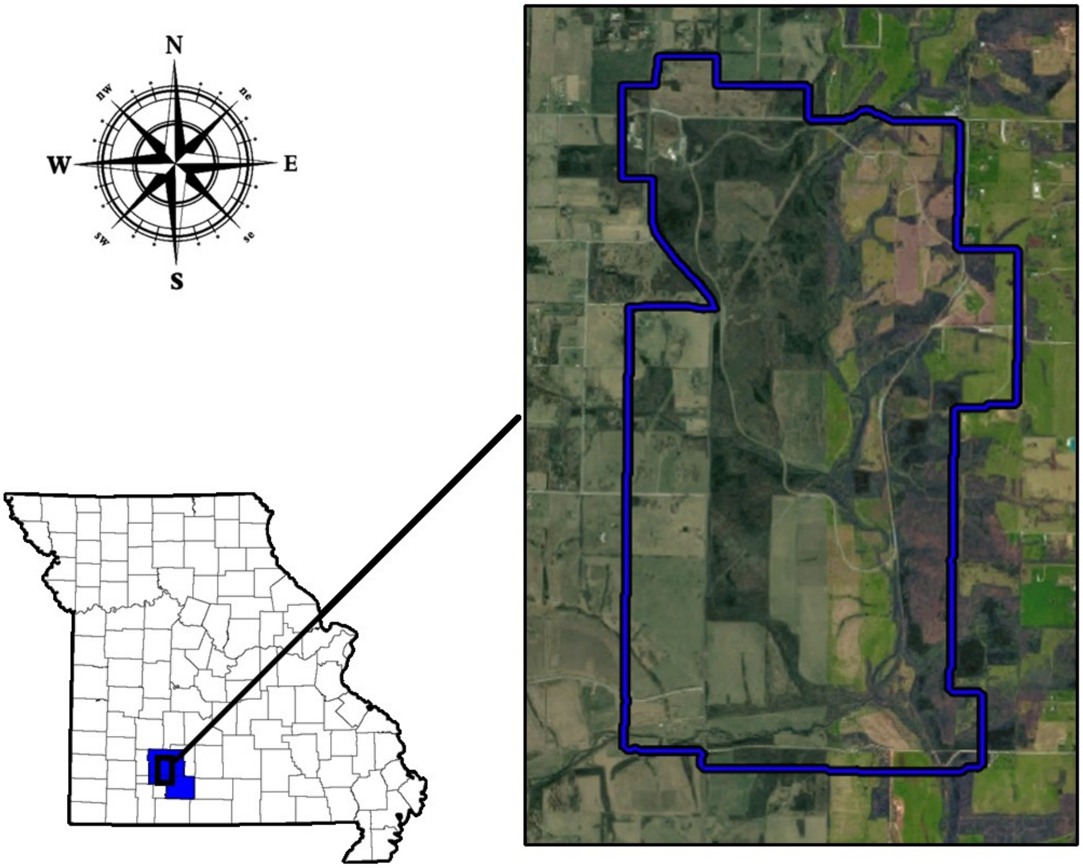

Figure 1. Location of Wilson’s Creek National Battlefield (37.1158° N, 93.4190° W) in Southwest, Missouri. Wilson’s Creek National Battlefield transverses the border of Greene and Christian counties.

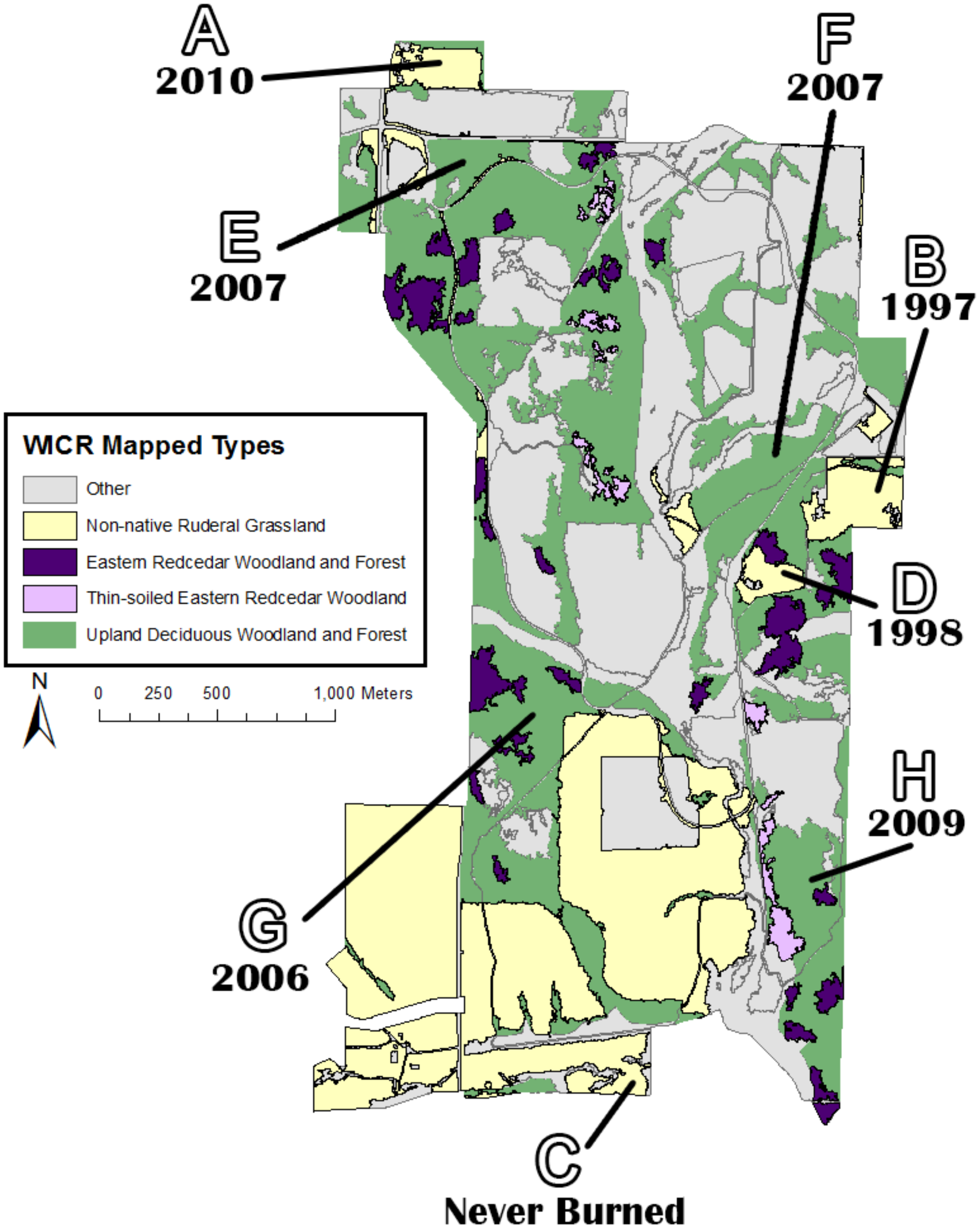

Figure 2. Sampled non-native ruderal grassland sites (A-D) and upland deciduous woodland and forest sites (E-H) at Wilson's Creek National Battlefield, Missouri. The year of the most recent burn is listed for each site. For site 'B' in the non-native ruderal grassland plot, only the eastern half of the patch was burned. Vegetation mapping was provided by the Missouri Resource Assessment Partnership (MoRAP) at the University of Missouri, and the Heartland Inventory and Monitoring Network (HTLN) of the National Park Service (Diamond et al., 2012).

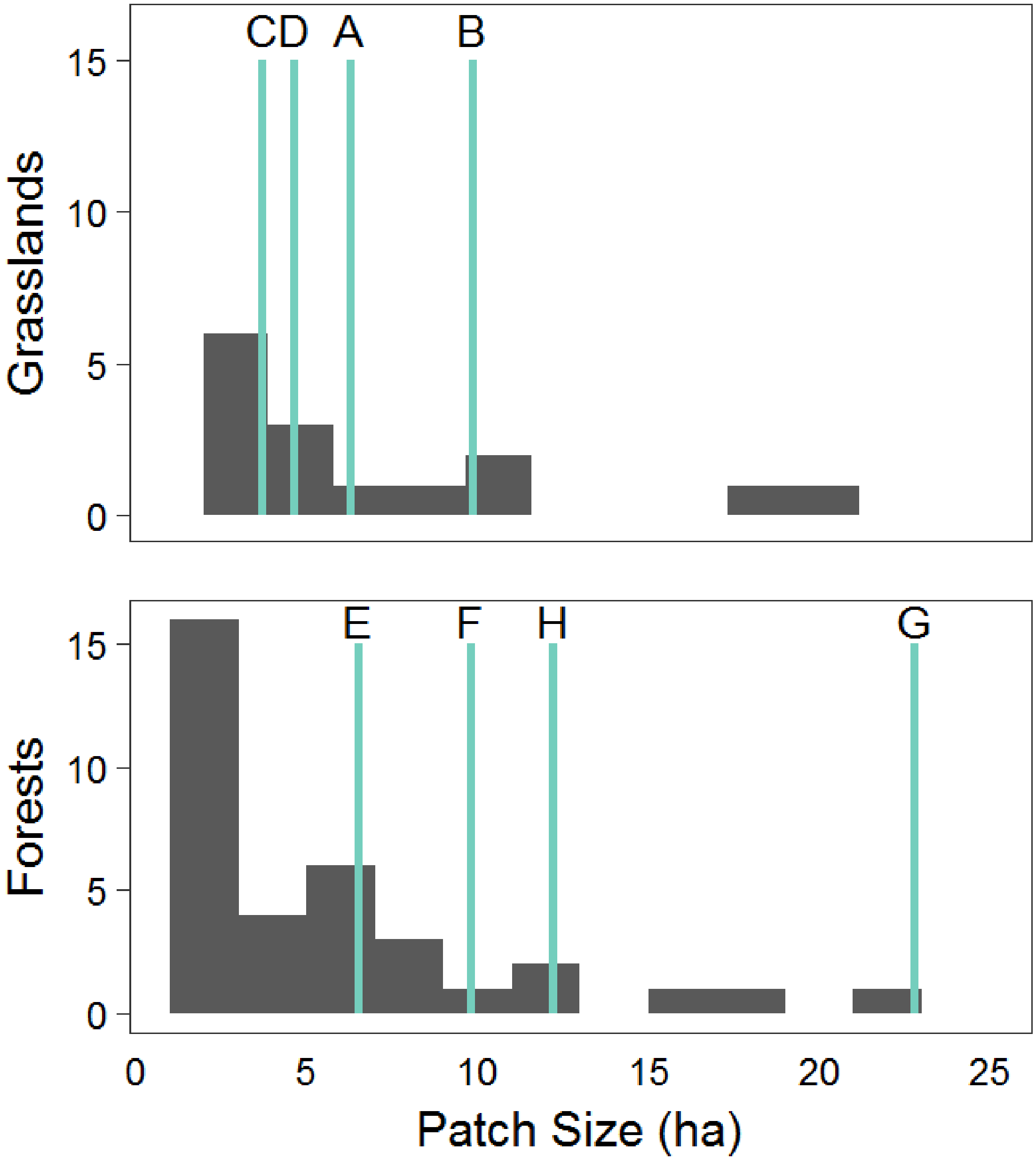

Figure 3. Patch sizes of all non-native ruderal grassland, and all upland deciduous woodland and forest vegetative classifications, and the area distributions of 4 sampled grassland patches (A-D), and 4 sampled forest patches (E-H) at Wilson’s Creek National Battlefield, Missouri.

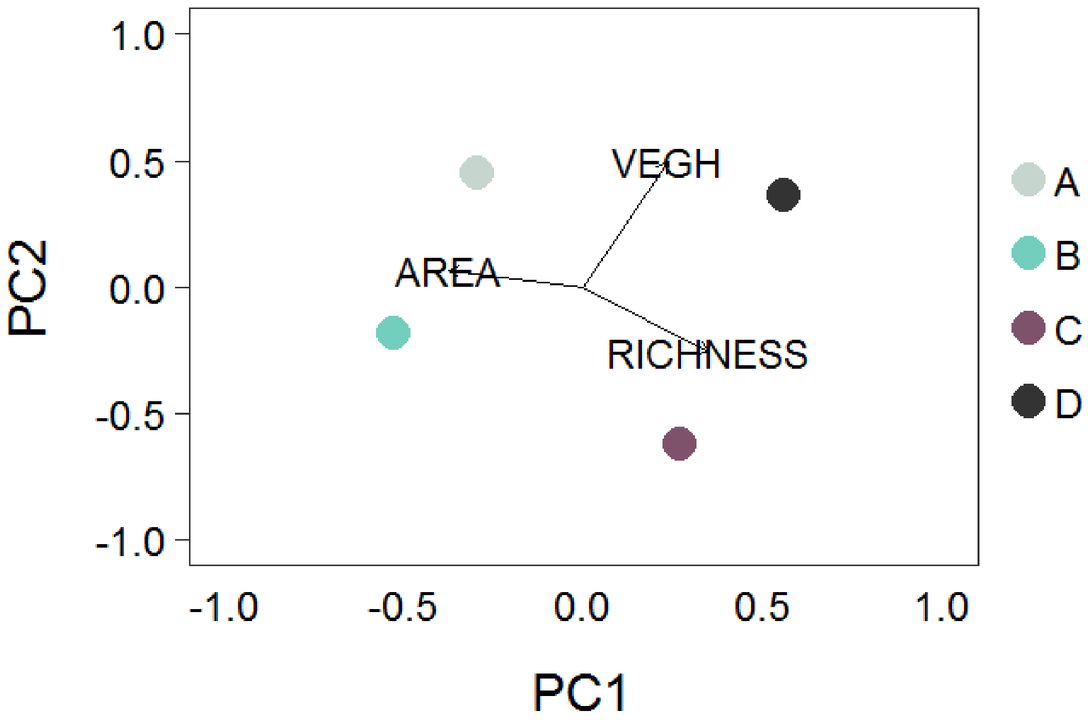

Figure 4. Principal component analysis of 4 non-native ruderal grassland patches (A-D), at Wilson's Creek National Battlefield, Missouri; 2016. Principal component 1 (PC1) contrasts area and Chao richness estimates and principal component 2 (PC2) contrasts herbaceous vegetation height (VEGH) and Chao richness estimates.

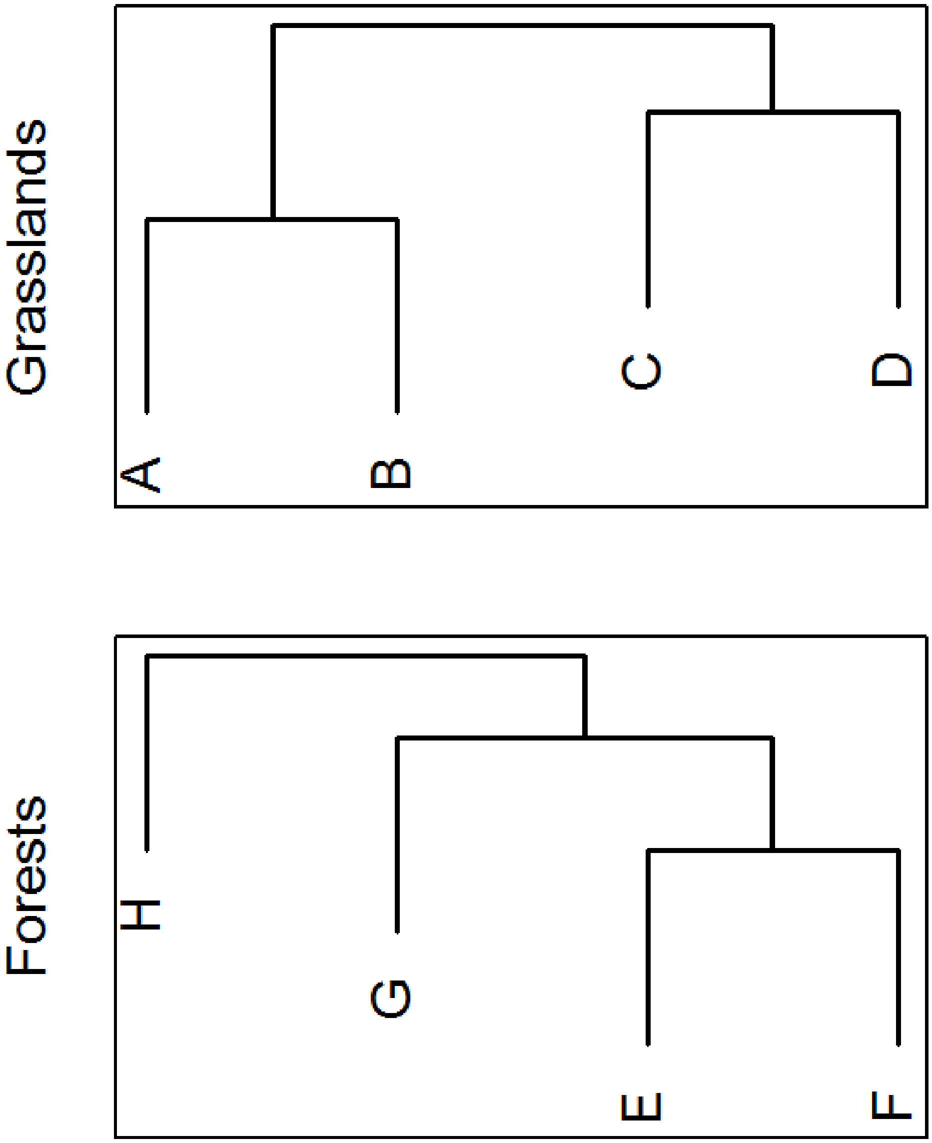

Figure 5. Cluster analysis of Bray-Curtis dissimilarity indices from 4 non-native ruderal grassland (A-D), and 4 upland deciduous woodland and forest (E-H) patches at Wilson's Creek National Battlefield, Missouri; July, 2016.

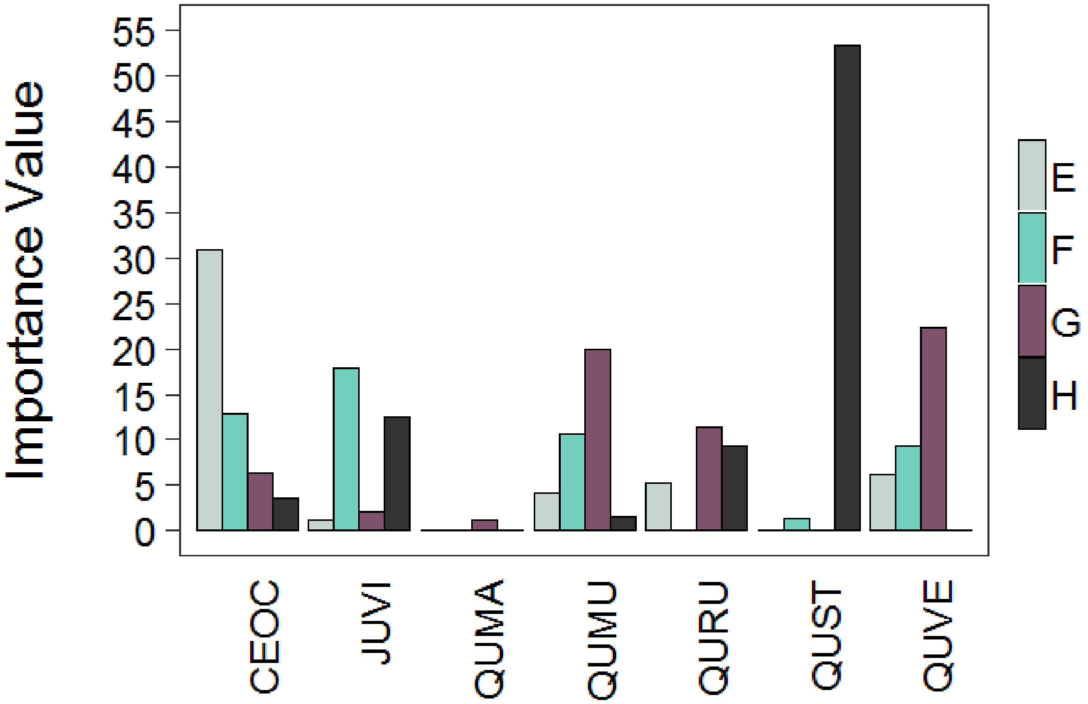

Figure 6. Relative importance values of tree species in four upland deciduous woodland and forest patches (E-H) at Wilson's Creek National Battlefield, Missouri; July, 2016. Tree species include hackberry (*Celtis occidentalis*; CEOC), Eastern red cedar (*Juniperus virginiana*; JUVI), blackjack oak (*Quercus marilandica*; QUMA), chinkapin oak (*Quercus, muehlenbergii*; QUMU), red oak (*Quercus rubra*; QURU), post oak (*Quercus stellata*; QUST), and black oak (*Quercus velutina;* QUVE).

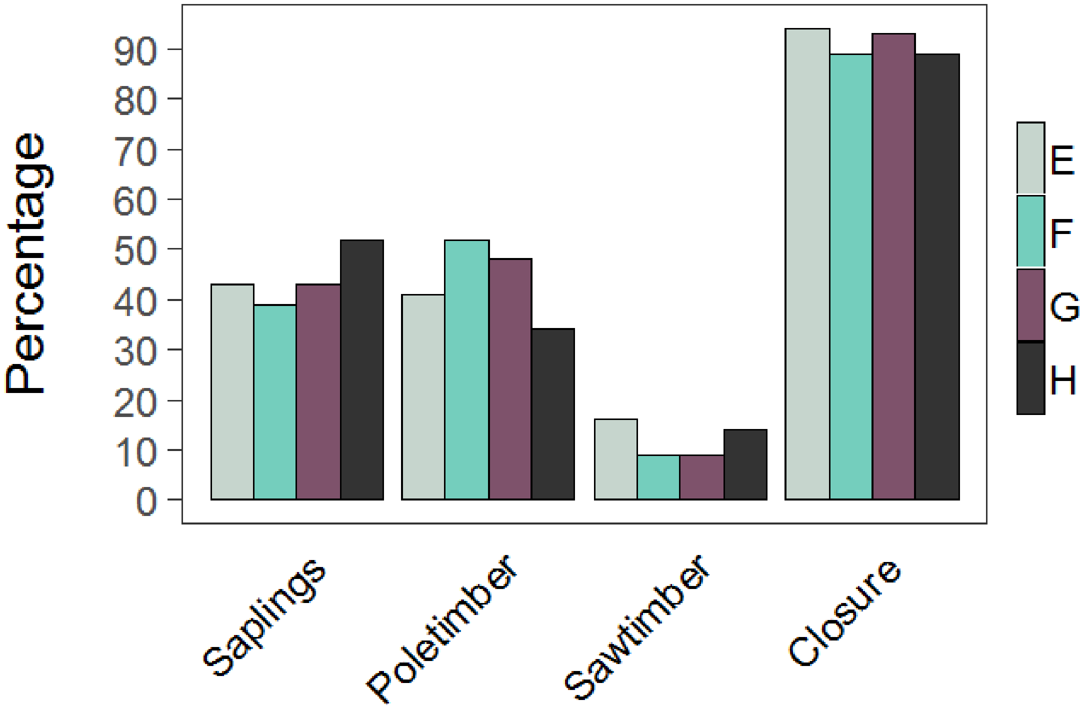

Figure 7. Tree cover and estimated canopy closure sampled along a centrally located 200m transect in four upland deciduous woodland and forest patches (E-H) at Wilson's Creek National Battlefield, Missouri; July, 2016.

CHAPTER 2: SMALL MAMMAL COMMUNITIES AT WILSON'S CREEK NATIONAL BATTLEFIELD

Introduction

Anthropogenic disturbance can cause fragmentation of a landscape, resulting in smaller remnant patches with varying degrees of isolation (Gaines, et al., 1997). Larger patches tend to have greater biodiversity because species with different ecological requirements can co-exist, so fragmentation is a recognized problem threatening biodiversity (Rybicki and Hanski, 2013). Wilson's Creek National Battlefield (WICR) is a mosaic landscape with high levels of fragmentation (Chapter 1; Annis et al., 2011). If suitable habitat patches become more isolated, extirpations are more likely, especially for larger animals that require more space (Kapos, et al., 2002) and species that are limited by dispersal because they cannot cross the surrounding matrix (Ferrari, et al., 2007; Hu, et al., 2012). While restoring the landscape to historic conditions and preserving natural vegetation has been a priority of park managers (National Park Service, 2003), it is also important to consider small mammal communities in order to reestablish ecological functions (DeGolier, et al., 2015).

Although certain changes in patches may be not be evident, small mammals can be good indicators of ecological shifts and habitat quality due to rapid population responses to disturbance (Goertz, 1964; Kalies, et al., 2012). Small mammals are not only good indicator species, but play key ecosystem roles in forest and grassland habitats (Logiudice et al., 2008; Pascarella and Gaines, 1991; Fantz and Renken, 2005; Stephens and Anderson, 2014). However, small mammal surveying was last conducted at WICR in 2004 (Annis et al., 2011) so more frequent sampling may be beneficial.

I sampled small mammals to provide WICR with a current estimate of community composition within the park. Specifically, I asked: 1) what are the driving forces of small mammal community patterns, and 2) how might we be able to predict small mammal occupancy in similar habitats? To answer these questions, I examined the relationships between habitat area and quality with small mammal habitat selection. I emphasized target vegetative patches (Chapter 1) to identify localized presence of small mammals. I fit detection models for species ubiquitously captured and fit occupancy models for species with enough encounter histories to allow for model convergence. I also examined community composition and abundance across sample patches. I performed statistical analyses in R (v. 3.3-2; R Core Team, 2016) and used the 'ggplot2' package (v.2.2-1; Wickham, H. 2009. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York. Available: http://ggplot2.org) to generate figures.

Methods

Study area. Wilson's Creek National Battlefield has undergone anthropogenic alterations including agricultural use and timber management (Diamond et al., 2013). Additionally, fire suppression has negatively influenced native communities by allowing non-native and invasive species to replace native vegetation (Annis et al., 2011). Today, habitats within the park are characterized by several non-native and invasive vegetation species (Chapter 1; Diamond et al., 2013), which may be unfavorable to certain small mammal species such as the Western harvest mouse (*Reithrodontomys megalotis* Baird), deer mouse (*Peromyscus maniculatus* Wagner), Elliot's short-tailed shrew *(Blarina hylophaga* Elliot), least shrew (*Cryptotis parva* Say), and the thirteen-lined ground squirrel (Matlack, et al., 2008), a species of concern in Missouri (Missouri Department of Conservation, 2017).

I selected 4 non-native ruderal grassland patches and 4 upland deciduous woodland patches because of their biological importance to small mammal species (Chapter 1; Fig. 2; Gillen and Hellgren, 2013; Nupp and Swihart, 1998; Reed et al., 2005; Schweiger et al., 2000). I considered each patch independent of each other because they were at least 500m apart, as measured in ArcMap (v.10.3; Lindstedt, et al., 1986; Pe'er et al., 2014). In each patch, I sampled a 200m transect with trap stations that consisted of one regular size (7.62×8.89×22.86 cm) and one large Sherman-live trap (10.16×11.43×38.10 cm) at 10m intervals. Traps were set in ideal locations; i.e. along woody debris or the base of trees, or along runways; traps were opened in the evening, and checked the following morning before dawn. Trapping was completed May–July 2016, and each patch was trapped 3 times (primary events) for 4 successive days (secondary events), totaling 12 trap nights per patch.

Small mammals. Upon initial capture, I marked individual rodents with a unique ear tag (Model 1001-A, National Band and Tag) and clipped fur in distinct patterns for shrews. I recorded the species, sex, pelage color, reproductive status, mass, ear length, tail length, and hind foot length for each animal. I combed individuals for two minutes to collect ectoparasites, and released the mammals at the capture site. Vouchered mammals were prepared as museum specimens and deposited as property of the National Park Service at Wilson's Creek National Battlefield. I stored ectoparasites in ethanol, and identified them at Missouri State University. All trapping and handling procedures were approved by the Institutional Animal Care and Use Committee at Missouri State University (IACUC ID: 15-026.0) and followed the guidelines of the American Society of Mammologists (Sikes et al. 2016).

Statistical analyses. For small mammal species with at least 4 captures, I modeled either occupancy and detection or detection alone (MacKenzie et al., 2006). I included Julian date as a detection covariate, as it might be associated with within-season variations such as increasing resource availability (Kalies et al., 2012) or population size (Moore and Swihart, 2005). I included primary and secondary sampling periods [i.e. trapping event (1–3), and successive trap day (1–4) within each trapping event] to account for heterogeneous detectability among individuals (trap happy or trap shy). Landscape covariates included average, minimum, and maximum temperature (LaZerte and Kramer, 2016), nightly sky illumination (Carley, et al., 1970), and precipitation within 24 hours of sampling (Thompson, et al., 2010). I obtained moonlight data, recorded as the proportion of the moon illuminated at midnight, from the U.S. Naval Observatory (http://aa.usno.navy.mil). Moonlight data was omitted from chipmunk analyses, as they are mainly diurnal animals (Snyder, 1982). I downloaded daily temperature and precipitation estimates coinciding with my sampling periods for Greene and Christian Counties from the PRISM climate website (PRISM Climate Group, 2004. Oregon State University Climate Data. Available: http://prism.oregonstate.edu). Because WICR crosses the border of these counties, I used the mean value of each weather variable per day.

For species with occurrence in all respective patches (i.e. all forest or all grassland patches), I fit logistic regression models to describe variation in detectability. I assumed survey histories for each species were a sequence of independent Bernoulli trials and generated presence/absence matrices, with detections (i.e. ‘successes’) denoted with a one, and non-detection (i.e. ‘failures’) with a zero. I constructed encounter histories for small mammals at each trap station. I used the detection covariates Julian date, trap day, precipitation, temperature, sky illumination, and trap station as a factor. Models were fit with the binomial ‘glm’ function with a logit-link in the ‘stats’ package. I used Hosmer Lemeshow goodness-of-fit tests for models with significant coefficients to compare the observed data with the predicted probabilities using the ‘hoslem.test’ function in the ‘ResourceSelection’ package (v.0.3-2; Subhash, et al., 2017). I compared AIC values and goodness-of-fit outputs to identify the best predictors of detection.

For species not detected in each of the respective patches and with a minimum number of detections, I modeled occupancy to address which covariates may be influencing habitat selection. I used detection covariates as previously noted and occupancy covariates included patch size (Vieira et al., 2009), herbaceous vegetation height (Washburn and Seamans, 2007), and proportion of bare ground (DeGolier et al., 2015; Kaminski, et al., 2007) for both grassland and forest patches. For forest patches, occupancy covariates also included the number and decay class of fallen logs, the number of rocks (Kaminski et al., 2007), saplings, pole, and sawtimber trees, and canopy closure. I also included the estimated tree density, tree cover, tree diversity, and the calculated importance values for oak species (*Quercus* spp.), hackberry (*Celtis occidentalis* L.), and Eastern red cedar (*Juniperus virginiana* L.).

As there were varying degrees of magnitude in the variables, they were standardized to a comparable scale by using the z-score function ‘scale’ in the ‘base’ package. This transformation also is expected to weight models appropriately and improve convergence (Fiske and Chandler, 2011; Kéry and Chandler, 2012). To avoid multicollinearity, I generated a Pearson’s correlation matrix of covariates with the ‘cor’ function in package ‘stats’ (v.3.3-2) and removed one of two highly correlated variables ($r \geq 0.70$) before model fitting that were of lesser biological importance, or the least standardized measure (Stephens and Anderson, 2014). I summarized encounter histories at the patch-level, rather than at individual trap locations as done in the detection-only species. I fit single-season occupancy models using the function ‘occu’ in the ‘unmarked’ package (v.0.11-0; Fiske and Chandler, 2011), which implements the zero-inflated binomial formula described by MacKenzie et al (2002).

I initially estimated detection probabilities by fitting models with either constant detection, as singular variables, or as covariate combinations, with only an intercept for occupancy. I avoided over-parameterizing my models by first fitting and assessing univariate detection covariates, and then fitting combinations of significant variables. I used the best-supported detection covariate in occupancy modeling, and repeated the same process for model fitting. I fit models to the best of my ability by adjusting the optimization method or providing starting values as necessary. I attempted to use any equally meaningful previously omitted variable in place of an originally selected variable if Markov-Chain-Monte-Carlo (MCMC) methods did not converge or resulted in extremely wide standard errors. I compared candidate models using Akaike's Information

Criterion adjusted for small sample size (AIC_C). I chose the preferred model by assessing AIC_C weights, and the standard errors of the slope estimates.

I fit closed population abundance models in each habitat patch for species with enough captures at patch-level to produce reliable estimates. I fit log-linear abundance models with ‘closed.p’ function, or the ‘closedp.bc’ function in the ‘Rcapture’ package (v.1.4-2; Rivest and Baillargeon, 2015). Both of these models account for heterogeneous individual behaviors, allowing for a more relaxed assumption of random sampling. The ‘closedp.bc’ applies a bias correction by modifying the capture frequencies in a Poisson regression to improve abundance estimations and variances when capture probabilities are small (Rivest and Lévesque, 2001).

I used the raw counts of unique individuals of each small mammal species to estimate richness and diversity in the each of the grassland and forest patches. To estimate species richness I used the ‘estimateR’ function in the ‘vegan’ package v. 2.4-3; Oksanen et al., 2017) to find corrected Chao and ACE metrics, which account for potential undetected species based on singletons and doubletons, and rare species. To summarize community structure, I calculated Simpson’s diversity index for each patch with the ‘diversity’ function in the ‘vegan’ package. I created rarefaction curves to compare richness among patches at a minimum sample size by using the ‘ggiNEXT’ function in the ‘iNEXT’ package (v. 2.0.14; Hseih, et al., 2016). I calculated Bray-Curtis dissimilarity indices between patches of a given type to assess differences in small mammal species composition.

Results

I captured 129 unique small mammal individuals representing 8 species over the course of 3,840 trap nights between May and July 2016 (Tab. 5). The hispid cotton rat (*Sigmodon hispidus* Say) was ubiquitous in all of the grassland patches, and the most commonly caught species in 3 of the 4 patches (Fig. 8). The prairie vole (*Microtus ochrogaster* Wagner) was the most common species in grassland patch C, but was missing from patch D (Fig. 8). The Western harvest mouse was also frequently captured (Fig. 8). In the grassland patches, species rarely encountered were Elliot's short-tailed shrew (2 captures), the deer mouse (1 capture), and the least shrew (1 capture; Fig. 8). The white-footed mouse (*Peromyscus leucopus* Rafinesque) occupied all of the forest patches, and was the most commonly caught species across all forest patches (Fig. 9). The deer mouse was captured to a lesser extent in patches E and F, but not found in G or H, and the Eastern chipmunk (*Tamias striatus* L.) was captured in 3 of 4 forest patches, although not frequently (Fig. 9). In forest patches, rarely encountered species were prairie vole, Elliot's short-tailed shrew, and the least shrew, each captured once (Fig. 9).

Detection models. I fit detection models for the hispid cotton rat and white-footed mouse for the grassland and forest patches, respectively. The only significant variable influencing the probability of cotton rat detectability was Julian date ($Z = 2.24$; $P = 0.02$; Tab. 6), but the Hosmer-Lemeshow goodness of fit test indicated the model was not a good fit ($P(\chi^2) < 0.05$; Tab. 6). Increased moon illumination had a significant influence on the white-footed mouse detectability ($Z = 5.6$; $P = 0.33$; Tab. 6), and Hosmer-Lemeshow goodness of fit tests indicated the model was a good fit ($P(\chi^2) = 0.33$; Tab. 6). Average daily temperature had a significant influence ($Z = -6.3$; $P < 0.05$) on the white-footed mouse detectability, but the Hosmer-Lemeshow goodness of fit indicated the model was not a good fit ($P(\chi^2) < 0.05$; Tab. 6).

Occupancy models. I fit occupancy models for the Western harvest mouse and prairie vole for the grassland patches. I fit occupancy models for the Eastern chipmunk, and deer mouse in the forest patches. Although Elliot's short-tailed shrew and the least shrew were captured in both habitats, sample sizes were too small to run occupancy or abundance models for either species.

The model with constant detection and occupancy had the lowest AIC_C value for the Western harvest mouse; however, the MCMC approaches failed to converge well when adding occupancy covariates, so this model was omitted. Additionally, MCMC approaches failed to converge for detection and occupancy models when multiple covariates were included, regardless if individual predictors were able to converge or not, so only singular predictors were used. Precipitation within 24 hours of trap night was the best-supported detection model that would allow for occupancy model convergence, and moon illumination was the next ranked detection model (Tab. 7). Both covariates had a positive relationship with Western harvest mouse detection, although neither of the beta slope estimates were significant ($\beta_{PRECIP} = 0.5$, $Z = 1.3$, $P = 0.2$; $\beta_{MOON} = 0.3$, $Z = 0.7$ $P = 0.5$; Tab. 7). The best-supported occupancy model included vegetation richness ($\beta = 0.6$, $Z = 0.5$, $P = 0.57$) and the second ranked model included patch area ($\beta = -1.0$, $Z = 0.6$, $P = 0.54$; Tab. 7).

Precipitation was the strongest predictor of detectability for the prairie vole, showing a significant and positive influence ($\beta = 1.3$, $Z = 2.0$, $P = 0.04$; Tab. 8). The detection-only model with moon illumination was ranked second, but the coefficient was not significant and the negative slope estimate approached zero ($\beta = -0.5$, $Z = -0.9$, $P = 0.34$; Tab. 8). Vegetation height was included in the preferred occupancy model and vegetation richness was included the next ranked model, but both coefficients were not significant ($\beta_{HEIGHT} = -3.6$, $Z = -0.9$, $P = 0.37$; $\beta_{RICH} = -9.9$, $Z = 0.5$, $P = 0.61$; Tab. 8). The MCMC approaches were able to converge for occupancy models with additional patch-level covariates, but resulted in wide standard errors that led to uncertain estimates.

I fit multivariate detection models for the Eastern chipmunk, but Julian date as a single predictor was the best-supported model. However, the relationship between Julian date and detection was not significant ($\beta = -0.9$, $Z = 0.9$, $P = 0.34$; Tab. 9). The second ranked detection-only model included Julian date and temperature but neither coefficient was significant ($\beta_{JULIAN} = -1.0$, $Z = -1.9$, $P = 0.34$; $\beta_{TEMP} = 0.3$, $Z = 0.5$, $P = 0.58$; Tab. 9). The top-ranked occupancy models were fit with singular predictors. The number of fallen logs was the preferred model and oak importance values was the second-best model. Both the number of fallen logs and oak importance values had positive coefficients but were not significant. ($\beta_{DWM} = 7.4$; $Z = 11$, $P = 0.38$; $\beta_{OAK} = 7.9$; $Z = 5.2$, $P = 0.38$; Tab. 9).

For the deer mouse, the top ranked detection-only model included precipitation as a covariate, although the coefficient was not significant ($\beta = -1.0$, $Z = -1.8$, $P = 0.06$; Tab. 10). The next ranked model included both precipitation and temperature as covariates. Both coefficients were negative, but neither was significant ($\beta_{PRECIP} = -1.0$, $Z = -1.8$, $P = 0.08$; $\beta_{TEMP} = -0.9$, $Z = -1.2$, $P = 0.24$; Tab. 10). The preferred occupancy model included tree density and the second ranked model included Eastern red cedar importance values, but both positive coefficients were not significant ($\beta_{TDEN} = 1.33$, $Z = 0.84$, $P = 0.40$; β_{ERC}

= 0.36, $Z = 0.30$, $P = 0.76$), while Eastern red cedar had relatively wide standard errors (Tab. 10).

Abundance models. The grassland species with enough captures to fit abundance models were the prairie vole, hispid cotton rat, and Western harvest mouse, while the forest species with enough captures were the white-footed mouse, deer mouse, and eastern chipmunk. The prairie vole was only captured enough in patch C to estimate abundance, and the model had wide standard errors (Tab. 5). The hispid cotton rat and the Western harvest mouse had the highest abundance in patch A, but the abundance estimates for the Western harvest mouse strongly differed than the observed data, and resulted in wide standard errors for individual patches and as a total estimate (Tab. 5). In the forests, patch E had the highest abundance estimates for all species, with the white-footed mouse having the highest observed and estimated abundance (Tab. 5). Patch G had the lowest abundance estimate of white-footed mice, while the deer mouse was not captured in patches G and H, and the eastern chipmunk was only captured enough times to provide abundance estimates in patch E (Tab. 5).

Community composition. Observed small mammal richness ranged from 1–6 species for both the grassland and forest patches. Estimated richness in the grasslands ranged from 1–21, and estimated richness ranged from 1–10 in the forests (Tab. 11). Patch A had the highest estimated grassland richness, while patch D had the lowest estimated richness (Tab. 11). In the forests, patch E had the highest richness values, while patch H had the lowest values (Tab. 11). ACE richness estimators resulted in higher values than Chao estimates, but both had narrow standard errors associated in all patches (Tab. 11), and showed similar trends as the observed richness as a function of patch size in the grasslands (Fig. 10), and the forests (Fig. 11). Similarly, small patches had higher richness than large patches, but the highest values fell into the mid-range of sampled patch sizes in the grasslands (Fig. 10), and forests (Fig. 11).

There were no differences in the rarefied richness values at a minimum sample size of 7 captured individuals in grassland patches A, B, and C (Fig. 12). However, the richness values in grassland D did not overlap with the 95% confidence intervals from any other patch, regardless of the sample size (Fig. 12). In the smallest subsample of 9 forest individuals, there are no differences in rarefied richness between patches E, F, and G, but there was low estimated richness in patch H (Fig. 13). However, as the sample size increases, the forest patches begin to fall out of the 95% confidence intervals of each other, and species richness becomes increasingly dissimilar (Fig. 13).

Simpson's diversity index values were more variable among grassland patches than forest patches (Tab. 11). Index values for grasslands were similar among patches A–C, but not in patch D; patch A had the highest diversity, while patch D had the lowest (Tab. 11). Among forest patches, diversity was highest at patch E, and lowest in patch H (Tab. 11). Small mammal community composition showed a nested pattern in the grasslands, with patch C being the most dissimilar, and patches B and D showing the most similar composition (Fig. 14). Among forest patches, community composition had a split pattern, with patches E and F similar, and patches G and H similar (Fig. 14). Overall, communities in grasslands differed from those in the forest (Fig. 14).

The most commonly collected tick was the black-legged tick (*Ixodes scapularis*), with the highest observed abundance in patch E (Tab. 12). The lone-star tick (*Amblyomma americanum*) was the second most common species collected, albeit it was much rarer than the black-legged tick. The American dog tick (*Dermacentor variabilis*) was collected only from forest patch F (Tab. 12). Patch E had greatest number of total ticks collected, and no ticks were combed from mammals in patch G (Tab. 12).

Discussion

Detection models. The model including Julian date was the best-supported model of cotton rat detection, but the model did not fit well based on Hosmer-Lemeshow goodness-of-fit tests. In Northeast Kansas, cotton rats have a high capture probability during the summer months, but it declines in the fall (Slade and Swihart, 1983). It is likely the model was a poor fit because cotton rats reach peak density and activity levels during the summer breeding season (Foster and Gaines, 1991), so capture frequencies did not differ enough during the short trapping period. Cotton rats were captured in all sample patches within WICR, as the disturbed nature of the grasslands potentially offer preferable habitat (Goertz, 1964).

Moon illumination had a positive effect on white-footed mouse detection, and the model was a good fit. The strength and the direction of the effect is surprising, as rodent activity often is decreased during bright nights to avoid predator detection (Fitzgerald and Wolff, 1988). White-footed mice had relatively equal detectability regardless of temperature, although studies have shown reduced *Peromyscus* activity when temperatures are not within an intermediate range of about 10°–20° Celsius (Marten, 1973), temperatures that occurred throughout the study period at WICR. Because white-footed mice are forest generalists (Anderson et al., 2006) and one of the most common

Eastern deciduous forest mammals (Wolf and Batzli, 2004), I expected them to occupy most, or all of the sampled forest patches.

Occupancy models. Although the best-supported model containing precipitation did not have a significant slope estimate for Western harvest mice, they have been shown to be more active during rainy nights (Webster and Jones Jr, 1982). Moon illumination had a positive effect on detectability, which was not expected (Carley et al., 1970; Webster and Jones Jr, 1982). The negative effect of patch size is in agreement with existing literature (Foster and Gaines, 1991), although Western harvest mice do tend to have higher persistence rates on larger patches (Foster and Gaines, 1991). Although listed as uncommon in the WICR database (https://irma.nps.gov), with the high spatial integrity indices of the grasslands (Chapter 1) and the dominance of cool-season grasses (Skupski, 1995; Webster and Jones Jr, 1982), it is more likely that Western harvest mouse could become common and abundant within the park.

The prairie vole is identified as one of the more common species at WICR (https://irma.nps.gov), but during my sampling I did not capture a large number of them. Prairie voles had higher detection on rainy nights. Increased activity on dark and rainy nights may be a tactic to avoid nocturnal predators during periods of required foraging (Carley et al., 1970; Vickery and Bider, 1981). I did not find a significant relationship with vegetation height or vegetative richness and occupancy, which may be due to the similar covariate values among patches (Chapter 1) and a small sample size. Positive relationships would be expected as vegetation cover provides protection from predation and abiotic conditions, allowing for successful movement, nesting, and overall persistence (Birney, et al., 1976; Smith and Batzli, 2006; Solomon, et al., 2005). Prairie vole populations are negatively associated with the presence and abundance of cotton rats (Frydendall, 1969), especially during summer (Glass and Slade, 1980a, 1980b) and in smaller areas where space is a limiting factor (Stalling, 1990). Where I found the largest number of prairie voles, there also was the lowest number cotton rats. However, my dataset is insufficient to test for a negative association at WICR.

The preferred detection-covariate for eastern chipmunks was Julian date, although the coefficient was not significantly different than 0. Downed woody material was included in the best-supported occupancy model, although the estimate of the coefficient had a large standard error. However, chipmunks do prefer habitats with fallen logs (Snyder, 1982) because they provide several functional roles as crevices, physical covers, runways, locational markers, and foraging sites (Zollner and Crane, 2003). Oak importance values were included in the second ranked model, again without a statistically significant coefficient. Eastern chipmunks can reach their highest densities where trees such as oak are present (Wolff, 1996), although this was not the case at WICR.

Detection of the deer mouse was most likely linked with precipitation patterns, as it was included in both of the top two ranked models for detection, albeit with a coefficient not different from 0. Getting wet increases the rate of heat loss (Vickery and Bider, 1981), which may explain why deer mice were more active on dry evenings, especially on cooler nights. The deer mouse subspecies that occurs in most of Missouri (*Peromysucs maniculatus bairdii*) tends to occur in grassland habitats rather than forests (Schwartz and Schwartz, 2016) whereas *P. maniculatus ozarkiarum* occurs in southwestern Missouri and was noted from grassy areas within a forest matrix (Brown 1963). Grasslands at WICR are characterized by dense ground cover (Chapter 1), which

P. maniculatus bairdii tends to avoid (Clark and Kaufman, 1991), but similar observations on *P. maniculatus ozarkiarum* are lacking. The exact relationships of deer mouse occupancy is unclear because coefficients for tree densities and Eastern red cedar importance were not different from 0. However, these covariates may play a role in predator avoidance on bright nights (Clark, 1983) in the forest patches.

Community composition. Mid-size patches had the highest observed richness in both grasslands and forest patches. Captures of both edge (i.e. chipmunk and white-footed mouse; Mahan and Yahner, 1998; Wolf and Batzli, 2004) and interior (i.e. deer mice and Western harvest mice; Bock et al., 2002) species may have led to higher richness and diversity in the mid-size patches (Press et al., 2013). However, rarefaction analysis indicates no real difference in richness among patches within habitat types. Both Chao and ACE richness estimators suggested incomplete sampling of the communities, however. High values of the estimators are due to few captures of rare species which could inflate the estimates. Increased sampling effort could yield more reasonable estimates for patches A, C, and E.

Communities differed between grassland and forest patches, although there was overlap in small mammal community composition, as there were 4 shared species. Patch differences in grasslands were driven by multiple captures of the prairie vole in patch C and few if any in the remaining patches in which cotton rat captures were high. Differences between A, B, and D can be characterized by a less diverse assemblage from in B and D relative to A. In forest patches, the main difference is the presence of deer mouse in patches E and F, and its absence in G and H.

Improvements to study design. Additional survey effort in terms of the number of patches, days, and traps, along with trap placement would lead to more confident community composition estimates and models that are more robust. Trapping during the spring and autumn months of peak activity may allow increased captures of Western harvest mice (Carley et al., 1970; Skupski, 1995; Webster and Jones Jr, 1982; Johnson and Gaines, 1988). Additionally, use of elevated traps (Cummins and Slade, 2007; Houser and Zollner, 2013) would increase capture success for Western harvest mice. Chipmunk capture success should increase by trapping during the daylight hours, as they are primarily diurnal (LaZerte and Kramer, 2016; Snyder, 1982). A long-term dataset would allow for examination of changes in occupancy due cyclical population dynamics or masting events that might be overlooked in a single season model (LaZerte and Kramer, 2016). Likewise, chipmunks respond positively to the presence of white oaks more than red oaks (Pyare, et al., 1993), so monitoring oak species might provide insight into chipmunk occupancy and abundance. Further investigation into the role of vegetative composition and interspecific interactions among species, particularly the prairie vole and the cotton rat, may show clearer trends and lead to a better understanding of small mammal occupancy at Wilson's Creek National Battlefield.

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Table 5. Unique small mammal individuals captured and (*abundance estimates ± standard errors*) at Wilson’s Creek National Battlefield, Missouri, May–July, 2016. Patches A–D are non-native ruderal grasslands, and patches E–H are upland deciduous woodland and forests. Small mammal species include the prairie vole (*Microtus ochrogaster*; MIOC), Elliot’s short-tailed shrew (*Blarina hylophaga*; BLHY), least shrew (*Cryptotis parva*; CRPA), deer mouse (*Peromyscus maniculatus*; PEMA), Western harvest mouse (*Reithrodontomys megalotis*; REME), hispid cotton rat (*Sigmodon hispidus*; SIHI), white-footed mouse (*Peromyscus leucopus*; PELE), and Eastern chipmunk (*Tamias striatus*; TAST).

5a. Non-native ruderal grassland patches A – D.

Species	A	B	C	D	Total
MIOC	1	1	5 (*17 ± 15*)	0	7 (*17 ± 15*)
BLHY	1	0	1	0	2
CRPA	1	0	0	0	1
PEMA	1	0	0	0	1
REME	7 (*30 ± 28*)	4 (*12 ± 10*)	1	0	12 (*68 ± 63)*
SIHI	14 (*29 ± 8*)	5 (*8 ± 3*)	1	7 (*17 ± 10*)	27 56 ± *13*)

5b. Upland deciduous woodland and forest patches E – H.

Species	E	F	G	H	Total
MIOC	1	0	0	0	1
BLHY	1	0	0	0	1
CRPA	1	0	0	0	1
PEMA	4 (*5 ± 0.1*)	3 (4 ± *1)*	0	0	7 (7 ± *1*)
PELE	26 (*35 ± 3*)	16 (*20 ± 2)*	10 (*13 ± 3)*	9 (*16 ± 6)*	61 (*81 ± 5*)
TAST	6 (*23 ± 21)*	1	1	0	8 (*23 ± 21)*

Table 6. Slope estimates (β) and standard errors ($\sigma\overline{x}$) of the best-fit detection models for the hispid cotton rat (*Sigmodon hispidus*) and the white-footed mouse (*Peromyscus leucopus*) at Wilson's Creek National Battlefield, Missouri; May–June, 2016.

	Covariate	$\hat{\beta}$	$\sigma_{\overline{x}}$	Z	p(>|z|)	χ^2	$P(\chi^2)$
Sigmodon hispidus							
	Julian date	0.02	0.01	2.20	0.02	43	< 0.05
Peromyscus leucopus							
	Illumination	1.75	0.31	5.6	< 0.05	9.2	0.33
	Temperature	-0.09	0.02	-6.3	< 0.05	33	< 0.05

Table 7. Detection (p), and occupancy (Ψ) model slope (β) estimates from best-fit models for the Western harvest mouse (*Reithrodontomys megalotis*) at Wilson's Creek National Battlefield, Missouri; May–June, 2016. Best-fit models were selected based on the Akaike's information criterion corrected for small sample size (AIC_C), and standard errors ($\sigma\overline{x}$). Detection (p) and occupancy (Ψ) covariates include precipitation within 24 hours of trap night (Rain), moon phase (Illumination), vegetation richness (Richness), patch size (Area), and constant detection or occupancy (.).

	Covariate	$\hat{\beta}$	$\sigma_{\overline{x}}$	Z	p(>|z|)	ΔAIC_C	AIC_{WT}	K
Detection								
	p(Rain) Ψ(.)	0.5	0.41	1.3	0.2	0.00	0.24	3
	p(Illumination) Ψ(.)	0.3	0.48	0.7	0.5	1.15	0.13	3
Occupancy								
	p(Rain) Ψ(Richness)	0.6	0.41	0.5	0.57	0.00	0.71	4
	p(Rain) Ψ(Area)	-1.0	0.39	0.6	0.54	1.81	0.29	4

Table 8. Detection (p), and occupancy (Ψ) model slope (β) estimates from best-fit models for the prairie vole (*Microtus ochrogaster*) at Wilson’s Creek National Battlefield, Missouri; May–June, 2016. Best-fit models were selected based on the Akaike's information criterion corrected for small sample size (AIC_C), and standard errors ($\sigma\bar{x}$). Detection (p) and occupancy (Ψ) covariates include precipitation within 24 hours of trap night, vegetation height and richness, and constant detection or occupancy (.).

	Covariate	$\hat{\beta}$	$\sigma_{\bar{x}}$	Z	p(>|z|)	ΔAIC_C	AIC_{WT}	K
Detection								
	p(Rain) Ψ(.)	1.3	0.68	2.0	0.04	0.00	0.66	3
	p(Illumination) Ψ(.)	-0.5	0.57	-0.9	0.34	4.07	0.09	2
Occupancy								
	p(Rain) Ψ(V. Height)	-3.6	4.0	-0.9	0.37	0.00	0.33	4
	p(Rain) Ψ(Richness)	-9.9	19	-0.5	0.61	0.25	0.30	4

Table 9. Detection (p), and occupancy (Ψ) model slope (β) estimates from best-fit models for the Eastern chipmunk (*Tamias striatus*) at Wilson's Creek National Battlefield, Missouri; May–June, 2016. Best-fit models were selected based on the Akaike's information criterion corrected for small sample size (AIC_C), and standard errors ($\sigma_{\overline{x}}$). Detection (p) and occupancy (Ψ) covariates include Julian date (Julian), average daily temperature (Temp), number of fallen logs (DWM), patch-level oak importance values (Oak), and constant detection or occupancy (.).

	Covariate	$\hat{\beta}$	$\sigma_{\overline{x}}$	Z	p(>|z|)	ΔAIC_C	AIC_{WT}	K
Detection								
	p(Julian) Ψ(.)	-0.9	0.4	0.9	0.34	0.00	0.45	3
	p(Julian+Temp) Ψ(.)	-1.0 0.3	0.5 0.6	-1.9 0.5	0.06 0.58	1.70	0.19	4
Occupancy								
	p(Julian) Ψ(DWM)	7.4	9.4	11	0.38	0.00	0.41	4
	p(Julian) Ψ(Oak)	7.9	-4.5	5.2	0.38	0.61	0.30	4

Table 10. Detection (p), and occupancy (Ψ) model slope (β) estimates from best-fit models for the deer mouse (*Peromyscus maniculatus*) at Wilson's Creek National Battlefield, Missouri; May–June, 2016. Best-fit models were selected based on the Akaike's information criterion corrected for small sample size (AIC_C), and standard errors ($\sigma_{\overline{x}}$). Detection (p) and occupancy (Ψ) covariates include precipitation within 24 hours of trapping (Rain), average daily temperature, tree density (TDEN), patch-level Eastern red cedar importance values (JUVI), and constant occupancy (.).

	Covariate	$\hat{\beta}$	$\sigma_{\overline{x}}$	Z	p(>|z|)	ΔAIC_C	AIC_{WT}	K
Detection								
	p(Rain) Ψ(.)	-1.0	0.54	-1.8	0.06	0.00	0.25	3
	p(Rain+Temp) Ψ(.)	-1.0 -0.9	0.57 0.77	-1.8 -1.2	0.08 0.24	1.70	0.19	4
Occupancy								
	p(Rain) Ψ(TDEN)	1.33	1.59	0.84	0.40	0.00	0.11	4
	p(Rain) Ψ(JUVI)	0.36	1.17	0.30	0.76	0.89	0.07	4

Table 11. Observed and estimated ACE and Chao small mammal richness (± *standard error*), and Simpson's diversity indices at four non-native ruderal grassland (A–D) and four upland deciduous woodland and forest (E–H) sampled patches at Wilson's Creek National Battlefield, Missouri; May–June, 2016.

Metric	A	B	C	D	E	F	G	H
Observed Richness	6	3	4	1	6	3	2	1
ACE Richness	21 *(± 2)*	4 *(± 1)*	13 *(± 2)*	1 *(± 0)*	10 *(± 1)*	4 *(± 1)*	3 *(± 1)*	1 *(± 0)*
Chao Richness	12 *(± 7)*	3 *(± 0)*	7 *(± 4)*	1 *(± 0)*	9 *(± 4)*	3 *(± 0)*	2 *(± 0)*	1 *(± 0)*
Simpson's Diversity	0.60	0.58	0.56	0.00	0.52	0.34	0.17	0.00

Table 12. Tick abundance collected from small mammals in four non-native ruderal grassland patches (A–D), and four upland deciduous woodland and forest patches (E–H) at Wilson's Creek National Battlefield, Missouri; May–June, 2016.

Species	A	B	C	D	E	F	G	H
Black-legged tick (*Ixodes scapularis*)	7	1	4	2	14	3	0	3
Lone-star tick (*Amblyomma americanum*)	3	0	1	0	4	0	0	0
American dog tick (*Dermacentor variabilis*)	0	0	0	0	0	2	0	0

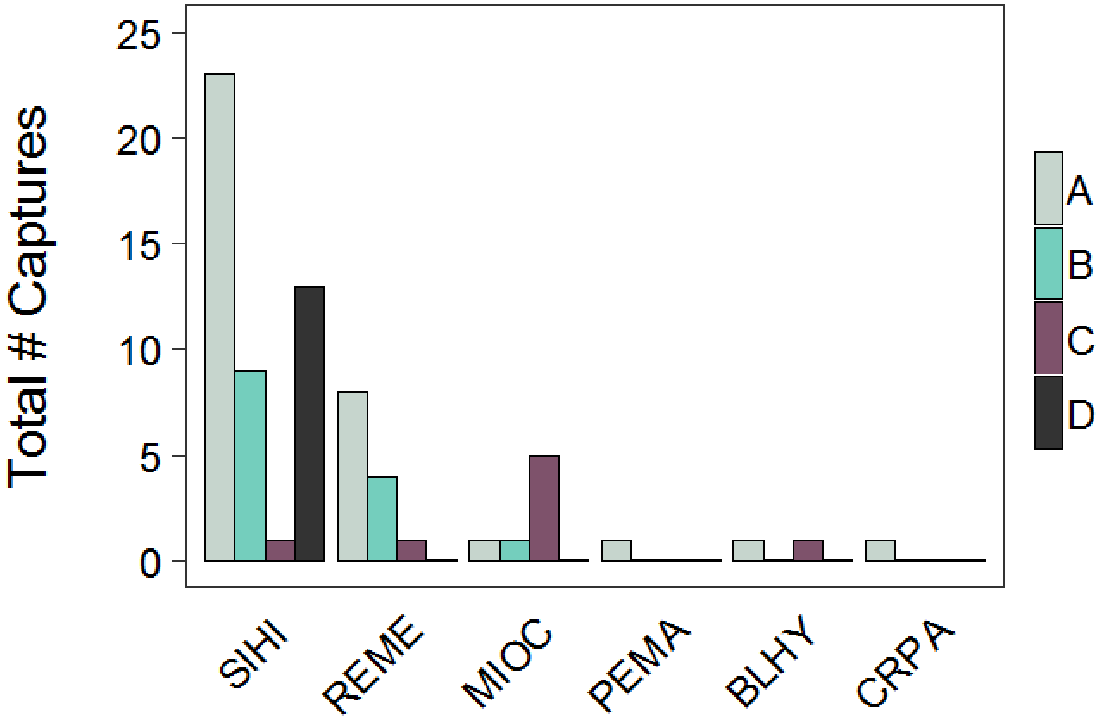

Figure 8. Small mammal capture events at four non-native ruderal grassland patches (A–D) at Wilson’s Creek National Battlefield, Missouri, May–July, 2016. Small mammal species include hispid cotton rat (*Sigmodon hispidus*; SIHI), Western harvest mouse (*Reithrodontomys megalotis*; REME), prairie vole (*Microtus ochrogaster*; MIOC), deer mouse (Peromyscus maniculatus; PEMA), Elliot’s short-tailed shrew (*Blarina hylophaga*; BLHY), and least shrew (*Cryptotis parva*; CRPA).

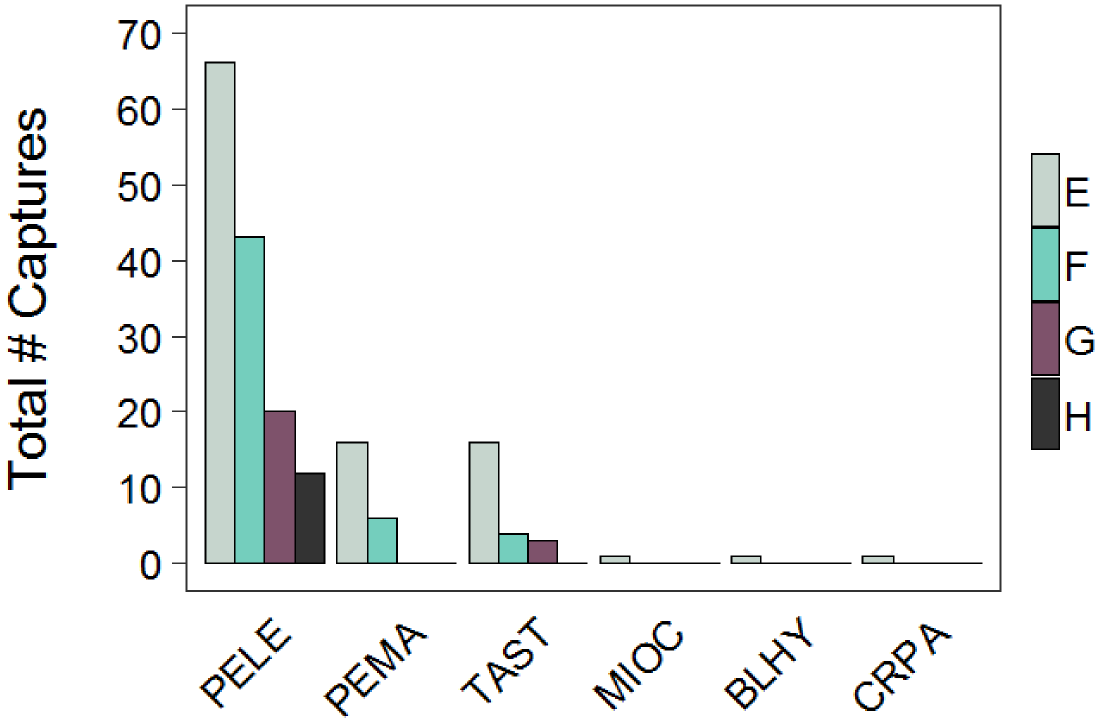

Figure 9. Small mammal capture events at four upland deciduous woodland and forest patches (E–H) at Wilson’s Creek National Battlefield, Missouri, May–July, 2016. Small mammal species include white-footed mouse (*Peromyscus leucopus*; PELE), deer mouse (*Peromyscus maniculatus*; PEMA), Eastern chipmunk (*Tamias striatus*; TAST), prairie vole (Microtus ochrogaster; MIOC), Elliot’s short-tailed shrew (*Blarina hylophaga*; BLHY), and least shrew (*Cryptotis parva*; CRPA).

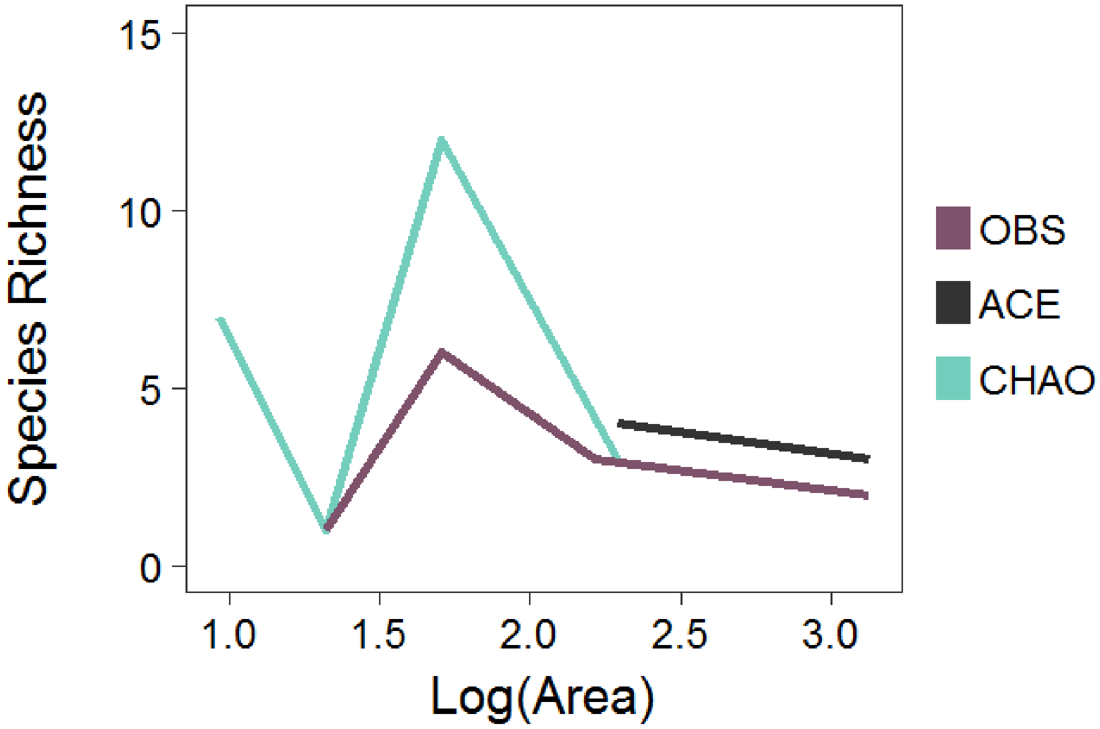

Figure 10. Observed (OBS), ACE, and Chao estimated richness of small mammal species as a function of patch size in four non-native ruderal grassland patches (A-D) at Wilson's Creek National Battlefield, Missouri; May–July Wilson's Creek National Battlefield, Missouri, May–July, 2016.

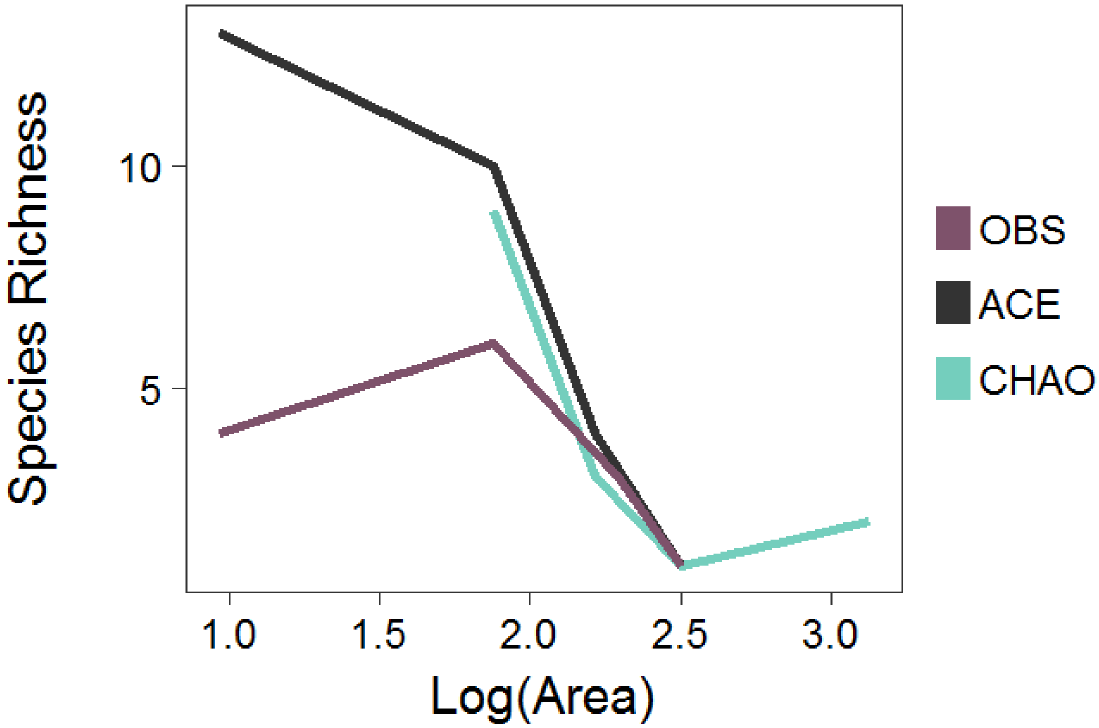

Figure 11. Observed (OBS), ACE, and Chao estimated richness of small mammal species as a function of patch size in four upland deciduous woodland and forest patches (E-H) at Wilson’s Creek National Battlefield, Missouri; May–July Wilson’s Creek National Battlefield, Missouri, May–July, 2016.

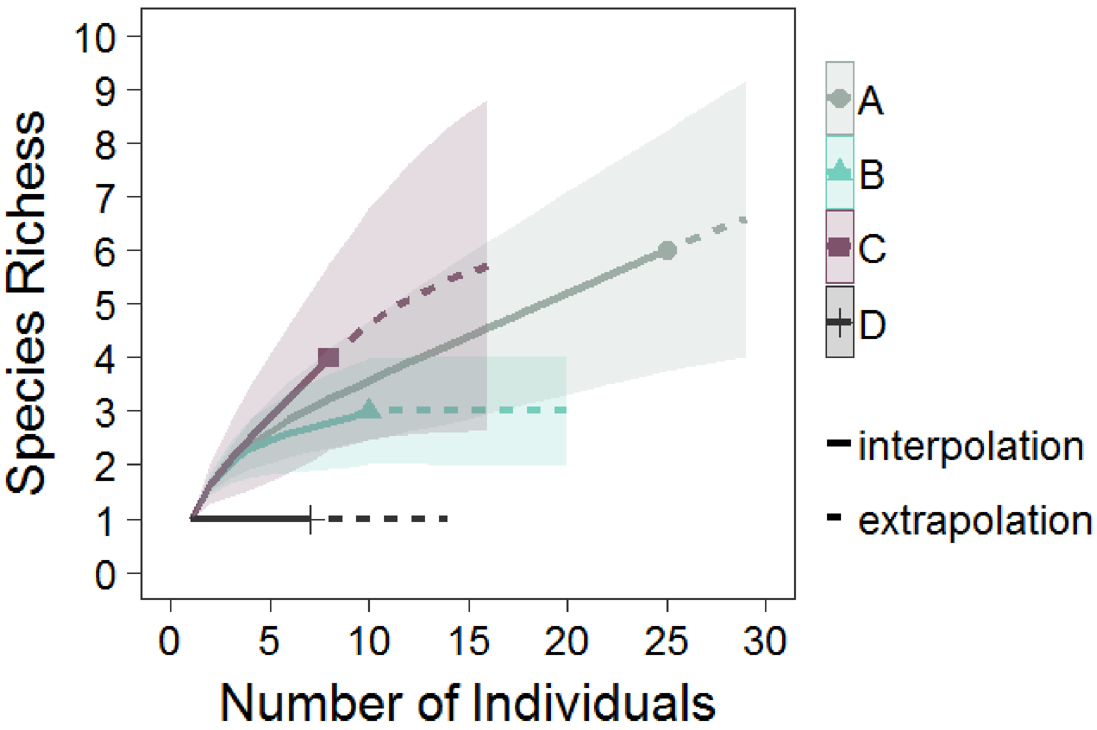

Figure 12. Rarefied small mammal species richness, and the associated 95% confidence intervals as a function of the number of individuals sampled in four non-native ruderal grassland patches (A-D) at Wilson's Creek National Battlefield, Missouri; May–July, 2016.

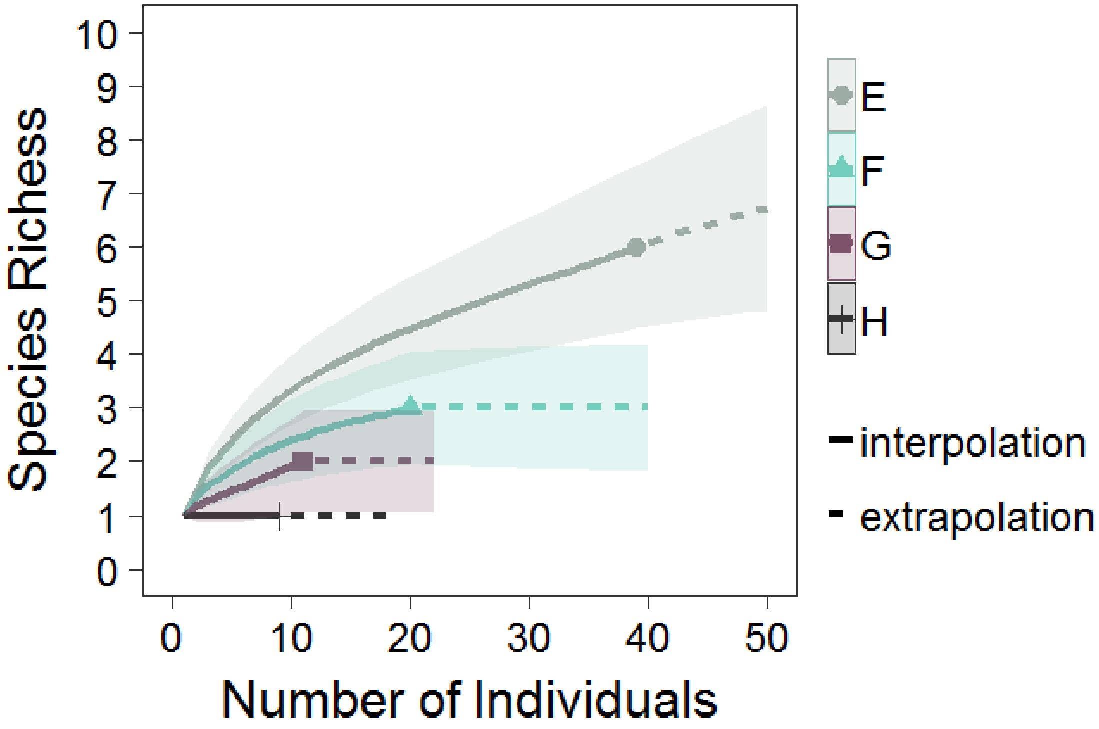

Figure 13. Rarefied small mammal species richness, and the associated 95% confidence intervals as a function of the number of individuals sampled in four upland deciduous woodland and forest patches (E-H) at Wilson's Creek National Battlefield, Missouri; May–July, 2016.

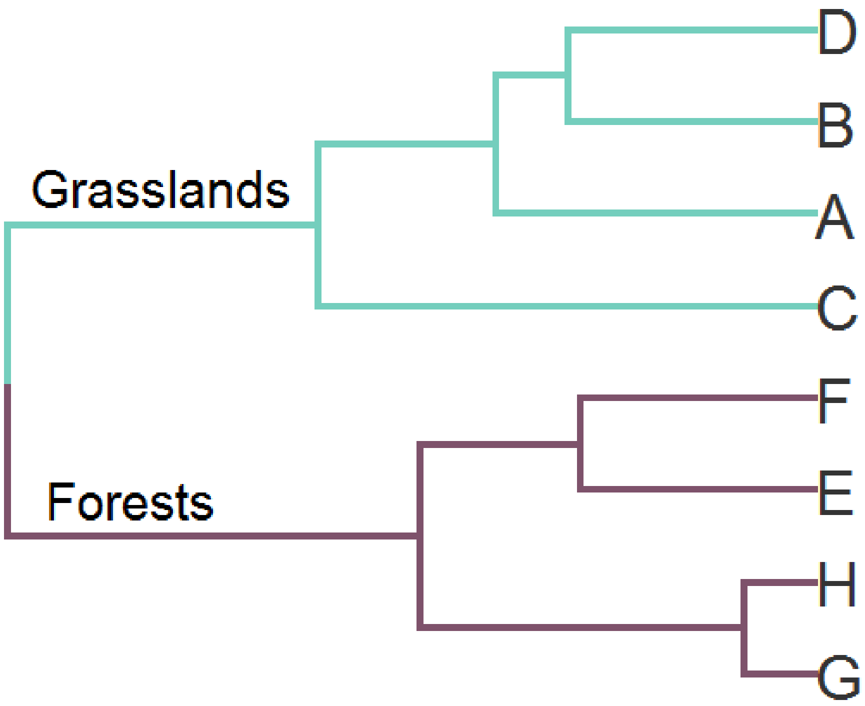

Figure 14. Bray-Curtis patch-level comparison of small mammal composition at four non-native ruderal grassland patches (A–D), and four upland deciduous woodland and forest patches (E–H) at Wilson's Creek National Battlefield, Missouri between May–July, 2016.

SUMMARY

Continued vegetation monitoring in my sampled patches would be beneficial, as they represent a wide breadth of the distribution of similar habitat within the park. Small mammal richness is expected to increase if non-native ruderal grasslands are restored back to native prairie vegetation (Clark et al., 1998; Mengak, 2004; Schweiger et al., 2000), but the Japanese honeysuckle in patch D is capable of outcompeting native vegetation. Prescribed burning and reseeding should result in small mammal succession following that of plant successional stage as specific habitat requirements are met (Schweiger et al., 2000). The deer mouse is expected to shift toward recently burned plots, whereas the Western harvest mouse, hispid cotton rat (Clark et al., 1998), Elliot's short-tailed shrew (Thompson et al., 2010), and prairie vole (Schweiger et al., 2000) are expected to occupy patches in later successional stages. However, if woody vegetation such as Eastern red cedar were to encroach upon grasslands from the surrounding forest patches, small mammal community structure is expected to shift along the gradient, potentially leading to a depauperate community dominated by white-footed mice (Horncastle et al., 2005).

Similar to the grasslands, forest richness is expected to increase with stand management (Nupp and Swihart, 2000). However, the low spatial integrity indices of the upland deciduous woodland and forests is worrisome because the persistence of many species depend upon the population dynamics in the surrounding landscape (Fantz and Renken, 2005). Small and isolated habitat patches are not suitable for species sensitive to fragmentation and may become homogenized by the white-footed mouse (Nupp and Swihart, 2000). This is especially problematic as white-footed mice are primary reservoirs for the Lyme disease spirochete, *Borrelia burgdorferi* (Ostfeld, et al., 1993). Furthermore, density and prevalence of infected ticks can increase with fragmentation of woodlands (Keesing et al., 2006; Logiudice et al., 2008; Mize, et al., 2011). Continuous habitats with greater biodiversity are more likely to contain non-competent reservoirs of *B. burgdorferi* and lower abundances of white-footed mice, which would reduce risk of human infection (Logiudice et al., 2008; Schmidt and Ostfeld, 2001; Turney, et al., 2014).

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APPENDICES

Appendix A. Abundance of tree, shrub, and herbaceous (non-grass) species found in non-native ruderal grassland patches (A-D) at Wilson's Creek National Battlefield, Missouri; July, 2016.

Species	Common Name	A	B	C	D
Ambrosia spp.	Ragweed	0	0	0	1
Apocynum cannabinum	Dogbane	0	1	0	0
Asclepias syriaca	Milkweed	0	1	0	0
Asclepias tuberosa	Butterfly Weed	2	0	0	0
Asteraceae	Asteraceae	3	0	0	1
Cirsium spp.	Native Thistle	2	0	0	2
Clematis spp.	Clematis	0	0	2	0
Daucus carota	Queen Anne's Lace	0	2	4	4
Desmodium spp.	Legume	0	2	1	5
Dianthus armeria	Deptford Pink	1	0	0	0
Erigeron spp.	Fleabane	0	0	1	1
Euphorbia spp.	Spurge	3	0	1	1
Eutrochium purpureum	Joe Pye Weed	1	1	0	0
Gleditsia triacanthos	Honeylocust	4	2	0	0
Hypericum perforatum	St. Johnswort	1	1	0	0
Ipomoea hederacea	Morning Glory	0	0	2	0
Juglans nigra	Black Walnut	0	0	1	0
Lespedeza cuneata	Sericea Lespedeza	1	8	0	1
Leucanthemum vulgare	Oxeye Daisy	0	0	1	0
Lonicera japonica	Japanese Honeysuckle	0	0	0	10
Medicago sativa	Alfalfa	0	0	1	0

(Appendix continued on next page.)

(Appendix A. Continued.)

Species	Common Name	A	B	C	D
Monarda citriodora	Horsemint	0	3	0	0
Opuntia humifusa	Eastern Prickly Pear	0	0	2	0
Passiflora incarnata	Purple Passion Flower	8	3	0	0
Prunus serotina	Black Cherry	2	0	0	5
Rhus copallinum	Winged Sumac	0	1	0	1
Rosa multiflora	Multiflora Rose	0	0	2	0
Rubus occidentalis	Black Raspberry	8	10	0	4
Rudbeckia triloba	Brown Eyed Susan	0	0	0	2
Ruellia humilis	Wild Petunia	0	0	1	0
Solanum carolinense	Carolina Horsenettle	3	0	3	1
Solidago ptarmicoides	Goldenrod	0	0	0	1
Symphoricarpos orbiculatus	Coralberry	6	6	1	7
Teucrium canadense	Canada Germander	1	6	0	5
Tilia americana	American Linden	0	0	0	2
Toxicodendron radicans	Eastern Poison Ivy	1	3	0	0
Tragia spp.	Tragia	0	0	2	0
Ulmus alata	Winged Elm	0	1	0	0
Verbesina alternifolia	Wingstem	0	0	1	0
Vernonia missurica	Missouri Ironweed	2	4	0	0
Vitis vulpina	Frost Grape	0	1	3	0

Appendix B. Abundance of tree species found in upland deciduous woodland and forest sites (E-H) at Wilson's Creek National Battlefield, Missouri; July, 2016.

Species	Common Name	A	B	C	D
Carya cordiformis	Bitternut Hickory	0	0	1	0
Carya ovata	Shagbark Hickory	0	2	1	0
Carya texana	Black Hickory	1	5	0	0
Carya tomentosa	Mockernut Hickory	0	1	4	3
Celtis occidentalis	Hackberry	21	9	5	2
Fraxinus americana	White Ash	0	0	2	1
Gleditsia triacanthos	Honey locust	0	0	4	1
Juglans nigra	Black Walnut	13	5	1	3
Juniperus virginiana	Eastern Red Cedar	1	10	2	8
Maclura pomifera	Osage Orange	5	3	0	0
Morus rubra	Mulberry	0	2	0	0
Prunus serotina	Black Cherry	0	0	2	3
Quercus marilandica	Blackjack Oak	0	0	1	0
Quercus muehlenbergii	Chinkapin Oak	3	6	16	1
Quercus rubra	Red Oak	1	0	3	5
Quercus stellata	Post Oak	0	1	0	25
Quercus velutina	Black Oak	3	5	8	0
Robinia pseudoacacia	Black locust	0	0	1	0
Sassafras albidum	Sassafras	0	3	5	0
Tilia americana	American Linden	0	0	2	0
Ulmus americana	White Elm	0	0	1	0
Ulmus rubra	Slippery Elm	13	11	1	1