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Observable Persistent Effects of Habitat Management Efforts
in the Ozark Highlands After 10 Years

A thesis submitted in partial fulfillment
of the requirements for the degree of
Master of Science in Biology

by

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Bachelor of Science in Biology, 2011

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Abstract

I investigated the lasting impacts of a management plan designed to improve oak regeneration and benefit wildlife in the Ozark Highlands in Madison, Co., AR. To assess the efficacy of the management plan, I used variables relevant to the success and establishment of oak trees. Controlled burns and selective logging were used to thin the canopy, increase ground level productivity, and increase the abundance of small mammals. I used measurements of overstory and understory densities, light availability, and the density of mice in the genus *Peromyscus* across time to look at the lasting impacts of management. Different treatment plots were used to investigate the impact of each management action separately (Burn or Cut) and in combination (Burn and Cut) relative to unaltered control plots. Measurements were compared between pre-treatment, post-treatment, and 10-years post-treatment time points. I found that a 10-year lapse in management resulted in a complete return to pre-treatment values in overstory density. I also saw a decline below pre-treatment values in understory density and *Peromyscus* density. Analysis of light availability at the forest floor revealed a persistent effect of treatment. I conclude that while initial treatment was effective, 10 years between management events is too infrequent to achieve the desired long-term changes within my study system. More frequent management may be more effective in meeting the management goals for this Ozark system.

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Introduction

Frequency and intensity of disturbance shape the composition and diversity of communities (Miller, Roxburgh, & Shea, 2011). In forested ecosystems, fire and severe storms are the most frequent and intense forms of natural disturbance (Schulte & Mladenoff, 2005). Frequency of wildfire and tree fall from storms affect the rate of nutrient recycling, extent of canopy openness, rate of understory recruitment into the canopy, and the degree to which the seed bank is utilized in forests (Roberts, 2004). Without these natural disturbance events, succession within a forest community continues towards some homogeneous climax as shade tolerant hard woods gradually outcompete herbaceous understory plants for space, light, and nutrients (Gorham, Vitousek, & Reiners, 1979; Odum, 1969).

In North America, natural disturbance regimes have been disrupted as a direct result of human settlement and land use. Management and resource exploitation for human interests has resulted in the use of unsustainable logging practices and fire suppression over the last 100 years (R P Guyette, Spetich, & Stambaugh, 2006; Marbut, 1914; Smith et al., 2004; Stambaugh & Guyette, 2006). Now, diverse systems once governed by semifrequent disturbance regimes of fire and harvest (Braun, 1964; Dey, Guyette, & Stambaugh, 2004; Guyette et al., 2006; Houck, 1908; Sabo, Lockhart, & Hilliard, 2004) have given way to even aged, closed canopy forests with low understory diversity (Spetich, 2002).

In the Oak-Hickory forests of the Ozarks, decades of fire suppression have altered the forest community. We now know that Oak trees (genus *Quercus*) are fire dependent (Oak-Fire Hypothesis; Abrams, 1992) and are in decline as a direct result of fire suppression (Abrams, 2005). In these systems, semi-frequent burning increases the likelihood that oak seedlings will establish and that oak saplings will not be outcompeted by fire-intolerant, fast growing trees such as maples (genus *Acer*) (Brose & Van Lear, 1998; Hutchinson, Long, Rebbeck, Sutherland, & Yaussy, 2012; Hutchinson, Sutherland, & Yaussy, 2005). Fire suppression over

the course of the 20th century has allowed the understory to become occluded, preventing shade-intolerant oak saplings from establishing and regenerating (Arthur, Alexander, Dey, Schweitzer, & Loftis, 2012; Barnes & Van Lear, 1998; Brose, Dey, Phillips, & Waldrop, 2013; Peterson & Reich, 2001). Oak saplings require ample light in the understory and are slow growing (Arthur et al., 2012; Crow, 1992). As a result, shade-tolerant plants have been able to outcompete oak seedlings throughout much of the Ozarks where the canopy is closed and fire is absent (Alexander, Arthur, Loftis, & Green, 2008). Many forested plots now lack an existing pool of oak saplings ready to be recruited into the canopy, making restoration efforts slow or altogether unsuccessful (Wendel & Smith, 1986). Historically, semi-frequent fires (every 2.8 years, on average; Cutter & Guyette, 1994), were sufficient to kill fast growing competitors such as maple trees, giving oak trees a competitive edge. Successful oak management requires the reintroduction of fire to kill fire-intolerant competitors. The creation of artificial tree fall gaps through selective logging can help to expedite the process by increasing light penetration to the forest floor; allowing mid-story oaks to recruit into the canopy.

Modern forest managers often turn to the shelterwood technique (Brose, Van Lear, & Keyser, 1999) to begin the restoration process in oak forests. The shelterwood technique employs an aggressive cut following a productive acorn mast year to open the canopy, increasing light at the forest floor. Oak seedlings are then able to establish. A prescribed burn is then carried out several years later to kill fire-intolerant saplings. Oak trees are able to regenerate after fire (Arthur et al., 2012) and continue to grow. Several burns are required to give oak trees the advantage over their faster growing competitors (Waldrop et al., 2008). Eventually, a period without fire is necessary (10-30 years) for mid-story oak trees to release into the canopy (Arthur et al., 2012; King & Muzika, 2014; Peterson & Reich, 2001). Management techniques require continued action in order to effectively alter the community composition in the long term; single treatments are ineffective (Brose et al., 2013).

Management practices designed to regenerate oak forests can benefit a variety of native Ozark flora and fauna. In the early stages of an oak regeneration management plan, the canopy is opened and fire is reintroduced. While the overstory community takes decades to recover (Richard P. Guyette & Kabrick, 2002), the understory community responds within the first 1-2 years after treatment (Waldrop et al., 2008) shifting to favor sun-loving grasses and forbs shortly after canopy thinning and the reintroduction of fire (Peterson, Reich, & Wrage, 2007). The diversity of sun-loving understory plants continues to increase with continued disturbance (Blake & Schuette, 2000; Dey & Hartman, 2005; Knapp, Stephan, & Hubbart, 2015; Waldrop et al., 2008). Disturbance is particularly effective in altering understory community because it removes leaf litter that may limit germination (Kinkead, Kabrick, Stambaugh, & Grabner, 2013) and destroys midstory plants that compete for light (Barnes & Van Lear, 1998). Small mammals also benefit from Oak regeneration focused management. In the first five years following treatment, there is an observable increase in *Peromyscus* abundance (Fantz & Renken, 2005; Fisher & Wilkinson, 2005; Zwolak, 2009) although the effect seems to dissipate without continued treatment (Fisher & Wilkinson, 2005; Martell, 1983). In Ozark Oak-Hickory forests, managers can expect to see an increase in small mammal abundance and understory diversity in the initial years following treatment. Maintaining these characteristics and achieving long term changes in habitat structure such as overstory composition and habitat openness, however, may require time and continued management actions.

At the study site, McIlroy Madison County Wildlife Management Area (MMCWMA) in Madison Co., AR a subset of the habitat was managed between 2007 and 2008 to restore the habitat and improve oak recruitment in distinct patches (Douglas, 2010). Portions of the site were selectively logged in the summer of 2007 and controlled burns were implemented in March of 2008. The goals of the treatments were to; 1. Increase light penetration to the forest floor by decreasing the density of overstory trees, 2. Decrease the density of fire intolerant trees and

saplings in the understory to increase plant diversity via the reintroduction of fire, and 3. Increase the abundance of small mammals. Douglas (2010) used mature tree density, understory plant density and diversity, and small mammal abundance at a selection of treated habitat plots to assess the effect of treatment over the first two years immediately following manipulation. Mammal abundance and plant density and diversity data were collected over the course of 5 years, but the most informative data came from two distinct time points: pre-treatment in 2005 and post-treatment in the spring of 2009 (2008 for understory). Comparison of pre and post-treatment data revealed; 1. A decrease in overstory density at cut sites, 2. A change in community composition and diversity favoring sun-loving herbaceous plants, grasses, and shrubs in the understory in all treatment types, and 3. An increase in small mammal abundance at treatment sites (B, C, and especially BC) (Douglas 2010). No further management actions have occurred since 2008.

The immediate impacts of the 2007 and 2008 management actions were quantified in the Douglas (2010) study. It is well understood that selective cuts and the reintroduction of fire have an immediate impact on an ecosystem and that those changes are beneficial to oak species in an Oak-Hickory forest (Abrams, 1992; Arthur et al., 2012; Brose, Van Lear, & Keyser, 1999). However, most studies of this type have focused on the long term impacts of multiple management events on oak recruitment. The longer-term (10+ year) impacts of a single treatment action are less well understood. Additionally, oak management functions by creating a habitat more suitable for oaks. Increases in oak survival and recruitment are brought about through the reintroduction of fire, decrease of competitive plants in the understory, and increase in light availability at the forest floor. As a result, management actions have immediate and lasting impacts on the plant community, habitat structure, available light environment, and population dynamics of primary consumers within an ecosystem; impacts that have traditionally been overlooked. Since few studies have examined the interplay between treatment and these

community level variables of interest, we lack a clear understanding of how the changes to the community caused by oak-specific management persist through time.

The management history at MMCWMA makes it a suitable study system to look at the condition of a treated oak forest after 10 years without continued management action. While Douglas (2010) was primarily concerned with changes in plant diversity and small mammal abundance immediately after land management efforts, my study focused on changes in community level variables affected by oak-specific management (habitat characteristics and *Peromyscus* density) through time. Specifically, I compared overstory tree density, understory densities, and small mammal density among pre-, post-, and 10-years post-management time points. I have also provided data linking management efforts to percent shade cover on the habitat substrate.

Methods

Description of the Field Site

The McIlroy Madison County Wildlife Management Area (MMCWMA), Ozark Natural Science Center (ONSC), and Bear Hollow Natural Area (BHNA) (Figure 1) encompass over 6,000 hectares of nearly contiguous woodland located in Madison County, Arkansas. The site is composed of steep canyons with exposed limestone bluffs carved out by the permanent and intermittent streams. The woodlands of the study site are primarily a mix of hickory (*Carya spp.*), oak (*Quercus spp.*), and pine (*Pinus spp.*) dotted with retention ponds and manicured food plots used for game management. The Arkansas Game and Fish Commission designed a new management plan to benefit game species by increasing oak recruitment and acorn mast crops to benefit turkey (*Meleagris gallopavo*) and white-tailed deer (*Odocoileus virginianus*) populations. The intended management would also serve to open up the canopy to increase solar radiation at ground level, rejuvenate the herbaceous seed-rich understory, and ultimately increase ground-level food availability for small mammals, including *Peromyscus*, on a subset of

the land. The management plan was implemented from early summer 2007 to mid-March 2008. Six plots (ranging from 4-26 ha) were selectively logged with the goal of reducing canopy cover density (due to imprecise record keeping, it is impossible to provide much specific detail about the harvests beyond that non-oak species were targeted). Six plots (three previously logged, three control) were burned in the spring of 2008 (completed March 12). The end result (Figure 1) was nine treatment plots; three cut only (C), three burn only (B), and three combination burn-cut treatments (BC). Three additional sites were selected as controls (Con) for comparison. Control sites were left unmanaged for the duration of the study.

Vegetation Sampling

Overstory Density

To build upon the previous work of Douglas (2010), I counted the number of mature trees making up the overstory at each of the 12 sites described above in the summer of 2018 (mid-July). I defined mature trees as any tree having a diameter at breast height (DBH, ~1.5m from above the forest floor) ≥ 10 cm. I used circular quadrats (Cox 1980; Lindsey et al. 1958; Patterson and James 2009) to sample the density of mature trees. From the center of each of my 12 study plots, I selected a random compass bearing (1-360) using a random number generator (Random UX, UX apps). Following the random bearing, I generated a second random number between 0 and 100 and walked that number of paces. I used this point as the center point of the quadrat, provided that the surrounding area appeared representative of the soil, topography, and vegetative community of the site as a whole (Daubenmire 1959). Using a 100m measuring tape, I measured 11.3 m from the center and outlined a circle with Area = 0.04ha. I used the outer boundary of the circle as the boundary for my sample. Every tree whose trunk had DBH ≥ 10 cm and was entirely within the bounds of the circular quadrat drawn was counted. I identified trees to genus (Moore, 1994). I followed the above procedure in

quadruplicate at each site, resulting in four estimates of overstory density per plot, 12 in total per treatment type.

Understory Density

I sampled understory vegetation at each site in the summer of 2018 (mid-July). A mid-summer date was chosen to ensure all understory vegetation of interest was at the peak of its growing-season productivity. I used circular quadrats (Area = 15m², r = 2.2m) (Cox 1980; Lindsey et al. 1958; Patterson and James 2009) to estimate understory vegetation density. Quadrat center locations were identical to those described above (see overstory density, adjusted quadrat radii: 2.2m). I measured any plant greater than 50cm in height but < 10cm DBH whose root base or a substantial portion (~50% or greater) of its foliage was bounded by the 15m² circle. Number of plants was recorded at four separate locations in each of the 12 study sites, resulting in 12 estimates of understory density per treatment type.

Small Mammal Sampling

I sampled the abundance of small mammals in the genus *Peromyscus* (IACUC protocol #17038) in spring (March 20-June 20) of 2017. Within each treatment plot (Figure 1), I selected a mammal trapping site (40m x 90m) that was roughly level and devoid of cliffs or rock features >5m tall. I used the same GPS coordinates as Douglas (2010) for 10 of the 12 sampling locations. The remaining two locations were selected within the same geographical bounds as Douglas (2010) but localities were altered slightly to accommodate an increase in sampling grid area relative to the previous methodology. I sampled small mammals at 12 sites; three Con, three C, three B, and three BC.

I used Sherman live traps (3"x3.5"x10") (LFAHD Folding Trap and 3310A Non Folding Trap, H.B. Sherman Traps) to trap small mammals once per season at each of my 12 plots (described above). I placed traps in gridded 5x10 arrays (Getz 1982; Gottfried 1982; Guthery

1982; Mihok 1982; White et al. 1982; Williams et al. 2001) with approximately 10m between each trap station (Figure 2). Due to equipment limitations, I was only able to trap three sites concurrently (50 traps each; 150 traps total). To minimize travel time between grids during setting and checking I grouped the 12 sites in threes by geographic proximity. Total trapping effort was composed of four separate trap “sessions” (Table 1).

I trapped for three consecutive nights in each session. Using a mixture of peanut butter and rolled oats (Francl et al. 2004) I baited and set each trap starting two hours before sunset each night. All traps were set by sunset. I placed a cotton ball in each trap to prevent hypothermia (Guthery 1982) and minimize in-trap deaths. I checked traps starting at sunrise the following morning. Depending on capture success, the last trap was usually checked between one and two hours post sunrise. I standardized setting and checking times relative to sunset and sunrise to prevent trapped animals from occupying traps for longer than 12 hours, reducing unnecessary stress or mortality. During the day, traps were left closed to prevent unintentional capture.

Every capture event was logged and I collected a series of data before the individual was released back into the environment. Data collected included trap location within the grid, genus of the animal, and if applicable the individual’s three digit ID. Each new individual was marked with a unique three-digit metal ear tag (Self Piercing Fish Tag, National Band and Tag Company). I identified individuals as members of the genus *Peromyscus* following the morphological characteristics outlined by Sealander and Heidt (1990).

Trap sessions were kept short (4 days, 3 nights) to avoid error associated with birth, death, immigration, or emigration. I assumed the population was closed over this length of time and that births and deaths would not impact my population estimate over 4 days (Williams et al., 2001; Seber, 2002). No individuals were captured at more than one study site, indicating that the short trap sessions successfully negated any confounding impact of migration. I also

assumed that no marks were lost between samples as no captured animals showed any evidence of a previously lost marks. Low sample size prevented me from testing whether all animals were equally likely to be trapped. I used an evenly spaced trapping grid and sampled all points within the grid with uniform effort to ensure equal probability of capture at all trap locations (Seber, 2002) to ensure that my sample was random and to meet the assumptions of equal capture probability as well as possible. Low capture rates across all sites prevented convergence of capture-recapture programs. To increase capture histories and allow convergence using a simple Lincoln Petersen estimator, I collapsed capture records across each 3-day sampling period. I treated the first two nights in each trapping session as the first sampling period, and treated the third night as the second sample to calculate my abundance estimates.

I estimated density (individuals per hectare) and variance using the Lincoln-Peterson method corrected for small sample size (Chapman 1951, Williams et al. 2001):

$$\hat{N} = \frac{(K+1)(n+1)}{(k+1)} - 1 \qquad \widehat{var}(\hat{N}) = \frac{(K+1)(n+1)(K-k)(n-k)}{(k+1)^2(k+2)}$$

where: N = Estimated number of animals in the population, n = number of animals captured in the initial sampling event, K = number of animals captured in the 2nd sampling event, and k = number of marked animals caught in the second sampling event. The Lincoln-Petersen method estimates abundance, and so to convert to a density, I first had to estimate the effective area sampled by my grids. Effective trapping area ($\hat{A}(\hat{W})$) was estimated following Dice (1938). I placed a boundary strip of width = 15.9 m around each grid shape (2005 and 2009 data = square, 2017 data = rectangular) (Figure 3). The width of this strip was estimated by taking the average of the greatest distance between capture locations (\bar{d} ; Williams et al, 2001) within a grid for each animal captured more than once. Due to limited data collection in 2005 and 2009, boundary strip width was based on the 12 observations from 2017 that fit the criteria. However,

since the estimate was close to other estimates of *Peromyscus* home range radii (13m: Wolff, 1985; <31m; Douglas, 2010; 18m-48m; McNab, 1963), I deemed it acceptable. The dimensions of each grid in conjunction with the estimated boundary strip were used to estimate the effective trapping area of each grid (Figure 3) using the following equations:

$$\hat{A}(\hat{W}) = \begin{cases} \text{For a square} & L^2 + 4L\hat{W} + \pi\hat{W}^2 \\ \text{For a rectangle} & LH + 2L\hat{W} + 2H\hat{W} + \pi\hat{W}^2 \end{cases}$$

The variance of ($\hat{A}(\hat{W})$) was approximated by including a variance term for the boundary strip, \hat{W} :

$$\widehat{var}[\hat{A}(\hat{W})] = \begin{cases} \text{For a square} & (4L + 2\pi\hat{W})^2 \cdot var(\hat{W}) \\ \text{For a rectangle} & (2L + 2H + 2\pi\hat{W})^2 \cdot var(\hat{W}) \end{cases}$$

where: L = Length, H = Height, and \hat{W} = width of the border strip. The top equations for both the effective trapping area and the associated variance were used to calculate each parameter for the square grids used in 2005 and 2009. The bottom equations were used for equations involving the rectangular grids used in 2017. I used the Delta method (Powell, 2007; Seber, 2002; Williams et al., 2001) to approximate the variance of the width of the boundary strip ($var(\hat{W})$) as $(\frac{1}{2})^2(var(\hat{d}))$ (Williams et al. 2001; pgs. 314-315). I assumed that small mammal density was a linear function, increasing with increasing area to satisfy assumptions necessary to apply the Delta method. I considered the assumption of a linear abundance to area relationship to be reasonable in the contiguous closed-canopy forest sampled in this study. With abundance, effective trapping area, and approximations of their associated variances, I calculated density (individuals per m²) for each trapping site in all years as well as approximate variances using the following equations:

$$\hat{D} = \frac{\hat{N}}{\hat{A}(\hat{W})} \quad \widehat{var}(\hat{D}) = \frac{\hat{N}^2 var[\hat{A}(\hat{W})]}{[\hat{A}(\hat{W})]^4} + \frac{var(\hat{N})}{[\hat{A}(\hat{W})]^2}$$

where: D = Density, N = Abundance, and $\hat{A}(\hat{W})$ = Effective trapping area. I used the three different trapping locations (i.e. Con1, Con2, Con3) as separate treatment replicates to estimate average *Peromyscus* densities for each treatment type. The variance of each average density was calculated by incorporating variance terms for each treatment replicate following the Delta method and the following equations:

$$\begin{array}{l} \hat{D}_{ConAVG} = \frac{\hat{D}_{Con1} + \hat{D}_{Con2} + \hat{D}_{Con3}}{3} \\ \hat{D}_{ConAVG} = \frac{1}{3}\hat{D}_{Con1} + \frac{1}{3}\hat{D}_{Con2} + \frac{1}{3}\hat{D}_{Con3} \end{array} \quad \left| \quad \begin{array}{l} \widehat{var}(\hat{D}_{ConAVG}) = \widehat{var}(\hat{D}_{Con1}) \left[\frac{\partial \hat{D}_{ConAVG}}{\partial \hat{D}_{Con1}} \right]^2 + \widehat{var}(\hat{D}_{Con2}) \left[\frac{\partial \hat{D}_{ConAVG}}{\partial \hat{D}_{Con2}} \right]^2 + \widehat{var}(\hat{D}_{Con3}) \left[\frac{\partial \hat{D}_{ConAVG}}{\partial \hat{D}_{Con3}} \right]^2 \\ \widehat{var}(\hat{D}_{ConAVG}) = \frac{1}{9} \widehat{var}(\hat{D}_{Con1}) + \frac{1}{9} \widehat{var}(\hat{D}_{Con2}) + \frac{1}{9} \widehat{var}(\hat{D}_{Con3}) \end{array} \right.$$

The examples shown above are for determining the average density and associated variance at control sites (where: D_{ConAVG} = Average treatment density at control sites and D_{Con1-3} = Point estimates of density at each site) although the process was identical for all other treatment types. The density point estimates for each of the three different sampling grids were averaged to attain an average treatment density (Left). The associated variance for the estimation of density incorporates the variance terms associated with each individual density estimate from trapping grids Con1, Con2, and Con3 (Right). Density estimates (individuals per m²) were converted to estimates of individuals per hectare with approximated variances following the equations:

$$D_{ha} = 10,000(D_{m^2}) \quad \quad \quad var(D_{ha}) = 10,000^2(var D_{m^2})$$

where: D_{ha} = density per hectare and D_{m^2} = Density per m². I constructed 95% confidence intervals using the calculated variance. To account for significant changes in *Peromyscus* densities at control sites over time, I calculated relative densities by a log ratio of each treatment density to the control density for that year (Powell et al., 2007) using the following equation:

$$\alpha_T = \text{Log} \left(\frac{D_T}{D_C} \right)$$

where: α_T = the relative density of a given treatment (T), D_T = *Peromyscus* density of a given treatment in a given year, D_C = *Peromyscus* density of the control for that year. Relative density estimates presented here are a measure of the difference between a treatment effect and its relevant control density. The variance of the log ratios (Powell, 1998; pg. 64) was given by the equation:

$$var(\alpha_T) = var(D_C) * \left(\frac{1}{D_C}\right)^2 + var(D_T) * \left(\frac{1}{D_T}\right)^2$$

where: α_T = the relative density of a given treatment (T), D_T = *Peromyscus* density of a given treatment in a given year, D_C = *Peromyscus* density of the control for that year. Variance values were used to construct 95% confidence intervals around each relative density estimate. A single estimate is presented for control and treatment types at each time point.

Note that Douglas initially recorded low abundance at all sites in 2005 using a square 40 m x 40 m grid with paired trap stations. In an effort to increase trap success through an increase in sampling area, I altered the sampling design in 2017 in favor of a single trap per trap-station resulting in an equal number of traps spread over roughly double the area (0.16 ha vs 0.36 ha). Initial estimates of abundance reflect the number of animals occupying effective trapping areas of 0.494 ha (2005 and 2009) and 0.853 ha (2017). The calculation of density and its associated variance followed here is my best effort to make estimates taken in 2005 and 2009 directly comparable to the estimates made in 2017. These estimates account for the differences in grid shape and area sampled making it unlikely that the change in methodology between years accounts for the differences seen in the data.

Light Penetration to the Forest Floor

In July of 2018, I used a novel methodology to estimate coverage of solar radiation impinging to the forest floor. Mid-summer was chosen to ensure peak density of foliage that may act to obstruct solar radiation penetration to the ground. At each rectangular, gridded array

used to sample small mammal abundance (described above) I established a transect that ran along the 90 m of the middle trap row (Figure 2). I sampled on days with no cloud cover between the hours of 1100 and 1300 when incidental solar radiation is assumed to be high, standardizing variation associated with angle of the sun. I used standard white ping pong balls (20mm radius) as a fixed surface area object and estimated light coverage on the upper hemisphere of the ball by eye to the nearest 5%. Three ping pong balls were dropped from chest height every three paces (~1-1.5m) and each was permitted to fall and bounce until stationary. If a ball was not directly on the ground (e.g. caught in waist level vegetation), it was recast. I garnered 100 samples for each plot following this methodology. Each site took approximately 15 minutes to sample. I assumed that the change in solar elevation angle between samples at mid-day over this amount of time was negligible. I used the data to generate solar radiation distributions and test for treatment dependent differences between sites. Observations were binned into one of five groups of 20% (0-20%, 21-40%, 41-60%, 61-80%, and 81-100%) depending on the estimation of light coverage to decrease the impact of observer error.

Statistical Analyses

All statistical analyses were conducted in RStudio Version 1.1.456 (<https://www.r-project.org>). All means are presented as mean \pm 2SE unless otherwise stated. Data included 2005, 2008 (understory), and 2009 (overstory and mammal abundance) measurements collected by Douglas, as well as new data from 2017 and 2018. To account for the effects of unbalanced sampling and random sampling effects, I constructed linear mixed effects models (function lme) for vegetation variables. I used Treatment type (Control, Cut, Burn, and BC) and Year (2005, 2008/2009, and 2017/2018) as fixed effects. Site number was included as a random effect to account for variance among treatment types due to random site differences. Residuals were tested for normality using the Shapiro-Wilks test. Overstory density estimates

met the assumptions of normality and were analyzed using parametric methods. Understory density was heavily right skewed (skewness = 0.719), data were square root transformed (skewness = 0.137) to meet assumptions of normality (Figure 4) and then analyzed using parametric methods. I used 2-way repeated measures ANOVA to look at the effects of treatment type on each response variable over time. Significant results were further analyzed post hoc via the Tukey-Kramer procedure which accounts for unbalanced sampling. Small mammal density estimates were uniformly very low and highly variable. I compared density estimates and their associated 95% confidence intervals to test for significant differences in density between treatments and across time points. Light penetration data were analyzed using an RxC (row by column) contingency table organized with treatment types as columns and counts of observations as rows (Table 2). A chi-square test statistic was used to test for a difference in light distribution among treatment types. The test statistic was calculated relative to an expected distribution derived from the data set as a whole. I adopted a 5% type I error for all tests.

Results

Overstory Density

A total of 88 overstory density observations were recorded across the 12 sites over the three distinct time points. Total number of mature trees ranged from 1 to 39 per 400m² across all treatments and time points. Average tree count was 17.34 ± 1.55 across all sites. Overstory data divided by treatment types and time points shows a spread in density initially achieved by treatment in 2009, followed by a return towards baseline observed 10-years post-treatment (Figure 5). Averages of all sites in each year (2005: 17.5 ± 4.30 , n = 11, 2009: 18.45 ± 3.48 , n = 29, 2018: 16.65 ± 1.74 , n = 48) are closely clustered. To investigate differences among treatment types over time I generated a mixed effects linear model;

$$(Overstory \sim Treatment * Year, random = \sim 1 | Site)$$

Where: “Overstory” = count of overstory trees per 400m², “Treatment” = treatment designation (Con, B, C, BC), “Year” = 2005, 2009, or 2018, and “Site” = individual site designation (i.e. Con1, Con2, or Con3). My model explained 49.4% of the variance (function r.squaredGLMM), an improvement of 16% over a fixed effects only model. Overstory density at C and B sites between 2005 and 2009 differed significantly (p-values = 0.01 and 0.0072 respectively) from what the model would predict under the null hypothesis of no effect. When the residuals of this model were tested for normality by the Shapiro –Wilk test the result was non-significant (W = 0.987, p-value = 0.562), indicating that parametric methods were appropriate. I used a 2-way repeated measures ANOVA to analyze the relationships between overstory tree density, year, and treatment type. Analysis of variance revealed that both treatment type (p = 0.0139) and treatment type by year interactions (p = 0.0104) were significant predictors of overstory density. Post hoc analyses reveal a single significant contributing contrast that accounts for the observed treatment type by year interaction; post-treatment and 10-years post-treatment at control sites (df = 68, t-ratio = 3.003, p = 0.0103).

Understory Density

I used 91 observations (11 from 2005, 32 from 2008, and 48 from 2018) to estimate understory density at 12 distinct plots of varying treatment regime before, after and 10-years after treatment. The average understory density in a 15m² quadrat was 47.15 ± 0.52 across samples ranging from 0 to 188. Mean understory density counts were 56.47 ± 13.61 pre-treatment, 65.87 ± 15.11 post-treatment, and 28.46 ± 8.21 10-years post treatment. There is little change in density between 2005 and 2009, but a clear decrease in understory density is seen across all treatments in 2018 (Figure 6).

I generated a mixed effects linear model;

*(Understory ~ Treatment * Year, random = ~1 | Site)*

where: “Understory” = count of understory plants per 15m², “Treatment” = treatment designation (Con, B, C, BC), “Year” = 2005, 2008, or 2018, and “Site” = individual site designation (i.e. Con1, Con2, or Con3). The model used understory as the dependent variable and treatment type and year as independent variables to further investigate variation over time and treatment type. My model included site number as a random effect and explained 59.6% of the variance (function `r.squaredGLMM`), an improvement of 19% over a fixed effects only model. The linear mixed effects model indicates sites sampled in 2018 had significantly lower understory densities (p -value = 0.0001) than predicted under the null hypothesis. When the residuals of the mixed effects model were tested for normality by the Shapiro –Wilk test the result was non-significant ($W = 0.990$, p -value = 0.698), indicating that parametric methods were sufficient. Analysis of variance revealed a significant effect of year ($p = < 0.0001$). I used the Tukey-Kramer procedure to evaluate which between-year comparisons most influenced understory density. Understory densities differed significantly at 10-years post treatment when compared to both pre-treatment ($df = 71$, t -ratio = 4.067, $p = 0.0004$) and post-treatment ($df = 71$, t -ratio = 7.299, $p < 0.0001$) values. Further evaluation indicates that densities in all treatment types (Con, C, B, and BC) differ significantly between post- and 10-years post-treatment. Comparison of 2005 and 2018 data however, reveals that differences in understory density only exist at Con ($df = 71$, t -ratio = 2.451, $p = 0.0436$) and C ($df = 71$, t -ratio = 3.943, $p = 0.0005$) sites, not B or BC sites.

Small Mammal Density

Thirty-eight estimates of small mammal density across the 3 time points and 4 treatments were used for analyses. Density estimates ranged from 0 to 46 individuals per ha (Table 3). Between year differences before (2005) and after (2009) treatment were most apparent at C (3.4 to 22.3) and BC (12.1 to 46.2) sites, indicating a positive association between canopy opening events and small mammal density. In 2017, all treatments had lower densities

than observed in 2009. When compared to 2005 values, 2017 densities were either equal to (B and C plots) or less than (Con and BC plots) the initially observed values. Between treatment comparisons in 2017 revealed no persistent differences among treatment types.

Density estimates at control sites differed significantly across time (Table 3). Relative density estimates corrected for the variation in densities observed at control plots (Figure 7). I compared 95% confidence intervals among treatment types in each year, as well as between years for each treatment type. There was a significant change in small mammal density at BC sites between 2009 (0.16 – 0.86) and 2017 (-0.41 – 0.13). I found no other evidence of significant treatment or time effects in comparisons of relative densities.

Light penetration

Light penetration to the forest floor was analyzed via chi-square contingency table. Under the null hypothesis there is no difference among treatment types after 10 years. I expected similar distributions of percent light availability regardless of treatment type. I anticipated observations of light penetration to be unevenly distributed among 20% bins, but that each 20% bin should be evenly occupied across treatment types. I generated expected distributions following Zar (2010), pooling all observations and calculating what proportion of observations should fall into each bin under the assumption of no persistent treatment effects (Table 2). Distributions of light availability measured in 2018 were compared to this expected distribution by treatment type (Figure 8). There was a significant difference among treatment types ($X^2 = 50.2$, $df = 12$, $p < 0.001$) from the expected distribution. Component chi-square term values from each individual observed:expected comparison were also viewed critically (X^2 values for each comparison; Table 2). The largest chi-square terms which contributed most to the overall chi-square test statistic of 50.2 were seen in the higher light bins (61-80% and 81-100%). Burn and Burn and Cut sites had a higher proportion of high light (61-100%) observations than expected, while Control and Cut sites had a lower proportion than expected

(Figure 8). Control sites also had a high chi-square value in the 0-20% bin where observations were more frequent than expected.

Discussion

Overstory Density

Selective logging was intended to thin the overstory to increase light penetration and productivity at the forest floor. The reintroduction of fire was expected to aid in the regeneration and establishment of oak trees while simultaneously killing fast-growing, fire-intolerant competitors. Due to the slow growth rate of hardwood trees I expected to detect some persistent effect of treatment even after 10 years. The lack of observable difference between 2005 and 2018 indicates an unforeseen return to pre-treatment densities. Tukey-Kramer comparisons show that Con sites differed significantly post and 10-years post-treatment. Continued recruitment at Con sites may explain this outcome and indicates that without treatment, the canopy will continue to increase in density, occluding sunlight. Lack of significant contrast between C, B, and BC sites between post- and 10-years post treatment indicates some positive impact of treatment with regards to rate of canopy closure relative to control. Analyses indicate that the active removal of adult trees does decrease overstory density immediately after selective logging occurs. However, 10 years without management results in a loss of the desired changes and a return towards baseline. The continued recruitment of trees into the adult size class between 2008 and 2018 is likely the result of fire-intolerant trees growing into the mid-story and recruiting into the canopy (Abrams, 1998) over a decade. The recruitment of trees into the mature size class seen at C, B, and BC sites over 10 years supports the idea that continued treatment is vital to the success of an oak regeneration-focused management plan. A study of species composition would be necessary to address this theory, but studies of the relative growth rates of oak (at low light; Farmer, 1975; Crow, 1992) and maple trees (Lorimer, 1981, 1984) seem to support this notion.

Understory Density

The goals of the initial management actions were to increase plant diversity and the density of seed producing, sun-loving plants in the understory. Prescribed fire was used to remove shade tolerant saplings and clear leaf litter to return nutrients to the soil and activate the seed bank while canopy thinning was intended to increase light availability and relax competition. Douglas (2010) initially found changes in understory community composition after treatment confirming the efficacy of the management plan. In contrast, comparison of pre- and post-treatment data using counts of plant density rather than a measure of diversity shows no change between pre and post treatment time points. I attribute this difference in observed effects to the difference in metric. Shifts in the community immediately following management likely manifest as changes in diversity rather than density and as such would only be detectible using a diversity-based metric.

Understory density measurements taken 10-years post treatment show a significant decrease compared to pre-treatment data, but do not differ from post-treatment values. High variance in 2008 likely accounts for lack of observed effect between post- and 10-years post-treatment. The highly variable understory densities post-treatment may be the result of non-equivalent burn intensities across sites (Brose & Van Lear, 1998) or variation in plant diversity that re-established after disturbance (Connell & Slayter, 1977). Without data collection designed to address these specific concerns it is impossible to draw a meaningful conclusion.

Analyses indicate a change in understory density over time due to the observed decrease in understory density in 2018 (Figure 6). The between-year differences responsible for the decrease observed between 2005 and 2018 are driven by Con and C sites. The significant decrease at control sites may indicate that the treatments administered were not responsible for the observed changes in understory density and that some external landscape scale factor was responsible. However, sites which included a burn treatment (B and BC) did

not experience the decrease in understory density, suggesting that treatment has had some lasting impact. The understory data supports the existing ideology that fire is vital to the structure of the understory in native Ozark forests (Peterson & Reich, 2001). Pittman and Krementz (2016) suggest that higher intensity burns can decrease large shrub abundance and increase small shrub abundance. My data may indicate that B and BC sites experienced more intense fire disturbance than expected. Insufficient fire return intervals can also result in dense mid-story and sapling layers rich in shade-tolerant species (Hutchinson, Long, et al., 2012; Hutchinson et al., 2005; Hutchinson, Yaussy, Long, Rebbeck, & Sutherland, 2012). There may be additional differences in the structure of the understory between time points that affects light availability or plant diversity; leading to the observed changes in understory density. Without current diversity data, accurate measures of light availability at all sites over time, or measures of fire intensity it is not possible to determine causation at this time. Anecdotally, the understory at all sites in 2018 was primarily composed of large saplings rather than smaller woody vegetation intermixed with grasses and seed producing shrubs. I postulate that the observed decrease in density is the result of competition. Shade tolerant woody vegetation has outcompeted the sun-loving grasses and shrubs present in 2008 resulting in lower understory densities composed of larger plants. Additional analyses of species composition in the system at present would be useful to further elucidate the lasting effects of treatment on understory community 10-years after treatment.

Small Mammal Density

The reintroduction of fire and the application of selective logging practices are known to increase small mammal abundance (Zwolak, 2009). Changes to the plant community affected through the reintroduction of fire and opening of the canopy were expected to increase the food availability for mice in the genus *Peromyscus*, resulting in an increase in population. Douglas (2010) initially estimated abundance in the spring and fall from 2005 – 2009 and found that

combination BC treatments effectively elevated mammal abundance post management. However, the effect initially reported is not observable using the relative density method followed here. The difference in observation is likely due to the difference in analyses and the aggregation of large variance terms in the calculation of relative densities.

Significant differences in *Peromyscus* density at control sites made direct comparisons of raw data unreliable. Douglas (2010) also noted the high variability in small mammal densities at control sites across time. Variance was primarily attributed to a large ice storm that affected the entire field site in January of 2009, immediately prior to the collection of the 2009 data set. The storm blanketed the site in 1-2 inches of ice, resulting in extensive tree fall that opened the canopy. Such a large disturbance event may not have affected all sites equally and likely explains a majority of the variation observed at control sites between 2005 and 2009.

The relative densities presented (Figure 7) provide the most conservative estimates of the effects that each treatment type may have over time. However, the patterns seen in the raw data (Table 3) agree with the work of many others and merit further discussion. The general patterns observed were: 1.) treatments involving selective logging (C and BC) increased small mammal abundance in the 1-2 years immediately following disturbance; 2.) burn only treatments (B) did not affect small mammal density; and 3.) 2017 densities were lower at all sites when compared to 2009 and did not differ significantly from the 2017 control.

The positive relationship between selective cutting and *Peromyscus* abundance has been widely reported (Fantz and Renken, 2005; Fisher and Wilkinson, 2005; Zwolak, 2009; Martell, 1983; Fuller et al., 2004). Although, most of these studies reported the highest densities between 1 and 5 years following disturbance. Martell (1983) and Zwolak (2009) also reported that the positive effects did not persist beyond a certain time span (Martell: effects dissipate after 4-23 years, Zwolak: no more than 10 years). Other studies (Kirkland, 1990; review) have reported a mixture of positive and negative impact of cutting on *Peromyscus*

density, but the nature of the relationship does seem to depend on the habitat type and intensity of disturbance. At my study site, it appears that C and BC treatments showed an increase in densities 2 years after the application of a selective cut, but that increase is no longer evident after 10 years.

Small mammal abundance and its response to the reintroduction of fire in fire dependent ecosystems seems to be more complicated than the relationship between *Peromyscus* abundance and selective logging. Cook (1959) reported that *Peromyscus* populations collapse immediately following a burn, but rebound several years after once seed production by ground level plants had resumed. Fisher and Wilkinson (2005) noted a similar pattern, but saw a continued increase in abundance correlated to increasing stand age without an initial period of population collapse. It has also been argued (Zwolak, 2009; meta-analysis) that the magnitude of a disturbance rather than the type determines the impact it has on the small mammal community. However, Zwolak (2009) claims that an intense wildfire typically increases small mammal abundance, while low intensity burns may have little to no effect. Without detailed data regarding the intensity of burning experienced at each of my sites, it is impossible to comment on this. It is unclear whether the positive impacts of fire would be evident in the year immediately following a burn, but my data suggest that no effects persist after 10 years.

Correcting for variation in the densities at control sites over time revealed one significant interaction; differences in density at BC sites relative to their controls between 2009 and 2017. However, density estimates for 2009 were the highest observed, while BC densities in 2017 were among the lowest (Figure 7). It is unsurprising that the comparison of these two extremes revealed a significant contrast. Detection of this treatment-related effect over time suggests that the high variance in my data may have confounded the detection of other significant treatment effects.

My data support Douglas' (2010) initial suggestion that BC treatments were the most effective in increasing small mammal abundance. The significant decrease in density seen between BC sites in 2009 and 2017 also suggests that small mammal density has decreased over time as a direct result of lack of treatment. Increased accuracy of density estimates may offer more insight into the effects that treatment has on small mammal populations. In the future, more intensive sampling may be necessary to better detect any treatment effects on small mammal densities in the Ozarks.

Light Penetration

I was unable to analyze light penetration values across time due to a lack of previous sampling. Using a chi-square contingency table I tested for differences due to treatment type in 2018. As a whole, the data do not conform to the expected distribution, indicating some difference between treatment types. The difference between individual treatment types and the expected distribution varies by type. B and BC treatments had greater light availability than expected, while Con and C treatments had lower availability. Differences in light availability may be the result of some persistent, community level impacts 10 years after treatment. However, without comparable data from alternate time points I cannot say that differences in treatment type are the cause of the observed difference in light distribution at the forest floor. Inconsistent effects of selective cutting between C and BC sites suggests that selective logging may not be the cause of the current differences in light environment. Conversely, an increase of light availability at B sites in combination with BC data suggest that the brief reintroduction of fire may be responsible for the differences among treatment types observed today. Future studies should take care to monitor light availability directly before, during, and after treatment to determine the relative efficacy of each treatment type on the light availability at the forest floor.

Conclusion

The goal of my study was to assess the long term efficacy of a habitat management plan implemented 10 years ago in the Ozark Highlands. Specifically, I sought to measure a set of variables correlated to the regeneration of Oak in an Ozark Oak-Hickory forest. The desired effects of management were to increase light availability, understory diversity, and small mammal density. Measurements taken directly after management efforts indicate the efficacy of controlled burns and selective logging actions. Controlled burns increased diversity of the understory community while selective cuts opened up space for new growth and decreased competition for light. Combination BC treatments effectively increased the population of mice in the *Peromyscus* genus.

After 10 years, the canopy opening effects of selective logging are no longer observable at any site. Density of understory plants has decreased to below pre-treatment values and anecdotally, diversity has shifted away from the desired understory of sun-loving grasses and herbaceous growth. Small mammal density has returned to pre-treatment levels or below at all sites. I did find a difference in light availability between treatment types, however, lack of comparable data obscures the exact relationship between treatment type and light penetration to the forest floor. Interestingly, B and BC sites had both increased light availability and a greater density of understory plants when compared to Con and C sites. The correlation between light penetration and understory density is not unexpected, but without more detailed data from previous time points it is not possible to determine whether this interaction is the result of the management actions taken in 2007 and 2008.

The lack of persistent community level effects after 10 years witnessed in my study were not unexpected given previous research into the historic burn regimes experienced by Ozark forests. Prior to extensive human settlement in 1820, Oak-Hickory forests of the Ozark Plateau were effected by fire an average of once every 7.7 years (Stambaugh & Guyette, 2006). During

this time period, burn frequency was dependent upon the frequency of ignition events (R P Guyette et al., 2006). Now, in the age of modern timber management (1921-2001) stands are often burned once every 4 years. Regular burn intervals assure that fire frequency and intensity is dictated by fuel loads and managers rather than stochastic ignition events. It has also been suggested that the frequency of burn events and the timing of burns in relation to the life cycle of the Oak tree influences the community level impacts that managers can expect following prescribed burns in an Oak-Hickory forest (Arthur et al, 2012). It is unsurprising that infrequent burning (< 1 fire per 7.7 years) has failed to produce a more diverse, open canopy system more similar to the pre-settlement Oak-Hickory forests that served as the impetus for the initial habitat manipulation in this Ozark system.

In the future, it may be fruitful to collect more long-term data on plant diversity and light availability at the forest floor whenever Oak restoration efforts are carried out. At present, my data suggest that in this study system, management actions should be implemented more frequently than once every 10 years to have a positive long-term impact on community structure and composition.

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Figures and Tables



Figure 1: Topographical Map of the MMCWMA/BHNA/ONSC site with borders of treatments and associated labels denoted. Con = Control, Burn = Prescribed Burn, Cut = Selective Logging, BC = Combination Selective Logging and Prescribed Burn.

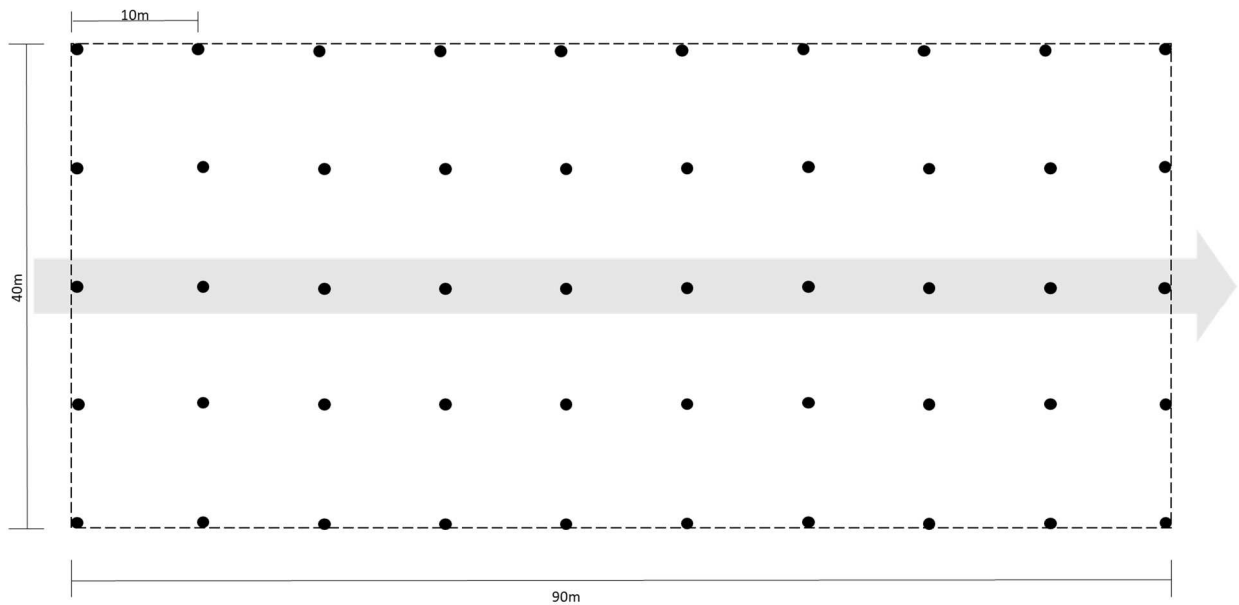


Figure 2: Grid array for mammal sampling. Black Dots represent trap locations, grey arrow indicates light penetration sampling vector.

Table 1: Timing chart of mammal sampling sessions. X denotes sampling, groupings of sites are static across season. Con=Control, Cut=controlled cut only, Burn=controlled burn only, BC=Combination Burn and Cut.

| Dates \ Site | April 6-9 | April 12-15 | June 10-13 | June 17-20 |
|--------------|-----------|-------------|------------|------------|
| Con1 | X | | | |
| Con2 | | | | X |
| Con3 | | | X | |
| Cut1 | X | | | |
| Cut2 | | | | X |
| Cut3 | | | | X |
| Burn1 | | | X | |
| Burn2 | | X | | |
| Burn3 | | | X | |
| BC1 | X | | | |
| BC2 | | X | | |
| BC3 | | X | | |

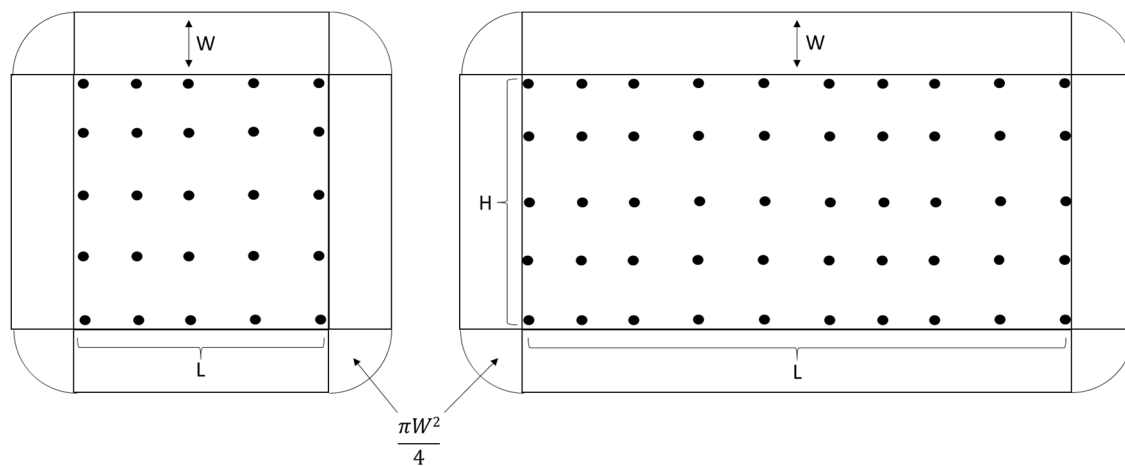


Figure 3: Diagram showing mammal trapping grid layout and area. The central part of each figure is the sampling grid, dots denote trap locations. The boundary strip of width $W=15.9\text{m}$ denotes the “effective trapping area”, the area inhabited by animals likely to be captured in traps. The square grid (left) was used in 2005 and 2009. All four sides are 40m long. The rectangular grid (right) was used in 2017 and had a height, $H=40\text{m}$, and a length, $L=90\text{m}$. Each corner is a quarter of a circle with radius $=W$.

Table 2: Chi-Square Row by Column (RxC) Contingency Table of Light Penetration in 2018. Each treatment type is shown separately, observation sums are calculated and displayed in "sum" column and rows. "Sum (Obs)" denotes the sum of all observations in both the Observed and Expected columns (They are the same), while "Sum(χ^2)" denotes the sum of chi-square terms. Sums of observed values were used to calculate an expected distribution used for chi-square analysis. Individual Chi-Square terms were summed to give a score for the analysis, $\chi^2 = 50.2$. Values that accounted for $\geq 25\%$ of a single treatment types' component χ^2 value are denoted with an asterisk, although the nature of these larger scores varies depending on whether the observed value is greater than or less than the expected value.

| | Control | | | Cut | | | Burn | | | Burn and Cut | | | Sum |
|------------------|----------|----------|----------|----------|----------|----------|----------|----------|----------|--------------|----------|----------|------|
| | Observed | Expected | χ^2 | Observed | Expected | χ^2 | Observed | Expected | χ^2 | Observed | Expected | χ^2 | |
| 0-20% | 183 | 159.25 | 3.5* | 168 | 159.25 | 0.48 | 147 | 159.25 | 0.94 | 139 | 159.25 | 2.57 | 637 |
| 21-40% | 26 | 31.75 | 1.04 | 39 | 31.75 | 1.66 | 33 | 31.75 | 0.05 | 29 | 31.75 | 0.24 | 127 |
| 41-60% | 36 | 30.75 | 0.90 | 35 | 30.75 | 0.59 | 25 | 30.75 | 1.08 | 27 | 30.75 | 0.46 | 123 |
| 61-80% | 14 | 24.00 | 4.2* | 25 | 24.00 | 0.04 | 35 | 24.00 | 5.0* | 22 | 24.00 | 0.17 | 96 |
| 81-100% | 41 | 54.25 | 3.2* | 33 | 54.25 | 8.3* | 60 | 54.25 | 0.61 | 83 | 54.25 | 15.2* | 217 |
| Sum (Obs) | 300 | | - | 300 | | - | 300 | | - | 300 | | - | 1200 |
| Sum (χ^2) | - | | 12.8 | - | | 11.1 | - | | 7.6 | - | | 18.7 | 50.2 |

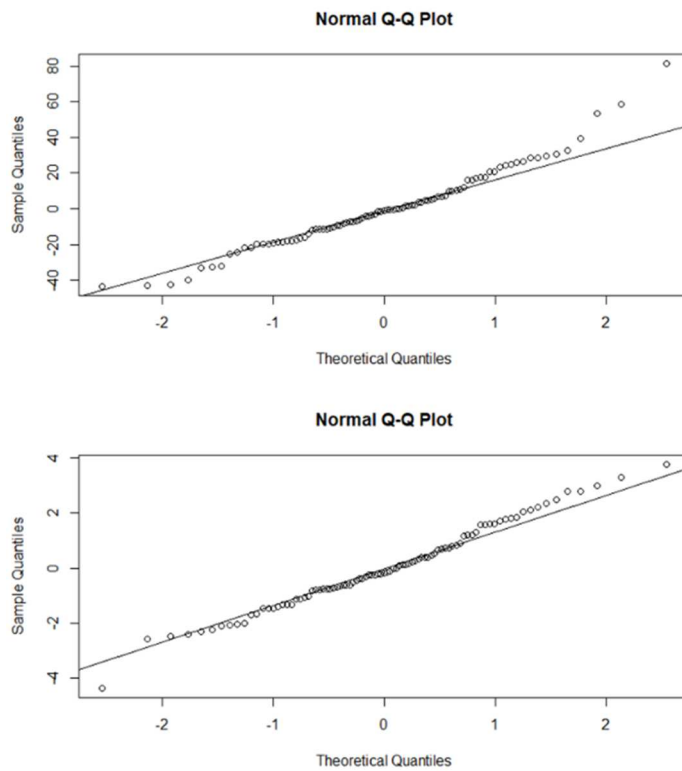


Figure 4: Q-Q plot comparing actual residuals of the understory density linear model to the theoretical residuals of a normal distribution. Raw data (top) shows a right skew, Square root transformed data (bottom) shows a better fit to a normal distribution.

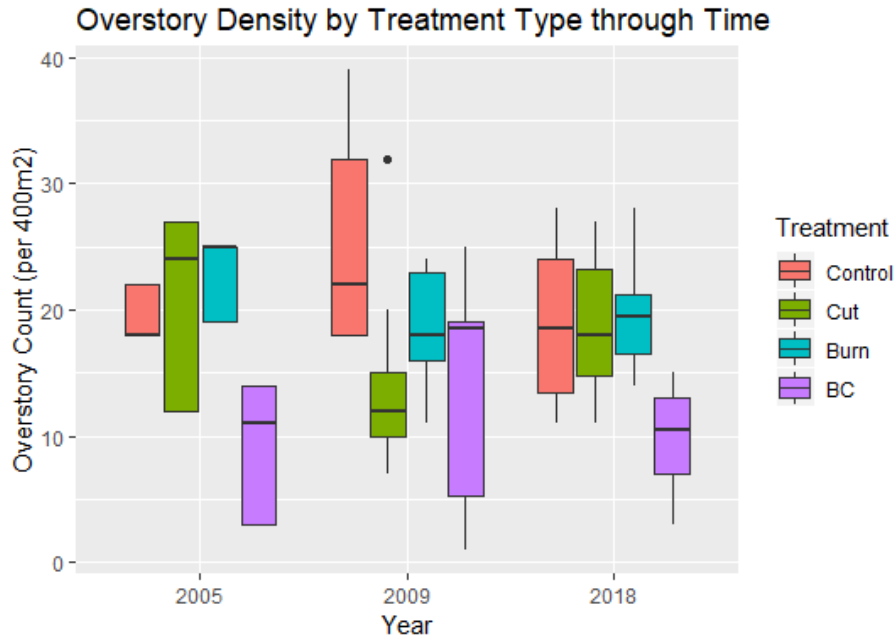


Figure 5: Overstory density raw data by treatment across time. 2005 data is based on limited data sets, resulting in skewed medians. Control =No Intervention, Burn = Prescribed Burn, Cut = Selective Logging, BC = Combination Selective Logging and Prescribed Burn.

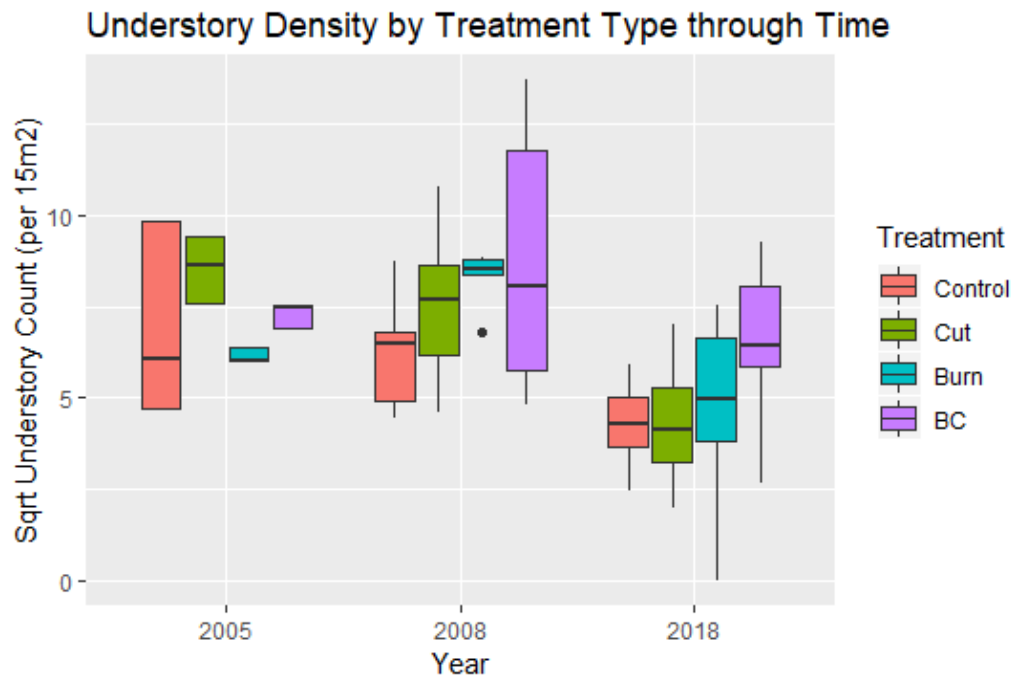


Figure 6: Understory density by treatment through time. Note that data is square root transformed to account for right skewed linear model residuals. Raw values range from 0 to 188 stems per 15m². Control =No Intervention, Burn = Prescribed Burn, Cut = Selective Logging, BC = Combination Selective Logging and Prescribed Burn.

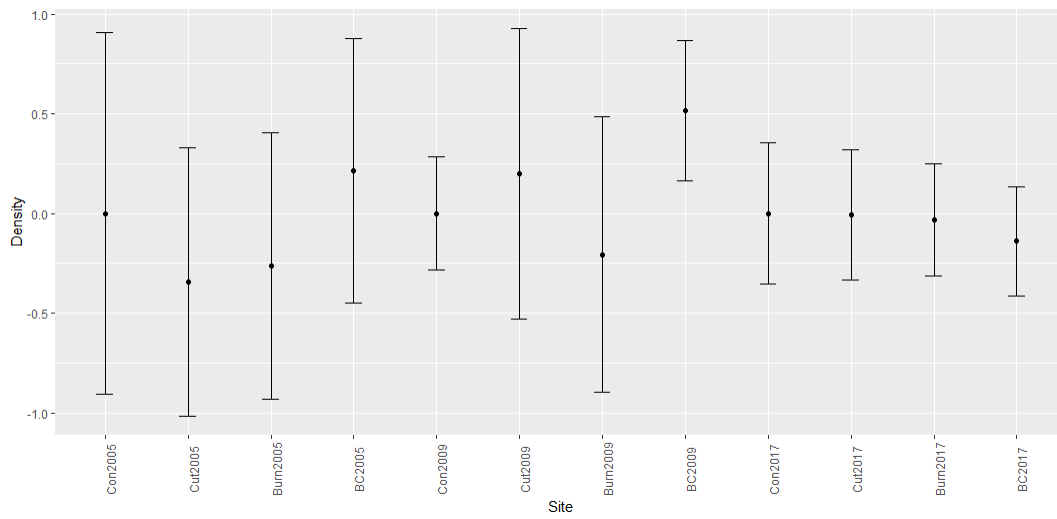


Figure 7: Relative Densities of Peromyscus at each treatment site relative to control for each year. Error bars represent a 95% Confidence Interval. Control densities are always an estimated relative density of 0, while point estimates for each treatment in that year reflect an increase or decrease in density relative to that control. Note the only non-overlapping CI's; BC2009 and BC2017

Table 3: Density estimates, associated variance approximations, and 95% CI spreads for small mammals in all years broken down by treatment types. These estimates have not been corrected for the significant variation in densities at control sites over time.

| Year | Site type | Density (per ha) | Variance | 95% CI |
|------|-----------|------------------|----------|-------------|
| 2005 | Control | 7.4 | 5.9 | 4.7 - 10.2 |
| | Burn | 4 | 0.14 | 3.6 - 4.5 |
| | Cut | 3.4 | 0.12 | 3.0 - 3.8 |
| | BC | 12.1 | 1.15 | 10.9 - 13.4 |
| 2009 | Control | 14.1 | 2.12 | 12.5 - 15.8 |
| | Burn | 8.8 | 8.77 | 5.4 - 12.1 |
| | Cut | 22.3 | 63.28 | 13.3 - 31.3 |
| | BC | 46.2 | 46.13 | 38.5 - 53.9 |
| 2017 | Control | 3.8 | 0.23 | 3.2 - 4.3 |
| | Burn | 3.5 | 0.05 | 3.3 - 3.8 |
| | Cut | 3.7 | 0.16 | 3.3 - 4.2 |
| | BC | 2.7 | 0.03 | 2.5 - 2.9 |

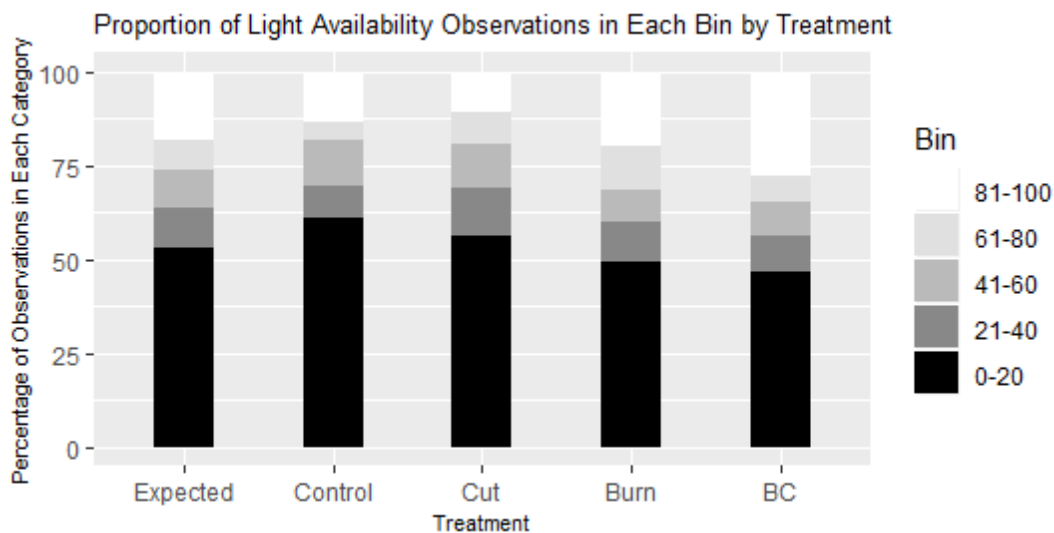


Figure 8: Light availability by treatment. "Expected" reflects the distribution used to calculate the chi-square test statistic, Control =No Intervention, Burn = Prescribed Burn, Cut = Selective Logging, BC = Combination Selective Logging and Prescribed Burn.