ABSTRACT

RAYBUCK, AMY LYNN. Short-term Response of Small Mammals and *Plethodon* Salamanders Following Oak Regeneration Silviculture Treatments. (Under the direction of Christopher E. Moorman and Christopher S. DePerno).

Upland, mixed-oak forests in the eastern US have experienced widespread oak regeneration failure, largely due to cessation of anthropogenic disturbance. Silvicultural practices used to promote advance oak regeneration may affect ground-dwelling wildlife. From May to August 2008 (pre-treatment), 2010, and 2011 (post-treatment), we trapped small mammals and *Plethodon* salamanders to assess changes to populations following three silvicultural practices (prescribed burn, midstory herbicide application, and shelterwood harvest) used to promote oak regeneration. We trapped in five replicates of the oak regeneration practices and a control using Sherman live traps (2008 and 2010) and drift fence arrays (2008, 2010, and 2011). We evaluated changes in species richness of small mammals, abundance of peromyscids, and relative abundance of masked shrews (*Sorex cinereus*), smoky shrews (*S. fumeus*), northern short-tailed shrews (*Blarina brevicauda*), Southern Appalachian salamanders (*Plethodon teyahalee*), and Southern gray-cheeked salamanders (*P. metcalfi*). Additionally, for *Plethodon*, we measured changes in juvenile proportion of captures and body condition indices. From pre- to post-treatment, changes in species richness, peromyscid abundance, and relative abundance of shrews and *Plethodon* salamanders were similar among treatments. Similarly, we did not detect any difference among treatments for changes in juvenile proportion of captures or body condition indices for *Plethodon*. However, there was a year effect for masked shrews, smoky shrews, northern short-tailed shrews, and Southern Appalachian salamanders with a greater increase in

captures in 2011 (i.e., second year post-treatment) than in 2010 (i.e., first year post-treatment). Our research indicates that in the short-term, small mammals and *Plethodon* salamanders can tolerate a wide range of forest disturbance following oak regeneration treatments. Lack of short-term response in herbicide and prescribed burn treatments was likely because of minor or transitory (i.e., dissipated by second year post-treatment) changes to forest structure. In shelterwood harvests, the initial impacts of reduced canopy and leaf litter cover were likely mitigated by rapid understory growth and the presence of residual logging debris. Moreover, high levels of precipitation in 2011 may have compensated for moisture reductions following prescribed burns and shelterwood harvests, thereby increasing captures in the second year post-treatment. Delayed treatment effects including post-herbicide midstory dieback and post-treatment sprouting in shelterwood harvests and future activities associated with oak regeneration systems (e.g., prescribed burns following shelterwood harvests) may have different effects on ground-dwelling wildlife. Therefore, long-term monitoring is warranted for small mammal and *Plethodon* salamander populations (e.g., > 3 years post-treatment).

Short-term Response of Small Mammals and *Plethodon* Salamanders Following
Oak Regeneration Silviculture Treatments

by
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DEDICATION

To my loving parents, Eugene and Joyce Raybuck

BIOGRAPHY

Amy Lynn Raybuck was born in Sewickley, Pennsylvania and grew up in New Sewickley Township, 45 minutes north of Pittsburgh. She is the daughter of Eugene and Joyce Raybuck and has two older brothers, Tom and Frank Raybuck. She graduated from Freedom Area High School in May 2003 and moved to Raleigh, North Carolina to attend N.C. State University for Fisheries and Wildlife Sciences. As an undergraduate student, Amy worked on two internships, the first in 2004 studying Amazonian parrots on Abaco Island in the Bahamas and the second in 2006 inventorying herpetofauna in Bull Neck Swamp in eastern North Carolina. In 2007, Amy worked as a teaching assistant at the N.C. State Fisheries and Wildlife Summer Camp teaching undergraduate students basic field techniques in wildlife biology. During her undergraduate career, Amy was on the Dean's List almost every semester and was awarded numerous scholarships including the Hofmann Forest Academic Scholarship, the N.C. Wildlife Federation Rocky River Trout Unlimited Scholarship, the T. Clyde and Sally Watts Auman Endowed Scholarship, the Thomas L. Quay Scholarship, and the NCSU Department of Forestry and Environmental Resources Academic Scholarship. In May 2008, Amy graduated with a Bachelors of Science degree in Fisheries and Wildlife Science and a Bachelors of Science degree in Zoology. Just days later, she began a research assistantship in the Fisheries, Wildlife, and Conservation Biology Graduate Program at N.C. State. In her graduate career, Amy was awarded additional scholarships and grants including the N.C. Wildlife Federation Grant, the NCSU Forestry and Environmental Faculty Fellowship for Excellence in Graduate Education and Service, the Department of Forestry Scholarship, and the College of Natural Resources Scholarship. In 2010, Amy was awarded a teaching assistantship with the Biology Department and taught two semesters of introductory laboratory biology to college freshman. Amy is active with N.C. Partners in Amphibian and Reptile Conservation and The Wildlife Society, presenting her research at national conventions in Monterey, California and Snowbird, Utah in 2009 and 2010, respectively. Additionally, she earned her Associate Wildlife Biologist certification in 2010 from The Wildlife Society. She hopes to work with the U.S. Fish and Wildlife Service following graduation in December 2011.

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Abstract

Upland, mixed-oak forests in the eastern US have experienced widespread oak regeneration failure, largely due to cessation of anthropogenic disturbance. Silvicultural practices used to promote advance oak regeneration may affect ground-dwelling mammals. From May to August 2008 (pre-treatment), 2010 (first year post-treatment), and 2011 (second year post-treatment), we trapped small mammals to assess changes in species richness and abundance following three silvicultural practices (prescribed burns, midstory herbicide applications, and shelterwood harvests) used to promote oak regeneration. We trapped small mammals in five replicates of the oak regeneration practices and a control using Sherman live traps (2008 and 2010) and drift fences (2008, 2010, and 2011). From pre- to post-treatment, we evaluated the change in estimated peromyscid abundance and relative abundance of masked shrews (*Sorex cinereus*), smoky shrews (*S. fumeus*), and northern short-tailed shrews (*Blarina brevicauda*). Additionally, we evaluated the change in species richness across treatments for both sampling techniques. For all measures analyzed (i.e., species richness, peromyscid abundance, and relative abundance of shrews), we did not detect a change from pre- to post-treatment among treatments following oak regeneration practices. However, there was a year effect for masked shrews ($P < 0.001$), smoky shrews ($P < 0.01$), and northern short-tailed shrews ($P < 0.001$) with a greater increase in captures in 2011 (i.e., second year post-treatment) than in 2010 (i.e., one year post-treatment). Our research indicates that in the short-term, small mammals (e.g., mice, voles, and shrews) can tolerate a wide range of forest disturbance following oak regeneration treatments. However, delayed treatment effects including post-herbicide midstory dieback and combinations of treatments (e.g., prescribed burns following shelterwood harvests) may compound effects on small mammal populations and warrant long-term research (>2 years post-treatment).

Keywords: herbicide, mice, oak regeneration, prescribed fire, shelterwood harvest, shrews, southern Appalachians, voles

1. Introduction

Mixed-oak (*Quercus* spp.) forest occupies over 50% of the forested land base in the Central Hardwood Region of the United States and is the dominant forest type in the Southern Appalachians (Sharitz *et al.*, 1992; Johnson *et al.*, 2002). Ecologically, mixed-oak forests are among the most productive terrestrial ecosystems providing valuable wildlife habitat and high biodiversity (Rodewald, 2003). However, mixed-oak forests are increasingly threatened by oak decline due to widespread regeneration failure (Oak *et al.*, 2004; Aldrich *et al.*, 2005).

Historically, disturbance events, including low-intensity surface fires, timber harvesting, and land clearing for agriculture promoted conditions conducive to oak establishment, development, and recruitment (Abrams, 1992; Sharitz *et al.*, 1992; Lorimer, 1993). Approximately 12,000 years ago, Native Americans used fire to concentrate game species, open forests, improve forage, and clear land for agriculture and settlement, all of which shaped the composition of hardwood forests (Komarek, 1974; Loftis and McGee, 1993; Delcourt and Delcourt, 1997). Later, English colonists burned and cleared land (Sharitz *et al.*, 1992) which reduced overstory and understory density and promoted successful oak regeneration (Signell *et al.*, 2005). Frequent disturbance discouraged fire-intolerant and shade-tolerant species that compete with oaks such as red maple (*Acer rubrum*), yellow-poplar (*Liriodendron tulipifera*), sourwood (*Oxydendrum arboreum*), rhododendron (*Rhododendron* spp.), and mountain laurel (*Kalmia* spp.).

During the last century, cessation of frequent disturbance and implementation of fire suppression policies have gradually changed oak forest structure and composition (Lorimer, 1993; Spetich and Parker, 1998; Abrams, 2005). Fire-intolerant species have become more prevalent and forests are dense with shade-tolerant shrubs and trees, especially on mesic upland sites (Clinton, 1989; Brose *et al.*, 2001). This shift to mesophytic forest is undesirable due to the ecological and economic benefits of oak-dominated forests (Guyette *et al.*, 2004; Nowacki and Abrams, 2008). Therefore, various silvicultural practices have been suggested to create conditions favorable for oak regeneration.

Oak regeneration practices, including prescribed fire and canopy reduction, result in a wide range of habitat changes to forests with a primary goal of increasing light to the forest floor to encourage oak sapling germination. Prescribed fire may create conditions conducive to germination by preparing a fertile seedbed and increasing light by removing thin-barked shrubs and trees in the midstory and herbaceous and woody vegetation in the understory (Van Lear and Watt, 1993; Wang *et al.*, 2005). Following a prescribed burn, leaf litter decreases, with up to six times more litter documented in unburned areas than in recently burned sites (Kirkland Jr. *et al.*, 1996; Greenberg *et al.*, 2006; Greenberg *et al.*, 2007; Waldrop *et al.*, 2007). Even-aged regeneration methods (e.g., shelterwood harvests) can produce conditions similar to medium to large-scale natural disturbance with reduced canopy cover and leaf litter, rapid re-sprouting of vegetation, and potential increases in downed woody debris (Brose *et al.*, 1999). Additionally, removal of the canopy creates a high-light environment, resulting in higher forest floor temperatures and consequently reducing soil moisture (Geiger, 1965; Chen *et al.*, 1999).

Small mammals (e.g., mice, voles, and shrews) can be affected by changes in forest characteristics following oak regeneration practices, largely because of their dependency on coarse woody debris (CWD), forest floor humic mat, and leaf litter for cover, nesting, foraging, and thermoregulation (Lanham and Guynn, 1996; Loeb, 1996). For example, changes to availability of foods such as acorns, seeds, arthropods, or vegetation may influence small mammal populations (Planz and Kirkland, 1992; McCracken *et al.*, 1999; Menzel *et al.*, 1999; Kaminski *et al.*, 2007). Although previous studies have not documented changes in shrew populations following prescribed fire or low-intensity timber harvest (Ford *et al.*, 1999; Keyser *et al.*, 2001; Ford *et al.*, 2002; Homyack *et al.*, 2005; Greenberg *et al.*, 2007; Zwolak, 2009), shrews may be susceptible to changes in CWD (Brannon, 2000; Ford *et al.*, 2000; Ford and Rodrigue, 2001; McCay and Komoroski, 2004; Kaminski *et al.*, 2007). Additionally, reductions in leaf litter may negatively impact shrew populations, especially following high-intensity disturbance (Ford *et al.*, 1997; Matthews *et al.*, 2009).

Because silvicultural practices are increasingly used to restore upland oak forests in the eastern United States, it is important to determine potential impacts on small mammals, which are an integral part of forest ecosystems. Small mammals increase functional diversity in ecosystems, recycle nutrients by processing vegetation, disperse seeds and fungal spores, and are a substantial prey base for raptors, reptiles, and other mammals (Cork and Kenagy, 1989; Carey and Johnson, 1995; Fedriani *et al.*, 2000; Schnurr *et al.*, 2004; Clotfelter *et al.*, 2007). As a result, small mammals are identified as potential indicators for sustainable forest management (Carey and Harrington, 2001).

Our objective was to determine short-term changes to small mammal populations following three oak regeneration practices. Using five 5-ha replicates of the oak regeneration treatments (prescribed fire, midstory herbicide application, and shelterwood harvest) and a control, we were able to compare response of populations among treatments using pre-treatment (baseline) and post-treatment data.

1.1 Study Area

Our study was conducted on Cold Mountain Game Land (CMGL) in Haywood County in western North Carolina. CMGL was managed by the North Carolina Wildlife Resources Commission primarily for diverse wildlife habitat. Located along the escarpment of the Blue Ridge Physiographic Province, CMGL encompassed ~5900 ha of second growth, upland mixed-oak forests. Elevations ranged from approximately 1,100-1,350 m, and terrain was mountainous with gentle to steep slopes. Oaks, hickories (*Carya* spp.), red maple, sugar maple (*Acer saccharum*), black cherry (*Prunus serotina*), and yellow-poplar were the predominant overstory trees. Species composition in the midstory consisted primarily of shade-tolerant species including sourwood, blackgum *(Nyssa sylvatica*), and red maple. Site index (base age 50) of oak ranged from 15 m on the xeric, poor quality sites to 27 m on mesic, high quality sites.

2. Material and Methods

Twenty 5-ha treatment units (3 treatments plus 1 control, 5 replicates of each) were located within CMGL. Unit locations were established in sites that met our selection criteria described below. Treatments (prescribed fire, midstory herbicide, and shelterwood harvest) were randomly assigned to each unit resulting in a completely randomized design (CRD). All units were separated by a >10-m buffer and contained mature (>70 years old), fully stocked, closed-canopied stands where oaks comprised at least 10% of the overstory tree basal area (≥ 25.0-cm dbh). We selected stands that contained >1,000 oak seedlings/ha, few ericaceous shrubs, ~2 m^2/ha of basal area (BA) beneath the main canopy, and no substantial disturbance within the last 15-20 years (Keyser *et al.*, 2008).

2.1 Treatments

Treatments were designed to evaluate the effectiveness of three oak regeneration practices: (1) 3 prescribed burns at ~4-year intervals, (2) midstory removal using herbicide competition control with re-application after ~3 years, and (3) shelterwood harvest with 30-40% BA retention followed by a prescribed fire after ~3 years. All three practices will eventually be followed by overstory removal ~11 years following initial treatments. This study encompassed one pre-treatment year and two post-treatment years, so we evaluated the response of small mammals to the first prescribed burn, initial midstory herbicide application, shelterwood harvest prior to burning, and controls.

Two of the five prescribed burn units were burned in April 2009 because weather and road conditions did not permit burning of all five units; the remaining three units were burned in April 2010. Thus, the prescribed burn treatment was separated into two treatments because of potential ecological differences related to time since burn.

In late summer 2008, prior to leaf fall, midstory trees (≥5.0 cm and <25.0 cm dbh) were treated with herbicide using the hack-and-squirt method where ~1 ml of diluted Garlon 3A solution was sprayed into a waist-high incision of each midstory tree marked for removal (Loftis, 1990). The goal of the herbicide treatment was to reduce total BA from below by 25-30% without creating new canopy gaps, primarily to increase photosynthetically active radiation (PAR) on the forest floor to promote oak seedling growth and successful recruitment into the canopy (Loftis, 1990).

From winter 2009 to early summer 2010, the shelterwood harvest was implemented with a goal of leaving approximately 30-40% of the original stand BA and enhancing light conditions on the forest floor (Brose *et al.*, 1999). The majority of leave trees were dominant or codominant oak trees. Most slash was left on-site.

During this study, no silvicultural manipulation occurred in the control plots.

2.2 Small Mammal Sampling

We sampled small mammals in all 20 units during mid-May to mid August of 2008 (pre-treatment) and 2011 (second year post-treatment). In 2010 (first year post-treatment), we sampled 19 units because a shelterwood unit was not harvested until mid-summer 2010. In 2008 and 2010, we used Sherman live traps and drift fence arrays for sampling, and in 2011 we only used drift fences.

In each treatment unit, we placed 60 Sherman live traps (7.7 X 9.0 X 23.3 cm) at 10-m intervals, in a rectangular 60- X 100-m trapping grid. Grids were centered approximately mid-slope of each unit and all traps were >10 m from treatment boundaries. We baited traps with raw oatmeal and supplied cotton balls for bedding. In 2008 and 2010, we trapped one replicate unit of each treatment and control simultaneously during each of five trapping periods. Traps were open continuously for seven nights and checked each morning. We ear-tagged all new captures, excluding shrews, with an individually numbered tag (size-1 Monel; National Band and Tag Co., Newport KY) and released individuals at the capture site (Greenberg *et al.*, 2006).

In 2008, we established six randomly oriented single-arm drift fence arrays within all 20 treatment and control units. Three of the drift fences were installed at a lower slope site (defined as the lower one-third of each unit) and three at an upper slope site (the upper one-third of each unit) (Greenberg and Waldrop, 2008). In two of the treatment replicates (one herbicide and one control), we were unable to establish an upper site due to steep and rocky terrain. By 2010, a fourth fence was installed at each lower and upper location of each unit to increase sampling effort.

Single-arm drift fences were >10 m apart and constructed of 7.6-m sections of aluminum flashing with a 19-L bucket buried at each end, flush with the ground. We placed a moist sponge in each bucket to provide moisture for captured animals. The drift fences were designed to capture reptiles and amphibians, but also captured many shrews. Drift fence arrays were continuously open from mid-May to mid-August. Animal handling methods followed guidelines approved by the American Society of Mammalogists (Gannon and Sikes, 2007) and were approved by the North Carolina State University Institutional Animal Care and Use Committee (Approval number 08-035-O).

2.3 Habitat Data

We obtained elevation measurements at upper and lower slope sites with a portable GPS device; upper and lower elevation measurements were averaged per unit. We recorded overall aspect of each unit as a binary value: 0 = south- and west-facing aspects, 1 = north- and east-facing aspects. We measured canopy cover and CWD in 2008 and 2010 and measured ground cover in 2008, 2010, and 2011. We measured canopy cover with a spherical densiometer at each drift fence and averaged per unit (Lemmon, 1956). At each drift fence in all units, we measured ground cover and CWD along a 15-m randomly oriented transect line originating from the bucket furthest uphill on each fence. Variables measured were percent cover of bare ground, leaf litter, understory cover (i.e., plants < 0.9 m), and CWD (≥12-cm diameter). We recorded 'start' and 'stop' distance for each category along each transect (e.g., bare ground from 3.1 m to 3.3 m = 0.2 m bare ground cover) and then summed the total distance along each transect. For understory cover, the start and stop measurements were determined by the potential cover (e.g., shading) provided by each plant. Percent cover for each category in a unit was determined by dividing the total summed distance of the category by 90 m (six transects per unit). We used the average percent cover of all six transects to estimate percent cover within a given treatment unit. For each piece of CWD, we recorded total length, bark class, and amount of decay. Bark class was visually categorized from 1-5: 1 = recently dead with 100% of bark, 2 = >70% of bark, 3 = 40-69% of bark, 4 = 10-39% of bark, and 5 = <10% of bark. Amount of decay was visually categorized from 1-6: 1 = no decay visible, 2 = slight decay, 3 = moderate decay, 4 = slight fragmentation, 5 = heavy fragmentation, and 6 = completely disintegrated but still distinguishable as CWD.

2.4 Analyses

We compared the change in canopy cover and CWD characteristics from 2008 to 2010 among treatments using a one-way analysis of variance (ANOVA). When models were significant, we used Tukey's Studentized Range (HSD) test to determine significant differences among treatment means. Because ground cover was measured twice post-treatment (i.e., 2010 and 2011), we performed a repeated measures ANOVA. We used a mixed model procedure with a random effect (i.e., treatment per unit) and fixed effects of treatment and year with an interaction. If significant, we tested for differences among the treatments using Tukey's Studentized Range (HSD) test.

We estimated abundance of peromyscids from Sherman live traps using closed population capture-recapture in Program MARK (White and Burnham, 1999). The closed captures option allowed modeling of the initial capture probability (p) and the recapture probability (c) to estimate population size (N) (Otis *et al.*, 1978; White and Burnham, 1999).

We lumped peromyscids for analysis because deer mice (*Peromyscus maniculatus*) and white-footed mice *(P. leucopus*) occur sympatrically above 800 m in the Appalachian Mountains and are difficult to distinguish in the field because of similarities in appearance (Schnurr *et al.*, 2004). Population estimates were log-transformed for normality. Our response variable was the change in population estimates from 2008 to 2010 (pre- to post-treatment), which we compared among treatments using a mixed model consisting of a random effect (i.e., treatment per unit) and fixed effects [e.g., treatment and two covariates (aspect and elevation)] and the interactions between treatment and the covariates. We removed covariates and their interactions from the model when they were not statistically significant. If any variable was significant, we tested for differences among the treatments using Tukey's Studentized Range (HSD) test.

We calculated relative abundance of small mammals captured in the drift fence arrays as the number of animals captured per 100 trap nights. Slope position was not an important predictor of small mammal response, so to reduce complexity of the analysis we averaged the upper and lower sites of each unit, which were log-transformed for normality. Our response variable was the change in relative abundance from 2008-2010 and 2008-2011. We analyzed the change in relative abundance of three species of shrews [e.g., masked shrews (*Sorex cinereus*), smoky shrews (*S. fumeus*), and northern short-tailed shrews (*Blarina brevicauda*)] among treatments using a repeated measures ANOVA with a covariance structure of first order autoregression [AR (1)]. The mixed model consisted of a random effect (i.e., treatment per unit) and fixed effects [e.g., treatment, year, and two covariates (aspect and elevation)] and the interactions between year and treatment and covariates and treatment. We removed interactions and covariates from the model when not statistically significant. If significant, we tested for differences among the treatments and used Tukey's Studentized Range (HSD) test.

For both sampling techniques (i.e., Sherman live traps and drift fence arrays), we analyzed change in species richness from pre- to post-treatment. To determine species richness in each treatment, we summed the total number of small mammal species captured in each unit. For Sherman live traps, our response variable was the change in species richness from 2008 to 2010 (we did not use Sherman live traps in 2011). Therefore, our model was identical to the analysis outlined above for population estimates of peromyscids from Sherman live traps. For drift fence arrays, our response variable was the change in species richness from 2008 to 2010 or 2008 to 2011 and our model was identical to analysis outlined above for relative abundance of small mammals from drift fence arrays. All statistical tests were conducted in SAS (v. 9.1.3, SAS Institute, Cary, NC).

3. Results

3.1 Habitat

Elevation ($F_{4, 14}$ = 2.12, P = 0.12) and aspect ($F_{4, 31}$ = 1.96, P = 0.13) were similar among treatments. In the first year post-treatment (2010), all habitat variables, except understory cover and CWD length, differed among treatments (Table 1). Canopy cover in the shelterwood harvest declined 40% post-harvest but changed little in all other treatments. Bare ground increased in prescribed burns of 2010 and shelterwood harvests by 48% and 33%, respectively. Leaf litter cover decreased by 64%, 41%, and 38% in prescribed burns of 2009, shelterwood harvests, and prescribed burns of 2010, respectively. CWD cover in shelterwood harvests increased by 5% post-harvest but changed little in other treatments.

Following the shelterwood harvest, CWD had more bark and was less decayed than the other treatments (Table 1).

In the second year post-treatment (2011), changes in percent bare ground were similar among treatments as leaf litter levels recovered in prescribed burns and understory cover increased in shelterwood harvests (Table 1). Leaf litter declined 46% in shelterwood harvests, but recovered to pre-treatment levels in both prescribed burn treatments. Understory cover increased 58% and 51% in shelterwood harvests and prescribed burns of 2009, respectively, whereas other treatments changed less dramatically (Table 1).

3.2 Small Mammal Sampling

Small mammals captured in Sherman live traps included four species of mice, three species of shrews, two species of voles, southern flying squirrels (*Glaucomys volans*), and eastern chipmunks (*Tamias striatus*) (Table 2). From 2008 to 2010 (e.g., pre- to post-treatment), the change in species richness of small mammals captured in Sherman live traps was similar among treatments ($F_{4, 9}$ = 0.83, P = 0.54, Table 3).

Peromyscids were the primary species captured in Sherman live traps, composing 86% and 82% of the total mouse captures in 2008 and 2010, respectively. In 2008, we captured 178 peromyscids 367 times in 7,648 trap nights. In 2010, we captured 131 peromyscids 270 times in 7,353 trap nights (Table 2). From 2008 to 2010 (e.g., pre- to post-treatment), the change in population estimates of peromyscids was similar among treatments ($F_{4, 14}$ = 1.16, P = 0.37, Figure 1).

The change in species richness of small mammals captured in drift fence arrays was similar among treatments ($F_{4, 14}$ = 0.62, P = 0.65, Table 3). We captured seven species of shrews, four species of mice, and two species of voles in drift fence arrays in 2008, 2010, and

2011 (Table 4). In 2008, we captured 768 masked shrews, 98 smoky shrews, and 91 northern short-tailed shrews in 15,552 trap nights. In 2010, we captured 671 masked shrews, 69 smoky shrews, and 164 northern short-tailed shrews in 19,848 trap nights. In 2011, we captured 597 masked shrews, 76 smoky shrews, and 203 northern short-tailed shrews in 17,800 trap nights (Table 4). The masked shrew was the most abundant species captured, accounting for 58%, 56%, and 55% of the small mammal captures in 2008, 2010 and 2011, respectively.

From pre-treatment (2008) to post-treatment (2010 and 2011), the change in relative abundance of masked shrews ($F_{4, 14} = 0.70$, $P = 0.60$), smoky shrews ($F_{4, 14} = 0.49$, $P = 0.74$), and northern short-tailed shrews ($F_{4, 14} = 1.84$, $P = 0.18$) was similar among treatments (Figure 2). However, there was a year effect for each species: masked shrews ($F_{1, 18} = 68.28$, $P < 0.001$), smoky shrews ($F_{1, 18} = 14.32$, $P < 0.01$), and northern short-tailed shrews ($F_{1, 18} = 21.19$, $P < 0.001$) with a greater increase in captures in 2011 (second year post-treatment) than in 2010 (first year post-treatment).

4. Discussion

Lack of small mammal response to the oak regeneration treatments was likely because of the ability of these species to tolerate a wide range of forest conditions. Similarly, other studies showed peromyscid and shrew populations were not affected by prescribed fire and timber harvests (Ford *et al.*, 1999; Keyser *et al.*, 2001; Ford *et al.*, 2002; Hood *et al.*, 2002; Homyack *et al.*, 2005; Greenberg *et al.*, 2006; Stratton and Clatterbuck, 2007; Matthews *et al.*, 2009). In fact, peromyscids are described as generalists, living under a wide range of temperatures and moisture conditions following disturbance (Getz, 1961; Dueser and Shugart, 1978; Mitchell *et al.*, 1997; Brannon, 2005).

Lack of response to oak regeneration treatments may be attributed to minor habitat changes in the herbicide and prescribed burn treatments and retention of important environmental components in shelterwood harvests. Following the herbicide application, there were no significant changes to small mammal habitat including understory, leaf litter, or CWD cover. Increased bare ground cover following prescribed burns and shelterwood harvests disappeared by the second year post-treatment due to rapid recovery of leaf litter and increases in understory cover , which may have alleviated possible stresses on moisture-dependent shrews. Additionally, residual piles of logging slash in shelterwood harvests may have sustained populations by providing food, travel corridors, and protection from predators (Planz and Kirkland, 1992; Loeb, 1999; Menzel *et al.*, 1999).

Fluctuating precipitation levels during sampling years may have mitigated potential effects of oak regeneration treatments on small mammal populations. Compared to 2008, we had increased numbers of captures of all three shrew species analyzed during the second year post-treatment (2011) compared to 2010, when there was less rainfall. In fact, the average precipitation for Haywood County from May to August of 2011 was 26 inches higher than May to August of 2010 (National Weather Service, 2011). Thus, greater rainfall in 2011 may have compensated for possible reductions in moisture caused by shelterwood harvests or prescribed fire. Similarly, Ford et al. (2002) concluded capture frequency of shrews was more influenced by differences in weather conditions between years than by differences in hardwood forest conditions ranging in age from recently clearcut to >60 years old. Additionally, other studies have documented rainfall as an important predictor of captures and/or activity of small mammals, including peromyscids and shrews (Drickamer and Capone, 1977; McCay, 1996; Brannon, 2002; Greenberg and Miller, 2004).

Delayed effects of treatments on forest composition and structure may cause long-term changes in small mammal abundance. Although a single herbicide application did not affect canopy, understory, leaf litter, or CWD cover, habitat changes may become more evident with future midstory dieback and increased photosynthetically active radiation (PAR) to the forest floor. Additionally, individual oaks left as seed trees in the shelterwood harvests may produce more acorns given limited competition and ample crown space, which could indirectly lead to increased number of small mammals in these areas (McCracken *et al.*, 1999; Schnurr *et al.*, 2004).

Planned activities associated with the oak regeneration systems, including repeated prescribed fires, repeated herbicide applications, and prescribed burns following shelterwood harvests, may have additive effects on small mammals (Matthews et al., 2009). The repeated prescribed burning required to facilitate oak seedling establishment may compound treatment effects (e.g., reduced leaf litter or duff layer) and impact litter-dependent species such as invertebrates and shrews (Van Lear and Watt, 1993; Coleman and Rieske, 2006; Matthews *et al.*, 2009). Additionally, the combination of shelterwood harvests and prescribed fires could result in substantial changes to habitat conditions compared to either disturbance alone. For example, Matthews et al. (2009) caught 77% fewer southeastern shrews (*S. longirostris*) in fuel reduction treatments where the understory had been mechanically thinned followed by two prescribed burns (3 years apart) than in either treatment alone (Matthews *et al.*, 2009).

4.1 Conclusion

In the short-term, we detected no changes in mouse or shrew abundance or species richness following oak regeneration treatments. Lack of response was likely due to the ability of peromyscids and shrews to tolerate a wide range of forest conditions following treatment disturbance, rapid establishment of understory cover in shelterwood harvests and prescribed burns, and residual piles of logging slash in shelterwood harvests. Longer-term studies are imperative to determine response of small mammals to delayed treatment effects and additive effects from long-term oak regeneration systems such as repeated prescribed fires, prescribed burns following shelterwood harvests, and overstory removal.

We recommend forest managers integrate multiple objectives when making management decisions. For example, the efficacy of each oak regeneration treatment should be considered in conjunction with conservation of focal wildlife species. Additionally, managers should consider landscape context when making stand level decisions. If landscape is primarily mature mixed-oak forest, smaller areas designated to oak regeneration will have small impacts on late-successional wildlife populations.

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Table 1. Change in habitat variables (± SE) from 2008-2010 and 2008-2011 for oak regeneration treatments on Cold Mountain Game Land, NC: control (CONT), herbicide (HERB), shelterwood harvest (SW), prescribed burn 2010 (RX1), and prescribed burn 2009 (RX2). Different letters indicate significantly different values (P < 0.05).

	Treatment						
Habitat Variable	CONT	HERB	SW	RX1	RX2	*F*	P_{trt}
2008-2010							
Canopy cover (%)	3.4±1.1A	-2.9±2.4A	-39.8±6.1B	3.5±2.0A	2.1±2.5A	29.62	<0.001*
Bare ground (%)	1.3±0.7AB	-4.9±1.4A	33.1±5.0C	13.9±7.5B	48.0±10.9C	12.90	<0.001*
Leaf litter (%)	1.8±1.6A	-6.4±1.8A	-40.9±5.9BC	-37.6±5.9B	-63.8±1.6C	13.87	<0.001*
Understory cover (%)	-4.6±1.6	3.3±0.6	2.9±3.3	13.2±1.4	9.3±6.1	1.93	0.13
CWD (%)	0.18±0.2A	0.75±0.2AB	5.08±2.0B	-0.90±2.0A	-3.07±0.4A	5.41	0.01*
CWD Length	0.30±0.2	0.13±0.1	-0.92±0.2	-0.97±1.0	-0.53±0.8	2.54	0.09
CWD Bark Class (1-5)	0.35±0.1A	0.55±0.2A	-1.43±0.4B	0.01±0.1A	-0.4±0.1AB	9.36	0.001*
CWD Decay Class (1-6)	0.33±0.2A	-0.6±0.2BC	-0.93±0.2C	-0.5±0.1AC	-0.71±0.2C	7.11	<0.01*
2008-2011							
Bare ground (%/ha)	-2.9±3.4	-1.0±3.9	15.8±11.7	5.8±3.9	0.1±10.8	1.93	0.13
Leaf litter (%/ha)	11.4±7.1AC	14.1±7.8A	-46.4±12.0B	-6.3±5.4C	1.3±2.4AC	14.47	<0.001*
Understory cover (%/ha)	22.7±4.2A	27.0±7.4A	57.9±5.7B	14.0±5.1A	50.7±9.9B	13.94	<0.001*

Table 2. Small mammals captured in Sherman live traps in oak regeneration treatments on Cold Mountain Game Land, NC. Traps were open for seven consecutive nights in each unit during 2008 (7,648 total trap nights) and 2010 (7,353 total trap nights).

Species	Pre-treatment 2008	Post-treatment 2010
Mice	217	153
Peromyscids (*P. maniculatus, P. leucopus*)	178	131
Woodland jumping mouse (*Napaeozapus insignus*)	36	21
Golden mouse (*Ochrotomys nuttalli*)	3	1
Shrews	37	21
Northern short-tail shrew (*Blarina brevicauda*)	34	20
Masked shrew (*Sorex cinereus*)	2	0
Smoky shrew (*Sorex fumeus*)	1	1
Voles	9	1
Southern redback vole (*Myodes gapperi*)	6	0
Woodland vole (*Microtus pinetorum*)	3	1
Southern flying squirrel (*Glaucomys volans*)	4	7
Eastern chipmunk (*Tamias striatus*)	1	0

Table 3. Mean species richness (± SE) of small mammals captured in Sherman live traps and drift fence arrays in oak regeneration treatments on Cold Mountain Game Land, NC: control (CONT), herbicide (HERB), shelterwood harvest (SW), prescribed burn 2010 (RX1), and prescribed burn 2009 (RX2). Sherman live traps were open for 7,648 and 7,353 trap nights in 2008 and 2010, respectively. Drift fence arrays were open for 15,552 trap nights in 2008, 19,848 trap nights in 2010, and 17,800 trap nights in 2011.

Year	**Sherman Live Traps**						
	CONT	HERB	SW	RX1	RX2	$F_{4,9}$	P_{trt}
2008	2.8 ± 0.7	4.8 ± 0.7	4.3 ± 0.9	3.3 ± 0.3	2.5 ± 0.5		
2010	2.4 ± 0.7	3.4 ± 0.7	2.3 ± 0.9	2.3 ± 0.3	1.5 ± 0.5		
2010-2008	-0.4 ± 0.5	-1.4 ± 0.8	-2.0 ± 0.7	-1.0 ± 0.0	-1.0 ± 1.0	0.83	0.54
	Drift Fence Arrays						
	CONT	HERB	SW	RX1	RX2	$F_{4,14}$	P_{trt}
2008	6.4 ± 0.7	8.4 ± 1.3	6.8±1.0	6.7 ± 0.7	6.5 ± 0.5		
2010	6.6 ± 1.5	7.8 ± 0.6	4.5±0.3	6.7 ± 1.7	4.0 ± 0.0		
2011	7.0 ± 0.5	7.2 ± 1.0	7.3±0.9	5.7 ± 0.9	5.5 ± 1.5		
(2010 + 2011) - 2008	0.4 ± 0.6	-0.9 ± 0.6	-0.9 ± 1.1	-0.5 ± 0.6	-1.8 ± 0.6	0.62	0.65

Table 4. Small mammals captured in drift fence arrays in oak regeneration treatments on Cold Mountain Game Land, NC. Drift fences were open for 15,552 trap nights in 2008, 19,848 trap nights in 2010, and 17,800 trap nights in 2011.

Species	Pre-treatment 2008	Post-treatment 2010	Post-treatment 2011
Shrews	1,239	1,105	1,005
Masked shrew (*Sorex cinereus*)	768	671	597
Northern short-tailed shrew (*Blarina brevicauda*)	91	164	203
Smoky shrew (*Sorex fumeus*)	98	69	76
Pygmy shrew (*Sorex hoyi*)	18	8	9
Rock shrew (*Sorex dispar*)	15	5	16
Least shrew (*Cryptotis parva*)	4	6	24
Water shrew (*Sorex palustris*)	1	0	0
Voles	44	48	25
Woodland vole (*Microtus pinetorum*)	25	20	23
Southern redback vole (*Myodes gapperi*)	19	28	2
Mice	41	35	50
Deer mouse (*Peromyscus maniculatus*)	12	13	18
White-footed mouse (*Peromyscus leucopus*)	16	13	11
Woodland jumping mouse (*Napaeozapus insignus*)	11	4	15
Golden mouse (*Ochrotomys nuttalli*)	1	3	5

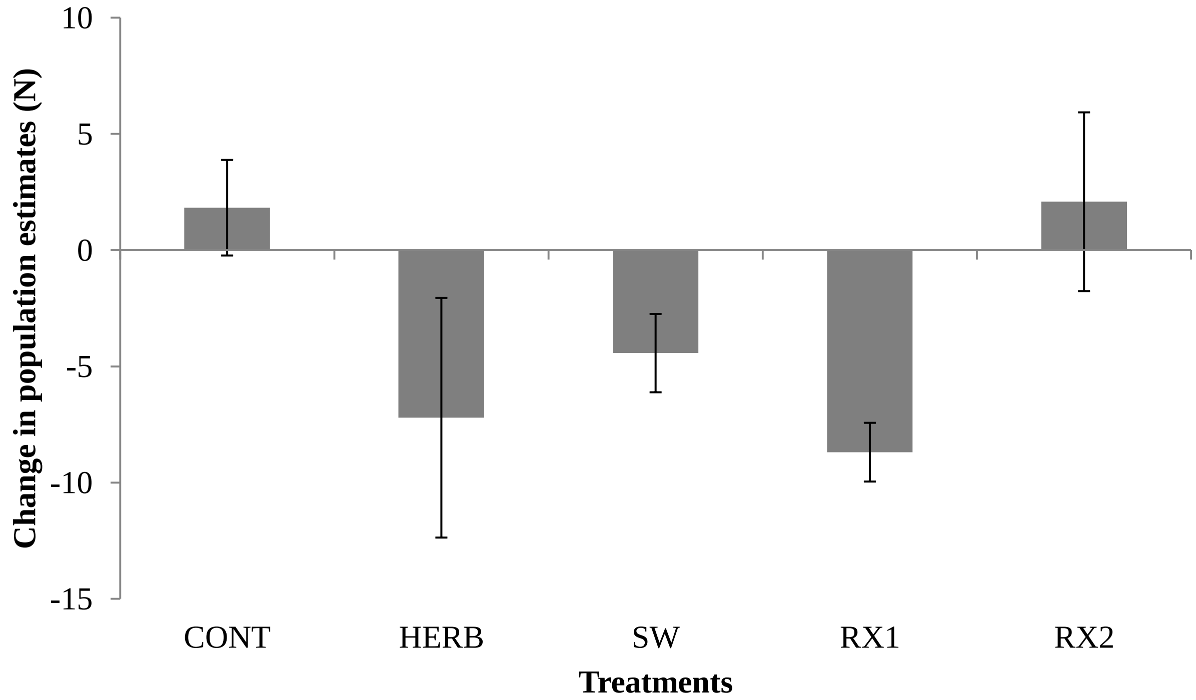

Figure 1. Change in mean population estimates (N) of *Peromyscus* spp. (± SE) from 2008 to 2010 from Sherman live traps in oak regeneration treatments on Cold Mountain Game Land, NC: control (CONT), herbicide (HERB), shelterwood harvest (SW), prescribed burn 2010 (RX1), and prescribed burn 2009 (RX2). Traps were open for 7,648 and 7,353 trap nights in 2008 and 2010, respectively.

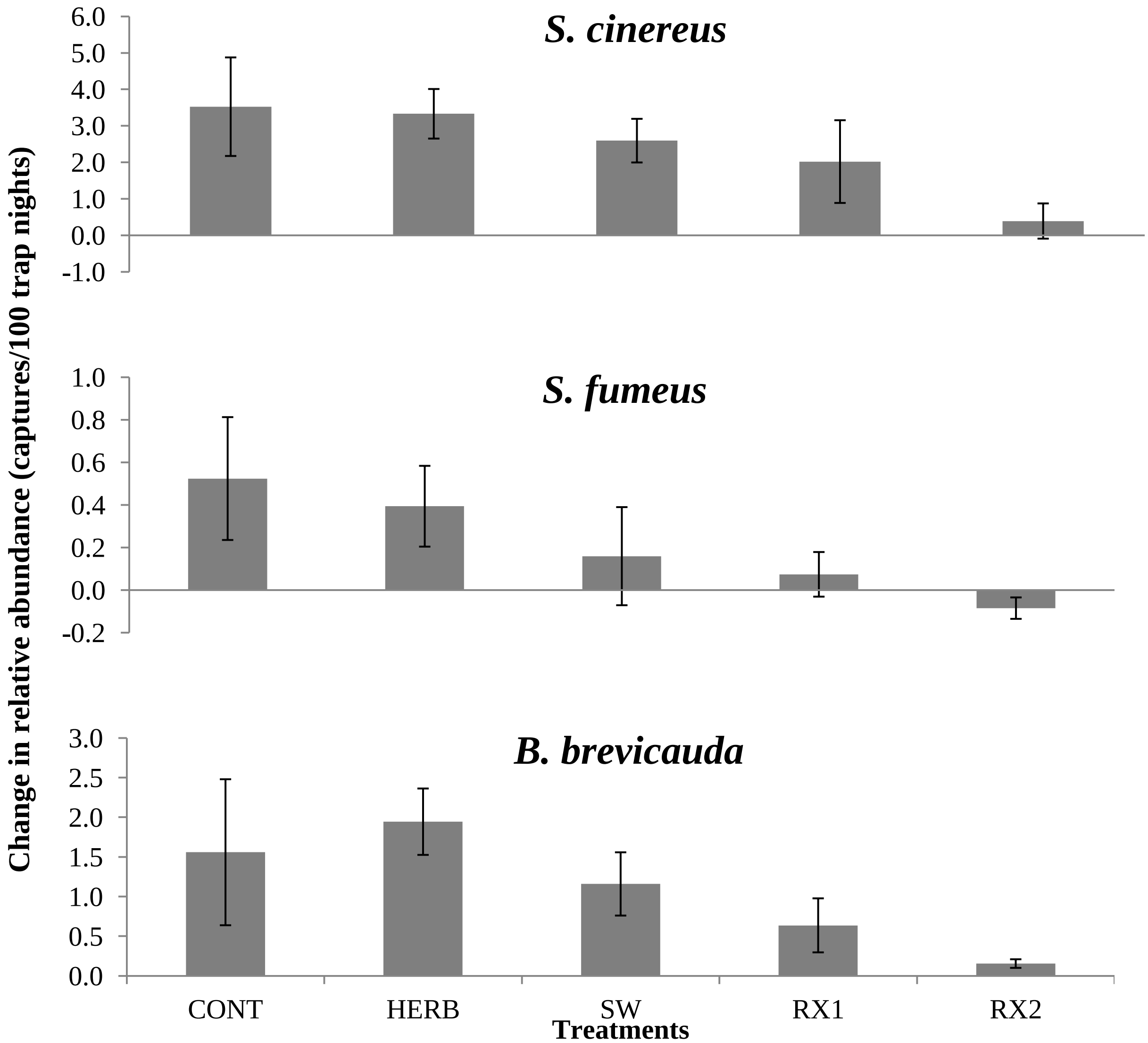

Figure 2. Change in relative abundance (captures/100 trap nights) (± SE) from pre-treatment (2008) to post-treatment (2010 and 2011) in oak regeneration treatments on Cold Mountain Game Land, NC: control (CONT), herbicide (HERB), shelterwood harvest (SW), prescribed burn 2010 (RX1), and prescribed burn 2009 (RX2) for masked shrews (*S. cinereus*), smoky shrews (*S. fumeus*), and northern short-tailed shrews (*B. brevicauda*). Drift fence arrays were open for 15,552, 19,848, and 17,800 trap nights in 2008, 2010, and 2011, respectively.

SHORT-TERM RESPONSE OF *PLETHODON* SALAMANDERS FOLLOWING OAK REGENERATION SILVICULTURAL TREATMENTS

Abstract

Plethodon salamanders are an important ecological component of eastern hardwood forests and may be impacted by forest disturbance caused by silvicultural practices for advance oak regeneration. From May to August 2008 (pre-treatment), 2010 (first year post-treatment), and 2011 (second year post-treatment), we trapped *Plethodon* salamanders to assess changes in relative abundance, percent of juvenile captures, and body condition indices following three oak regeneration silvicultural practices (prescribed fire, midstory herbicide application, and shelterwood harvest). We trapped Southern Appalachian salamanders (*P. teyahalee*) and Southern gray-cheeked salamanders (*P. metcalfi*) in five replicates of the oak regeneration practices and a control using drift fences with pitfall traps. From pre- to post-treatment, the change in relative abundance of both species was similar among treatments; however, there was a greater increase of captures of the Southern Appalachian salamanders in 2011 (second year post-treatment) than in 2010 (first year-post-treatment). Additionally, the change in percent of juvenile captures and body condition indices for both species was similar among treatments. Lack of short-term salamander response in herbicide and prescribed burn treatments was likely because of minor or transitory changes to forest structure. In shelterwood harvests, the initial impacts of reduced canopy cover and leaf litter were likely mitigated by rapid post-treatment sprouting and residual logging debris. Moreover, high levels of precipitation in the second year post-treatment may have compensated for moisture reductions caused by shelterwood harvests and prescribed burns. Because other studies contradict our results on longer time scales (e.g., > 3 years post-treatment), continual monitoring of *Plethodon* spp. is warranted.

Keywords: herbicide, oak regeneration, *Plethodon*, prescribed fire, shelterwood harvest, salamanders, Southern Appalachians

1. Introduction

Ecologically, salamanders are an important component of eastern hardwood forests in the United States. In fact, in some areas of the Appalachians, biomass of salamanders is twice that of birds and equal to small mammals (Burton and Likens, 1975). As secondary consumers, salamanders are an important food source for reptiles, birds, and mammals and regulators of forest floor invertebrates (Whitaker and Rubin, 1971; Burton and Likens, 1975; Pough, 1983; Petranka, 1998; Wyman, 1998; Davic and Hartwell Jr., 2004).

Plethodontid salamanders (e.g., genus *Plethodon*) are especially susceptible to anthropogenic changes to forest. Plethodontids are ectothermic terrestrial salamanders with direct development, and lacking an aquatic larval stage. Lacking lungs, *Plethodon* perform dermal respiration (Petranka, 1998; Pough, 2007) and require moist substrates and high relative humidity (Shelford et al., 1913; Bogert, 1952; Spotila, 1972; Feder, 1983). Thus, forest disturbance that drastically reduces canopy and leaf litter cover will elevate forest floor temperatures (i.e., increased risk of desiccation), reduce primary cover (i.e., increased risk of predation), and compromise prey habitat (Ash, 1988; Petranka, 1994; Ash, 1997; Harpole and Haas, 1999; Knapp *et al.*, 2003; Homyack and Haas, 2009; Matthews *et al.*, 2010). Additionally, following disturbance, terrestrial salamanders may alter aboveground activities (e.g., foraging and mating) or trade growth and reproduction for increased basic maintenance costs (Homyack *et al.*, 2011). Estimates of the time required for recovery of *Plethodon* populations to pre-disturbance levels range from 20 to over 100 years (Moorman *et al.*, 2011). Thus, silviculture practices that alter forest habitat can have negative effects on *Plethodon* populations.

Oak regeneration practices such as prescribed fire, midstory removal using herbicides, and shelterwood harvests result in a wide range of habitat changes. Prescribed fire can remove thin-barked shrubs and trees in the midstory and herbaceous and woody vegetation in the understory (Van Lear and Watt, 1993). Following a prescribed burn, leaf litter decreases, with up to six times more litter documented in unburned than in recently burned sites (Kirkland Jr. *et al.*, 1996; Greenberg *et al.*, 2006; Greenberg *et al.*, 2007; Waldrop *et al.*, 2007). While midstory removal using herbicides and even-aged harvests (e.g., shelterwood harvests) increase light to the forest floor, the latter can produce conditions similar to medium to large-scale natural disturbance. In addition to reductions in leaf litter, rapid re-sprouting of vegetation, and increases in downed woody debris, shelterwood harvests reduce canopy cover creating a high-light environment which elevates forest floor temperatures, consequently reducing soil moisture (Geiger, 1965; Brose *et al.*, 1999; Chen *et al.*, 1999).

Previous research on the effects of silvicultural disturbances on salamanders often involve faulty experimental designs that focus on insufficient measures of population response (e.g., abundance) and lack replication (deMaynadier and Hunter Jr., 1995; Russell *et al.*, 1999). In fact, a review of forest management effects on southeastern herpetofauna identified only six studies with pre- and post-treatment data, treatment replication, or proper spatial and temporal referencing (Russell *et al.*, 2004). Also, demographic characteristics such as age and sex measures are commonly ignored in favor of abundance estimates, which may not accurately portray the health of the targeted taxa (Ash *et al.*, 2003). Therefore, we developed a large-scale replicated study to determine the change in relative abundance of *Plethodon* salamanders following three oak regeneration treatments among treatments. We compared changes in relative abundance of salamanders from pre-treatment (baseline) to post-treatment among the oak regeneration treatments and a control. Additionally, we evaluated the change in proportion of juvenile captures and the change in body condition indices among treatments following treatment implementation.

1.1 Study Area

Our study was conducted in Haywood County, western North Carolina on Cold Mountain Game Land (CMGL). CMGL encompassed ~5900 ha of second growth, upland mixed-oak forests with elevations ranging from ~1,100-1,350 m. CMGL was managed by the North Carolina Wildlife Resources Commission primarily for diverse wildlife habitat and was located along the escarpment of the Blue Ridge Physiographic Province. Terrain was mountainous with gentle to steep slopes with predominant overstory trees of oak (*Quercus* spp.), hickory (*Carya* spp.), red maple (*Acer rubrum*), sugar maple (*Acer saccharum*), black cherry (*Prunus serotina*), and yellow-poplar (*Liriodendron tulipifera*). The midstory consisted primarily of shade-tolerant species including sourwood (*Oxydendrum arboreum*), blackgum *(Nyssa sylvatica*), and red maple.

2. Material and Methods

In CMGL, we had 20 5-ha units: 3 treatments plus 1 control, 5 replicates of each. Treatment units met selection criteria described in Keyser et al. (2008). Treatments (prescribed fire, midstory removal using herbicide, and shelterwood harvest) were assigned randomly to each treatment unit resulting in a completely randomized design (Keyser *et al.*, 2008).

2.1 Treatments

Treatments were designed to evaluate three oak regeneration practices: (1) 3 prescribed burns at ~4-year intervals, (2) midstory removal using herbicide with re-application after ~3 years, and (3) shelterwood harvest with 30-40% basal area (BA) retention followed by a prescribed fire after ~3 years. All three practices will be followed by overstory removal ~11 years following initial treatments. Because this study only encompassed one year pre-treatment and two years post-treatment, we evaluated the response of *Plethodon* salamanders to the first prescribed burn, initial midstory herbicide treatment, shelterwood harvest, and controls.

Because weather and road conditions did not permit burning of all five units in 2009, two of the five replicate units were burned in April 2009 with the remaining three units burned in April 2010. Thus, the prescribed burn treatment was separated into two treatments because of ecological differences related to time since burn.

In late summer 2008, competing midstory trees (≥5.0 cm and <25.0 cm dbh) were treated with herbicide (i.e., Garlon 3A) using the hack-and-squirt method where ~1 ml of diluted solution was sprayed into a waist-high incision of each midstory tree marked for removal (Loftis, 1990). The goal of this treatment was to reduce total BA by 25-30% without creating new canopy gaps. Ideally, increases in photosynthetically active radiation (PAR) on the forest floor would promote oak seedling growth and successful recruitment into the canopy (Loftis, 1990).

The shelterwood harvest was implemented from winter 2009 to early summer 2010 with the goal of leaving approximately 30-40% of the original stand BA and enhancing light conditions on the forest floor (Brose *et al.*, 1999). The majority of leave trees were dominant or codominant oak trees and most slash was left on-site.

During this study, no silvicultural manipulation occurred in the control plots.

2.2 Salamander Sampling

We sampled salamanders from mid-May to mid-August in 2008 (pre-treatment), 2010 (first year post-treatment), and 2011 (second year post-treatment). In 2008 and 2011, we sampled salamanders in all 20 units. In 2010, we were unable to sample in one shelterwood harvest unit because it was not harvested until mid-summer 2010.

We established six randomly oriented single-arm drift fence arrays in 2008 within all 20 units (Todd *et al.*, 2007) with three fences installed at a lower slope site (e.g., lower one-third of each unit) and three fences at an upper slope site (e.g., upper one-third of each unit) (Greenberg and Waldrop, 2008). In two of the treatment replicates (one herbicide and one control), we were unable to establish an upper site due to steep and rocky terrain. A fourth fence was installed at each lower and upper location that had fences by 2010. Drift fences were >10 m apart and constructed of 7.6-m sections of aluminum flashing with a 19-L bucket buried at each end, flush with the ground. We placed a moist sponge in each bucket to provide moisture for captured salamanders, which were recorded for species, age, body and tail length, and weight. In 2008 and 2010, new captures were injected with visible implant elastomer (VIE).

2.3 Habitat Data

In each unit, we used a portable GPS device to measure elevation at upper and lower slope sites. In each unit, we recorded overall aspect as a binary value: 0 = south- and west-facing aspects or 1 = north- and east-facing aspects. In 2008 and 2010, we measured canopy cover and CWD and in all years, we measured ground cover. We measured canopy cover with a spherical densiometer at each drift fence and averaged per unit (Lemmon, 1956). We measured ground cover and CWD along a 15-m randomly oriented transect line at each drift fence starting from the bucket furthest uphill. We measured percent cover of bare ground, leaf litter, understory cover (i.e., plants < 0.9 m), and CWD (≥12-cm diameter). Along each transect, we recorded 'start' and 'stop' distance for each category and then summed the total distance along each transect. For understory cover, measurements were determined by the potential cover (e.g., shading) provided by each plant. In each unit, percent cover for each category was determined by dividing the total summed distance of the category by 90 m (six transects per unit). We recorded total length, bark class, and amount of decay for each piece of CWD. Bark class was visually categorized from 1-5: 1 = recently dead with 100% of bark, 2 = >70% of bark, 3 = 40-69% of bark, 4 = 10-39% of bark, and 5 = <10% of bark. Amount of decay was visually categorized from 1-6: 1 = no decay visible, 2 = slight decay, 3 = moderate decay, 4 = slight fragmentation, 5 = heavy fragmentation, and 6 = completely disintegrated but still distinguishable as CWD.

2.4 Analyses

We used a one-way analysis of variance (ANOVA) to compare the change in canopy cover and CWD characteristics from 2008 to 2010 among treatments. Because percent ground cover was measured twice post-treatment (e.g., 2010 and 2011), we performed a repeated measures ANOVA using a mixed model procedure that consisted of a random effect (i.e., treatment per unit) and fixed effects of treatment and year with an interaction. In all habitat analyses, when models were significant, we used Tukey's Studentized Range (HSD) test to determine significant differences among treatment means.

Slope position was not an important predictor of *Plethodon* response from pre- to post-treatment, so we did not include this variable in the analysis. For each salamander species captured, we calculated relative abundance as the number of new animals captured per 100 trap nights. Relative abundances were then log-transformed for normality. Our response variable was the change in relative abundance from 2008 to 2010 or 2008 to 2011. We analyzed the response of *Plethodon* and two individual species for which we had >70 captures per year [i.e., Southern gray-cheeked salamanders (*P. metcalfi*) and Southern Appalachian salamanders (*P. teyahalee*)] among treatments using a repeated measures ANOVA with a covariance structure of first order autoregression [AR (1)]. The mixed model consisted of a random effect (i.e., treatment per unit) and fixed effects [i.e., treatment, year, and two covariates (aspect and elevation)] and the interactions between year and treatment and covariates and treatment. We removed interactions and covariates from the model when not statistically significant. If significant, we tested for differences among the treatments using Tukey's Studentized Range (HSD) test.

We evaluated the change from pre-treatment to post-treatment for percent of juvenile captures and body condition indices for Southern gray-cheeked salamanders and Southern Appalachian salamanders. We determined body condition indices by dividing weight (grams) by total length (millimeters) (Karraker and Welsh Jr., 2006). Recaptures or individuals missing tails or parts of tails were excluded. Body condition indices were then log-transformed for normality. Similar to the relative abundance, our response variable (i.e., the change in percent of juvenile captures and body condition indices from 2008-2010 and 2008-2011) was compared among treatments using a repeated measures ANOVA and was identical to the modeling and analysis outlined above. All statistical tests were conducted in SAS (v. 9.1.3, SAS Institute, Cary, NC).

3. Results

3.1 Habitat

Elevation ($F_{4,\ 14}$ = 2.12, P = 0.12) and aspect ($F_{4,\ 31}$ = 1.96, P = 0.13) were similar among treatments. From 2008 to 2010, changes in most habitat variables differed among treatments. Canopy cover in the shelterwood harvest declined 40% post-harvest. Bare ground increased in prescribed burns of 2010 and shelterwood harvests by 48% and 33%, respectively. Leaf litter decreased by 64%, 41%, and 38% in prescribed burns of 2009, shelterwood harvests, and prescribed burns of 2010, respectively. In shelterwood harvests, CWD cover increased by 5%, had more bark, and was less decayed. From 2008 to 2011, changes in percent bare ground were similar among treatments as leaf litter levels recovered in prescribed burns and understory cover increased in shelterwood harvests. Leaf litter declined 46% in shelterwood harvests, but recovered to pre-treatment levels in both prescribed burn treatments. Understory cover increased 58% and 51% in shelterwood harvests and prescribed burns of 2009, respectively (Table 1).

3.2 Salamander Sampling

In 2008, 2010, and 2011 we captured 132, 139, and 90 Southern gray-cheeked salamanders and 114, 78, and 91 Southern Appalachian salamanders, respectively. Drift fences were open for 15,552 trap nights in 2008, 19,848 trap nights in 2010, and 17,800 trap nights in 2011.

From pre- to post-treatment, the changes in relative abundance of the Southern gray-cheeked salamander ($F_{4,\ 14}$ = 0.97, P = 0.46) and the Southern Appalachian salamander ($F_{4,\ 10}$

= 1.60, P = 0.25) were similar among treatments (Figure 3). However, there was a year effect for the Southern Appalachian salamander ($F_{1, 18}$ = 20.63, P < 0.001) with a greater increase in captures in 2011 (i.e., second year post-treatment) than in 2010 (i.e., one year post-treatment). For the Southern Appalachian salamander, there was an interaction between treatment and aspect ($F_{4, 10}$ = 4.01, P = 0.04); however, change in relative abundance was similar among treatments on north and east-facing aspects ($F_{3, 19}$ = 1.79, P = 0.18) and on south- and west-facing aspects ($F_{4, 19}$ = 2.34, P = 0.09). The change in the proportion of captured juvenile Southern gray-cheeked salamander ($F_{4, 14}$ = 1.44, P = 0.27) and the change in proportion of captured juvenile Southern Appalachian salamander ($F_{4, 14}$ = 0.22, P = 0.92) were similar among treatments. Similarly, following treatment, the changes in body condition indices of the Southern gray-cheeked salamander ($F_{4, 14}$ = 2.46, P = 0.09) and the Southern Appalachian salamander ($F_{4, 14}$ = 1.67, P = 0.21) were similar among treatments.

4. Discussion

We did not detect short-term effects of the oak regeneration treatments on *Plethodon* abundance. Similarly, other studies indicate salamanders generally increase or show no response to herbicide applications (Cole *et al.*, 1997; Harpole and Haas, 1999; Hood *et al.*, 2002; Brunjes *et al.*, 2003; Homyack and Haas, 2009) or a single prescribed burn (Ford et al., 1999; Keyser et al., 2004; Greenberg and Waldrop, 2008; Ford et al., 2010). Greenberg and Waldrop (2008) observed that *Plethodon* salamanders showed no detectable change following a single, low-intensity prescribed fire with moderate reductions in leaf litter. Further, Keyser et al. (2004) reported no detectable differences in abundance of *P. cinereus* after high-intensity prescribed burns that reduced leaf litter significantly. In our study, the prescribed fires of 2010 were low-intensity because of high moisture in burned areas;

whereas in 2009, the burns were hotter and removed most of the leaf litter. However, salamander abundance after the low- and high-intensity burns changed similarly to unburned areas, indicating short-term environmental changes following single prescribed fires are not enough to influence salamander abundance. Moreover, increases in bare ground immediately after prescribed burns disappeared by the second year post-treatment, mainly due to rapid recovery of leaf litter and understory cover which may have mitigated possible stresses on moisture-sensitive *Plethodon*.

Although shelterwood harvests resulted in substantial reductions in canopy and leaf litter cover and increases in bare ground, salamander captures did not decline < 2 years post-harvest. Similarly, in North Carolina, *Plethodon* showed high site fidelity up to two years following timber harvests (Messere and Ducey, 1998; Ford *et al.*, 2000; Bartman *et al.*, 2001), even in the first year following clearcutting (Ash, 1988).

In the shelterwood harvests, it is possible that with increased levels of CWD, salamanders had sufficient cover, travel corridors, and foraging opportunities (Grover, 1998; Morneault *et al.*, 2004; Patrick *et al.*, 2006; Rundio and Olson, 2007). Additionally, emigration or mortality due to starvation and dehydration, which normally begins late in the first year and in subsequent years (Knapp *et al.*, 2003), may have been offset by high levels of precipitation in the second year post-treatment. Average precipitation for Haywood County from May to August of 2011 was 43 inches, compared to 17 inches in 2010 a difference of 26 inches of rainfall (National Weather Service, 2011). Thus, higher rainfall may have compensated for reductions in moisture, especially in shelterwood harvests on xeric aspects, essentially maintaining microclimatic conditions within the tolerance limits of *Plethodon* at these sites. Additionally, the relatively cool climate of our high elevation sites may have mitigated some of the microclimatic conditions associated with salamander declines following timber harvests elsewhere (Harper and Guynn Jr., 1999).

However, following the oak regeneration treatments, delayed population response of *Plethodon* could cause delayed changes in abundance and demographic parameters, especially in shelterwood harvests. *Plethodon* are vulnerable to loss of canopy cover with a potential five-fold decline in abundance following timber harvest (deMaynadier and Hunter Jr., 1995). Hence, it is possible that our short-term sampling window (e.g., < 2 year post-harvest) was not sufficient to detect declines of terrestrial salamanders following shelterwood harvests. In fact, other studies have noted major declines in salamander abundance 2-10 years after harvests with >50% canopy removal (Pough *et al.*, 1987; Ash, 1988; Petranka *et al.*, 1993; Petranka, 1994; Reichenbach and Sattler, 2007). Ash et al. (2003) noted that *P. jordani* captures in 10-year old clearcuts had proportionately fewer juveniles and adult males in reproductive condition. We did not observe a treatment difference in the proportion of juveniles captured, suggesting that juvenile dispersal and differences in survival among age classes were not factors affecting salamander populations in the short-term. Declines in reproductive success may not be evident until >3 years following forest disturbance (Ash *et al.*, 2003; Patrick *et al.*, 2006; Cummer and Painter, 2007; Homyack and Haas, 2009; Matthews *et al.*, 2010), indicating the importance of long-term studies to fully assess effects of timber harvest on salamander abundance.

Planned activities associated with the oak regeneration methods we studied, including repeated prescribed fires, repeated herbicide applications, and prescribed burns in the shelterwood harvest units may have additive effects on salamanders (Matthews *et al.*, 2010). The repeated prescribed burning required to facilitate oak seedling growth to a competitive size prior to canopy removal (Van Lear and Watt, 1993) could compound treatment effects (e.g., decreased soil moisture and reduced leaf litter) and impact plethodontid salamanders by decreasing relative humidity in burrows (Floyd *et al.*, 2001). Additionally, the combination of shelterwood harvests and prescribed fires may result in a more substantial change in habitat conditions compared to any of these disturbances alone. For example, Matthews et al. (2010) captured 68-72% fewer salamanders in mechanical fuel reductions followed by two prescribed fires (3 years apart) than in twice-burned treatments alone.

4.1 Conclusion

We did not detect any short-term changes to populations of Southern Appalachian salamanders or Southern gray-cheeked salamanders following oak regeneration treatments. Minor changes to habitat structure following herbicide treatments, rapid re-establishment of understory cover in shelterwood harvests and prescribed burns, and residual piles of logging slash in shelterwood harvests may have contributed to lack of short-term response. However, lags in demographic response to treatments may cause delayed (>2 years) declines in *Plethodon* populations in the longer-term. Moreover, planned activities associated with the oak regeneration systems, such as prescribed burns following shelterwood harvests, may have additive effects on salamanders. Therefore, long-term studies of salamander response to varying frequencies and combinations of oak regeneration treatments are imperative. We recommend forest managers integrate multiple objectives into management plans and consider efficacy of oak regeneration treatments in concert with conservation of sensitive wildlife species like salamanders. Additionally, small-scale silvicultural activities in landscapes dominated by mature mixed-oak forest likely have no influence on regional wildlife abundance.

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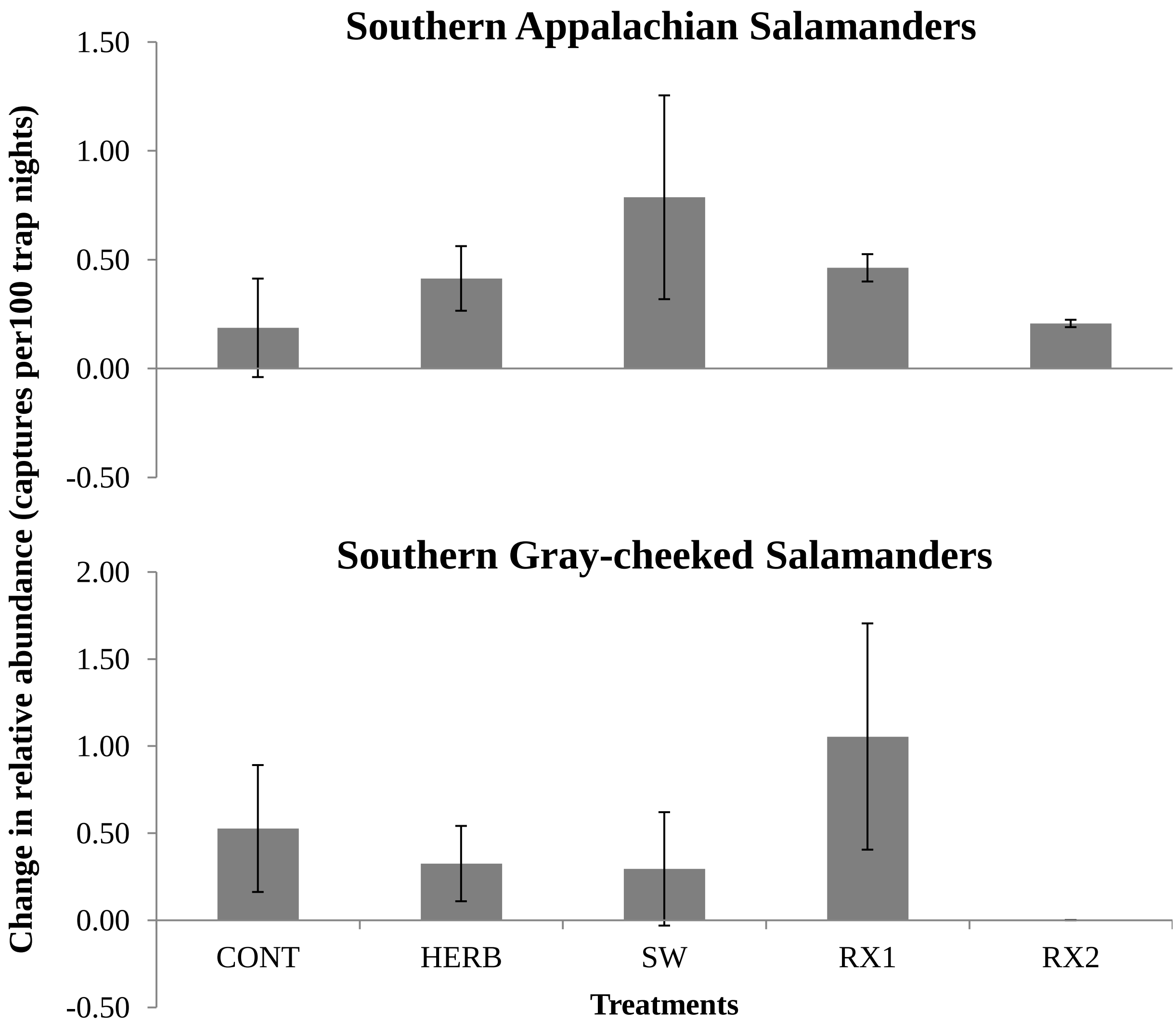


Figure 3. Change in relative abundance (captures/100 trap nights) (± SE) from pre-treatment (2008) to post-treatment (2010, 2011) in oak regeneration treatments on Cold Mountain Game Land, NC: control (CONT) (n = 5), midstory herbicide (HERB) (n = 5), shelterwood harvest (SW) (n = 4), prescribed burn 2010 (RX1) (n = 3), and prescribed burn 2009 (RX2) (n = 2) for the Southern Appalachian salamander (*P. teyahalee*) and the Southern gray-cheeked salamander (*P. metcalfi*). Drift fence arrays were open for 15,552, 19,848, and 17,800 trap nights in 2008, 2010, and 2011, respectively.

APPENDICES

Appendix A

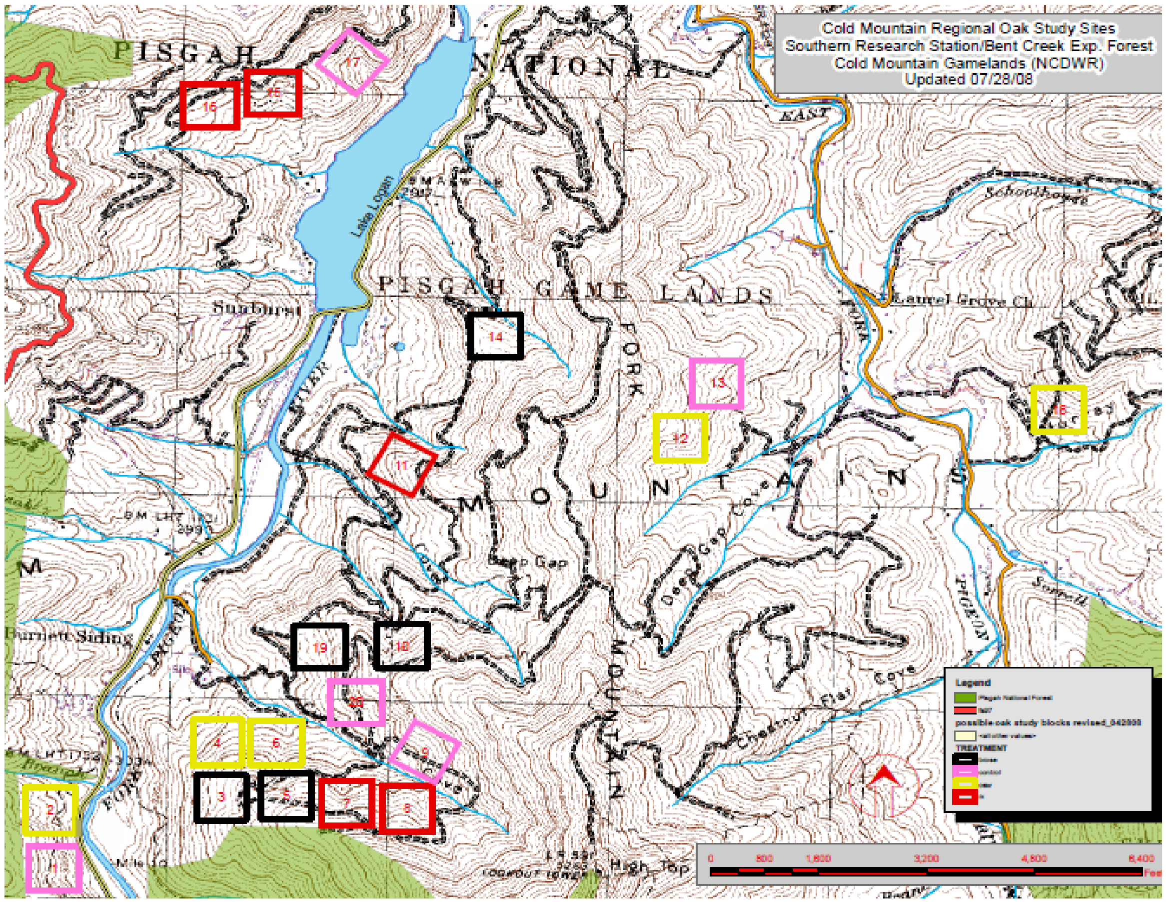
List of species captured in oak regeneration treatments on Cold Mountain Game Land, North Carolina in 2008, 2010, and 2011 using Sherman live traps and drift fence arrays. Sherman live traps were open for seven consecutive nights in each unit during 2008 (7,648 total trap nights) and 2010 (7,353 total trap nights). Drift fence arrays were open for 15,552, 19,848, and 17,800 trap nights in 2008, 2010, and 2011, respectively.

Common Name	**Species**	**2008**	**2010**	**2011**	**Total**
Masked Shrew	*Sorex cinereus*	799	671	617	2087
Northern Short-tail Shrew	*Blarina brevicauda*	130	204	211	545
Southern Gray-cheeked Salamander	*Plethodon metcalfi*	132	139	92	363
Southern Appalachian Salamander	*Plethodon teyahalee*	114	78	108	300
Smoky Shrew	*Sorex fumeus*	102	72	80	254
White-footed Mouse	*Peromyscus leucopus*	101	121	11	233
Deer Mouse	*Peromyscus maniculatus*	105	23	18	146
Blue Ridge Two-lined Salamander	*Eurycea wilderae*	11	32	61	104
Eastern Newt	*Notophthalmus viridescens*	26	49	15	90
Pygmy Salamander	*Desmognathus wrighti*	5	40	44	89
Woodland Vole	*Microtus pinetorum*	29	21	27	77
Woodland Jumping Mouse	*Napaeozapus insignus*	47	21	0	68
Ringneck Snake	*Diadophis punctatus*	28	19	19	66
Eastern Garter Snake	*Thamnophis sirtalis*	33	15	14	62
Southern Red-backed Vole	*Myodes gapperi*	25	28	2	55
American Toad	*Bufo* (*Anaxyrus*) *americanus*	5	17	27	49
Rock Shrew	*Sorex dispar*	17	5	17	39
Pygmy Shrew	*Sorex hoyi*	20	8	9	37
Least Shrew	*Cryptotis parva*	5	6	25	36

Southern Flying Squirrel	*Glaucomys volans*	4	7	20	31
Wood Frog	*Rana sylvatica*	12	3	13	28
Allegheny Mountain Dusky Salamander	*Desmognathus ochrophaeus*	4	3	14	20
Five-lined Skink	*Eumeces fasciatus*	2	2	14	18
Golden Mouse	*Ochrotomys nutalli*	4	4	5	13
Fowler's Toad	*Bufo* (*Anaxyrus*) *fowleri*	0	3	9	12
Blackchin Red Salamander	*Pseudotriton ruber schencki*	4	4	3	11
Eastern Red-backed Salamander	*Plethodon cinereus*	3	1	4	8
Coal Skink	*Eumeces anthracinus*	1	3	3	7
Eastern Mole	*Parascalops breweri*	2	3	2	7
Eastern Box Turtle	*Terrapene carolina*	5	2	0	7
Star-nosed Mole	*Condylura cristata*	0	4	3	7
Timber Rattlesnake	*Crotalus horridus*	2	2	1	5
Worm Snake	*Carphophis amoenus*	4	0	0	4
Black Rat Snake	*Elaphe obsoleta*	3	0	0	3
Eastern Fence Lizard	*Sceloporus undulatus*	1	1	1	3
Eastern Chipmunk	*Tamias sciurus*	1	1	0	2
Water Shrew	*Sorex palustris*	1	0	0	1

Appendix B

Cold Mountain Game Land, NC: control – CONT (pink: 1, 9, 13, 17, 20), midstory herbicide – HERB (yellow: 2, 4, 6, 12, 18), shelterwood harvest – SW (black: 3, 5, 10, 14, 19), prescribed burn 2010 – RX1 (red: 7, 8, 11), and prescribed burn 2009 – RX2 (red: 15, 16).



Appendix C

GPS coordinate (UTM) locations where drift fence arrays were installed at lower and upper sites of each unit on Cold Mountain Game Land in Haywood County, NC.

Unit	Treatment	Lower Site	Upper Site
1	CONT	17S0323583, 3917142	17S0323388, 3917166
2	HERB	17S0323521, 3917439	17S0323383, 3917420
3	SW	17S0324315, 3917501	17S0324315, 3917510
4	HERB	17S0324252, 3917869	17S0324219, 3917702
5	SW	17S0324532, 3917595	17S0324528, 3917455
6	HERB	17S0324470, 3917843	17S0324501, 3917717
7	RX1	17S0324778, 3917557	17S0324793, 3917428
8	RX1	17S0325087, 3917529	17S0325077, 3917400
9	CONT	17S0325092, 3917682	-
10	SW	17S0325068, 3918335	17S0325073, 3918224
11	RX1	17S0324959, 3919183	17S0325116, 3919125
12	HERB	17S0326364, 3919321	17S0326199, 3919314
13	CONT	17S0326501, 3919590	17S0326371, 3919629
14	SW	17S0325419, 3919863	17S0325424, 3919753
15	RX2	17S0324406, 3920919	17S0324403, 3921085
16	RX2	17S0324108, 3920871	17S0324130, 3921015
17	CONT	17S0324800, 3921073	17S0324699, 3921189
18	HERB	17S0327997, 3919399	-
19	SW	17S0324566, 3918282	17S0324734, 3918246
20	CONT	17S0324810, 3917967	17S0324906, 3918001