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ScienceDirect

Forest Ecology and Management 234 (2006) 355–362

Forest Ecology and Management

www.elsevier.com/locate/foreco

Response of white-footed mice (*Peromyscus leucopus*) to fire and fire surrogate fuel reduction treatments in a southern Appalachian hardwood forest

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Received 12 April 2006; received in revised form 25 July 2006; accepted 26 July 2006

Abstract

An experiment conducted as part of the multidisciplinary National Fire and Fire Surrogate Study was designed to determine effects of three fuel reduction techniques on small mammals and habitat structure in the southern Appalachian mountains. Four experimental units, each >14-ha were contained within each of three replicate blocks at the Green River Game Land, Polk County, NC. Treatments were (1) prescribed burning (B); (2) mechanical felling of shrubs and small trees (M); (3) mechanical felling + burning (MB); (4) controls (C). Mechanical understory felling treatments were conducted in winter 2001–2002, and prescribed burning was conducted in March 2003. After treatment, there were fewer live trees, more snags, and greater canopy openness in MB than in other treatments. Leaf litter depth was reduced by burning in both B and MB treatments, and tall shrub cover was reduced in all fuel reduction treatments compared to C. Coarse woody debris pieces and percent cover were similar among treatments and controls. We captured 990 individuals of eight rodent species a total of 2823 times. Because white-footed mice composed >79% of all captures, we focused on this species. Populations in experimental units increased 228% on average between 2001 and 2002, but there was no evidence of an effect of the mechanical treatment. From 2002 to 2003, all units again showed an average increase in relative population size, but increases were greater in MB than in the other treatments. Age structure and male to female ratio were not affected by the fuel reduction treatment. Average adult body weight declined from 2001 to 2002, but less so in M than in units that remained C in both years. The proportion of mice captured near coarse woody debris was similar to the proportion captured in open areas for all treatments, indicating that white-footed mice did not use coarse woody debris preferentially or change their use patterns in response to fuel reduction treatments. Land managers should understand possible effects of different fuel reduction treatments on white-footed mouse populations, as they are an important component of the fauna and food chain of deciduous southern Appalachian forests.
Published by Elsevier B.V.

Keywords: White-footed mice; *Peromyscus leucopus*; Small mammals; Fuel reduction; Prescribed burn; Shrub reduction; Coarse woody debris

1. Introduction

Buildup of forest fuels such as thick shrub cover or woody debris contributes to the potential for wildfire in many ecosystems. In the southern Appalachian mountains, frequent burning was used by native Americans to improve conditions for travel and game, and later by European settlers to improve grazing for livestock (Brose et al., 2001). Little is known about fire frequencies and intensities in the southern Appalachians prior to human influence. Lightning-caused fires are rare (Harmon, 1982), but frequencies differ with topography and associated forest types (Delcourt and Delcourt, 1997). However, several studies from geographically disparate locations indicate that fire was relatively common in oak forests of the eastern and central United States before the recent era of fire suppression (see Schuler and McClain, 2003).

In the early 1900s forest fires began to be viewed as destructive, and they were suppressed or excluded where possible (Brose et al., 2001). Fire exclusion led to higher mid- and understory densities of shade-tolerant trees and shrubs, especially on mesic upland sites (Brose et al., 2001). Today, prescribed burning is employed as a forest management tool for ecosystem restoration, oak regeneration, understory control, and fuel reduction. Mechanical methods can be employed to

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0378-1127/$ – see front matter. Published by Elsevier B.V.
doi:10.1016/j.foreco.2006.07.022

reduce the forest understory in lieu of prescribed fire when burning is not feasible or practical. Recently, fuel reduction techniques – prescribed fire or fire surrogates – have received national attention (Graham et al., 2004). Yet, the impacts of such habitat manipulations on populations of small mammal are not well understood, especially in southern hardwood ecosystems.

Changes in habitat structure due to intense disturbance have the potential to affect rodent populations and community composition. Habitat features, such as shrub and canopy cover, snags, and down coarse woody debris (CWD) provide cover and nest sites. Coarse woody debris also harbors fungi and invertebrate food sources for some rodents (Loeb, 1996). Increased light availability following disturbance may increase food availability by promoting plant productivity, fruit and seed production (Blake and Hoppes, 1986; Greenberg et al., in press), and higher densities of flying and foliar arthropods (Campbell et al., in press; Whitehead, 2003). However, population densities are not necessarily reflective habitat quality (Van Horne, 1983). Disturbed areas could have a positive or negative effect on demographic or fitness parameters such as reproductive rates, dispersal patterns, body weight, and survival (Van Horne, 1983; Sullivan, 1979).

Peromyscus spp., especially white-footed mice, are common in most eastern deciduous forest types (Godin, 1977). Reported densities of white-footed mice range from 3.7 to 93.4/ha, and vary among years and habitats (Brooks et al., 1998). These mice are important seed dispersers, and predators of insects, bird eggs, and seeds (Wolf and Batzli, 2004). White-footed mice are an important prey item for carnivorous mammals, raptors and snakes (Sullivan, 1990). Their far-reaching influence on forest dynamics is illustrated by their role in controlling gypsy moth (*Lymantria dispar*) by preying on pupae (Elkinton et al., 1996) and their role as secondary hosts for Lyme disease (Jones et al., 1998).

In 2000, the National Fire and Fire Surrogate (NFFS) Study was initiated by the Joint Fire Science Program to research impacts of fuel reduction treatments on multiple components of forested ecosystems across the United States (Youngblood et al., 2005). In 2001 the Green River Game Land in Polk County, NC, was selected to represent the southern Appalachian upland hardwood forest ecosystem in the NFFS. This site was added to the original study through funding from the National Fire Plan. As part of the NFFS we studied the response of white-footed mice (the only species that was captured in high numbers) to three fuel reduction treatments (prescribed burning, mechanical understory removal, and mechanical understory removal + prescribed burning) in the southern Appalachians.

Land managers need to know how different fuel reduction practices affect small mammal populations to better integrate wildlife management with forest management. In this paper we examine how fuel reduction treatments by prescribed burning and (or) mechanical understory removal affect habitat structure and white-footed mouse populations. Specifically, we examined whether population estimates, individual body weight, sex structure, or age structure varied among treatments. We also compared capture rates near versus distant from CWD to determine whether CWD use by white-footed mice differed among treatments, and whether they used it preferentially over open microsites.

2. Study area and methods

2.1. Study area

Our study was conducted on the Green River Game Land in Polk County, NC. The Game Land is managed by the North Carolina Wildlife Resources Commission, and lies within the mountainous Blue Ridge Physiographic Province of Western North Carolina. Soils were primarily of the Evard series (fine-loamy, oxidic, mesic, Typic Hapludults), which are very deep and well drained in mountain uplands (USDA Natural Resources Conservation Service, 1998). There were also areas of rocky outcrops in steeper terrain. Forest stands were composed mainly of oaks (*Quercus* spp.) and hickories (*Carya* spp.). Shortleaf (*Pinus echinata*) and Virginia (*P. virginiana*) pines were found on ridgetops, and white pine (*P. strobus*) and yellow poplar (*Liriodendron tulipifera*) occurred in moist coves. Thick shrub layers occurred throughout much of the study area. Predominant shrubs were mountain laurel (*Kalmia latifolia*) along ridge tops and on upper southwest-facing slopes, and rhododendron (*Rhododendron maximum*) in mesic areas. Elevation ranged from approximately 366–793 m.

2.2. Study design

We selected three study areas (blocks) within the Game Land (see Waldrop, 2001). Study blocks were selected based upon stand size (large enough to accommodate all four treatments), stand age, cover type, and management history to ensure that baseline conditions were consistent among the treatments. First and second order streams bordered and (or) traversed all three replicate blocks. Stand ages varied from 80 to 120 years (Waldrop, 2001). Four experimental units, each >14 ha were contained within each block. This unit size allowed for 10-ha treatment core areas, each surrounded by a 20 m buffer. None of the units had been thinned during the preceding 10 years and none had been burned in at least 5 years.

Three treatment regimes and an untreated control (C) were randomly assigned to the four experimental units within each block. Treatments were (1) fuel reduction by mechanical understory felling in winter 2001–2002 (M); (2) fuel reduction by prescribed burning in March 2003 (B); (3) fuel reduction by mechanical understory felling in winter 2001–2002 and prescribed fire in March 2003 (MB). The shrub layer was removed using chainsaws, and included all mountain laurel, rhododendron, and trees >1.8 m tall and <10.0 cm in diameter at breast height (dbh). Fuels were not removed for economic reasons, but felled stems were cut repeatedly to reduce piles to less than 1.2 m tall. Prescribed burns were conducted in B and MB treatments on 12 March or 13 March 2003; burning was done 1 year after felling to allow decomposition of some fuels so that fire intensity would be reduced. One block was burned by hand ignition using spot fire and strip-headfire techniques. The two other blocks were burned as a single unit. Backing fires were set along fire lines by hand followed by spot fires set by a helicopter using a plastic sphere dispenser. See Phillips et al. (2006) for methods used to measure fuel loadings, fire temperature, and fire behavior.

2.3. Small mammal trapping

We live trapped small mammals during July and August 2001 (pre-treatment), 2002 (after only mechanical treatments had been implemented in M and MB treatments), and 2003 (after all fuel reduction treatments had been implemented). We established square or rectangular trapping grids (depending on the shape of treatment stands) at least 25 m from treatment edges. We used 60–70 Sherman live traps (7.7 cm × 9.0 cm × 23.3 cm) spaced at approximately 25-m intervals to cover 2.6–3.3 ha in each treatment stand. Traps were baited with oatmeal and shelled whole peanuts. To assess whether trap placement near CWD influenced capture rates (e.g., Bowman et al., 2000), we placed grid traps either adjacent to CWD (defined as ≥10 cm diameter and ≥1 m long), or in the open (defined as ≥1 m away from CWD) in approximately equal proportions. Trap placement at these two microsites (CWD or in the open) was haphazard rather than systematic, as CWD was not consistently available at or near alternate 25-m grid spacing intervals. This was done during all years, but we recorded trap placement (CWD or open) only during 2002 and 2003. Traps were open continuously for 10 nights and checked each morning. All experimental units within a given block were trapped simultaneously; blocks were trapped successively. Small mammals were identified, weighed and measured (head–body and total length), sexed, tagged in the right ear with an individually numbered tag (size 1 Monel; National Band and Tag Co., Newport, KY), and released at the capture site. Trap number and microsite (during 2002 and 2003) was recorded for all captured animals.

We determined that all *Peromyscus* spp. trapped were *P. leucopus* rather than *P. maniculatus* based on head–body:tail ratio (tail length < head–body length) and absence of a tail tuft (Wolff et al., 1983). In addition, all live specimens captured 1 day, and several dead specimens from this study were confirmed to be *P. leucopus* by S. Miller (curator of mammals, Clemson University, Bob & Betsy Campbell Museum of Natural History); a few specimens were assigned a unique catalogue number and deposited in the museum (Accession # 1026) after identification.

2.4. Habitat measurements

Pre-treatment habitat variables including live tree and snag (≥10 cm dbh) densities, percent tall (≥1.4 m ht) shrub cover, coarse woody debris (≥1 m in length and ≥15 cm large-end diameter within transect), and canopy openness were measured in all treatment areas. These variables were measured again during the growing season immediately post-treatment (2002 for M and 2003 for C, B, and MB). Trees and snags (≥10 cm dbh) and percent cover of tall (≥1.4 m ht) shrubs were measured within 10, 0.05-ha plots that were spaced systematically within each treatment. Coarse woody debris was measured within 4 × 20 m belt transects originating at gridpoints that were spaced at 50-m intervals throughout treatment areas. Depth of leaf litter and duff was measured at three locations along each of three randomly oriented, 15-m transects originating at grid points that were spaced at 50-m intervals throughout treatment areas. Canopy openness was measured beginning in 2002 (prior to canopy disturbance and thus considered pre-treatment) at two randomly selected points within each treatment during summer (leaf on) using a spherical densiometer held at breast height.

2.5. Statistical analyses

We had occasional problems in 2001 and 2002 with traps that were snapped and moved, likely by raccoons, and in 2003 substantial numbers of traps in all experimental blocks and treatments were disturbed. We also occasionally found animals dead in traps. Therefore, we used closed mark-recapture models described by Otis et al. (1978) to estimate population size in each experimental unit each year. We conducted the analysis using the program CAPTURE (Rexstad and Burnham, 1999). These models are designed to account for unequal catchability and trapping effort in the mark-recapture data. Prior to analysis, the mark-recapture datasets for each treatment stand and year were edited by first deleting animals found dead in traps on trapping nights 1–8. In 2003, we deleted data from any trap night in which more than 2/3 of the traps were dysfunctional. Thus, individuals that were captured only on dysfunctional nights were eliminated from CAPTURE datasets. We then ran CAPTURE for any dataset that had at least 20 different individuals captured and let CAPTURE choose the most appropriate estimator. For datasets with at least 10 individuals but less than 20, we used the Model M_h estimator as a default, as it is known to be the most generally robust estimator in CAPTURE.

There were seven datasets with <10 individuals, and for these we used the number of different individuals captured as the capture estimate. A common technique for these very small sample size situations is to use an average capture probability derived from comparable units with adequately large sample size (in our case we used units from the same year and block), to adjust for imperfect detectability. Because capture probabilities were large, this procedure resulted in estimates that were the same as the number of individuals captured. We calculated the final estimate of population size by taking each estimate from CAPTURE and adding to it the number of dead animals and the number that were discounted when we eliminated dysfunctional trap nights during the trapping session. Only white-footed mice were included in statistical analyses because sample sizes of other small mammal species were low.

The basic experimental design of our experiment was a two-way randomized block design with repeated measures over years. However, because treatments were initiated incrementally in different years, a straightforward standard analysis was not possible. We therefore performed two separate ANOVAs, each of which used data from two consecutive years to test for differential effects of treatments implemented between those 2 years. For each ANOVA population estimates were first natural-log transformed to reduce heteroscedasticity and to estimate effects on a multiplicative scale. Then for each experimental unit, we subtracted the estimate for the first year from the second year. This difference represents the relative change in the population in the unit between the 2 years. These differences were then analyzed with a simple randomized block ANOVA, followed by a Tukey multiple comparison procedure.

The first ANOVA used data from 2001 (all pretreatment) and 2002 (mechanical treatments in two of the four units in each block), and thus the only comparison of interest is whether units that received mechanical treatment (C–M) responded differently than those that remained as controls (C–C). In our analyses we considered the two experimental units per block (two C–C and two C–M in each of the three blocks, in 2002) to be independent replicates because treatments were assigned independently, and because white-footed mouse movement among the experimental units was minimal. The second ANOVA used data from 2002 and 2003, and four 'treatments' are involved: was C and remained C (C–C), was M and remained M (M–M), was C and changed to B (C–B), and was M and changed to MB (M–MB).

We used the same approach using adult (>15 g) (Wolff, 1985) body weight, sex structure (% male of total captures), and age structure (% adult of total captures) of white-footed mice, to test for differential effects of treatments implemented between 2001 and 2002, and between 2002 and 2003. Because a Student's t-test showed that body weights of males and females were similar, both males and females were included in adult body weight comparisons. We also used Student's t-tests to determine whether males outnumbered females, or whether adults outnumbered juveniles using data pooled across treatments and years, if ANOVA detected no differences among the treatments.

We used a two-way randomized block ANOVA to test for differences in white-footed mouse capture ratios (number captured:number of traps per microsite) between microsites (open or CWD), among treatments, and for treatment × microsite interactions during 2002 and 2003 (microsite data were not recorded during 2001). Because we detected no effect of microsite and no treatment × microsite interaction effect during either year we performed a Student's t-test to test for differences in capture ratios between the two microsites using pooled data for all treatments and years. In our tests we assumed that capture rates reflected microsite use, and that potential undetermined differences in detectability between the two microsite types would be consistent among treatments, and therefore not bias treatment comparisons.

In order to avoid bias that could result if the same individuals were captured multiple times and potentially at the same locations, we included only first-captures in analyses of microsite, body weight, sex structure, and age structure. We assumed that trap tampering (snapped, empty traps) affected traps at open and CWD microsites similarly. In our tests we assumed that potential differences in detectability between adults and juveniles, or males and females were consistent

Table 1
Mean (±S.E.) number of pre-treatment (2001), and post-treatment (2002 for M; 2003 for B, C, and MB) live trees and snags (per ha), percent cover of tall (>1.4 m ht) shrubs, forbs, and coarse woody debris, coarse woody debris density (pieces per ha), leaf litter depth (cm), and percent canopy openness, in three treatments: burned (B), mechanical understory felling (M), mechanical understory felling followed by burning (MB), and controls (C) (n = 3 each), Green River Game Land, Polk County, NC

Habitat feature	Measurement	Treatment				Randomized block ANOVA			
		B	C	M	MB	$F_{(2,6)\text{block}}$	P_{block}	$F_{(3,6)\text{trt}}$	P_{trt}
Live trees (ha)	Pre-treatment	568.7 ± 29.3	566.0 ± 10.6	602.0 ± 18.1	506.7 ± 33.8	0.73	0.5196	2.40	0.1662
	Post-treatment	539.3 ± 30.0 A	550.7 ± 15.0 A	588.0 ± 11.0 A	379.3 ± 43.5 B	1.26	0.3494	11.59	0.0066
Snags (ha)	Pre-treatment	62.7 ± 6.7	74.0 ± 8.3	55.3 ± 4.7	67.3 ± 14.1	0.73	0.5206	0.69	0.5916
	Post-treatment	72.7 ± 19.0 A	68.0 ± 9.0 A	52.7 ± 4.4 A	152.0 ± 25.3 B	0.32	0.7396	5.99	0.0309
Tall shrubs (%)	Pre-treatment	7.6 ± 2.9	14.2 ± 4.7	15.0 ± 3.9	9.6 ± 3.3	0.47	0.6445	0.79	0.5444
	Post-treatment	4.7 ± 2.8 A	20.0 ± 3.9 B	1.4 ± 0.1 A	0.2 ± 0.2 A	0.60	0.5763	10.80	0.0078
Forbs (%)	Pre-treatment	3.5 ± 1.3	3.6 ± 1.6	1.8 ± 0.9	3.3 ± 2.0	8.63	0.0172	0.91	0.4909
	Post-treatment	2.1 ± 0.4	2.8 ± 1.6	2.4 ± 1.3	2.0 ± 0.6	10.35	0.0133	0.07	0.9737
CWD (no./ha)	Pre-treatment	179.9 ± 38.1	138.9 ± 35.7	129.1 ± 10.5	197.9 ± 82.5	3.54	0.0964	0.73	0.5709
	Post-treatment	160.5 ± 52.6	128.1 ± 31.3	134.9 ± 22.6	141.0 ± 60.8	3.91	0.0817	0.17	0.9128
CWD (%)	Pre-treatment	1.2 ± 0.3	1.0 ± 0.3	1.1 ± 0.2	1.7 ± 0.7	6.71	0.0295	1.16	0.3998
	Post-treatment	1.2 ± 0.3	0.9 ± 0.3	1.0 ± 0.2	1.2 ± 0.5	3.74	0.0880	0.26	0.8518
Leaf litter depth (cm)	Pre-treatment	4.8 ± 0.3	5.0 ± 0.1	5.0 ± 0.2	5.1 ± 0.3	0.59	0.5847	0.20	0.8955
	Post-treatment	0.9 ± 0.1 A	4.2 ± 0.5 B	5.5 ± 0.2 C	0.5 ± 0.1 A	2.79	0.1389	116.14	<0.0001
Canopy openness (%)	Pre-treatment	6.2 ± 0.3	6.8 ± 1.0	8.3 ± 1.2	8.5 ± 2.6	3.41	0.1024	0.75	0.5614
	Post-treatment	2.6 ± 1.1 A	1.6 ± 0.4 A	3.0 ± 0.8 A	12.8 ± 5.0 B	2.09	0.2047	6.27	0.0280

F- and P-values for block effects (P_{block}) and treatment effects (P_{trt}) for each year. Differences among treatments within years are denoted by different letters within rows.

among the treatments, and therefore the comparison of proportion of captures among treatments would be unbiased.

We used randomized block ANOVAs to test for differences in habitat features, both pre- and immediately post-treatment.

3. Results

Fire intensities varied within and among sites but were generally moderate to high. Flame lengths of 1–2 m occurred throughout all burn units but in one block reached up to 5 m in localized spots where topography or intersecting flame fronts contributed to erratic fire behavior. Loading of fine woody fuels in MB, where the shrub layer was felled, was essentially double that in C and M. Measured temperatures were generally below 120 °C in B sites but sometimes exceeded 800 °C in MB. A detailed description of fire behavior in this study is given by Phillips et al. (2006).

Prior to treatment implementation, the number of live trees and snags per ha, percent cover of tall shrubs, CWD, leaf litter, and canopy openness were similar among treatments (Table 1). Post-treatment measurements (<1 year after) showed fewer live trees, more snags, and greater canopy openness in MB than in the other treatments. Leaf litter depth was reduced by burning in both B and MB treatments, and tall shrub cover was reduced by all fuel reduction treatments. However, post-treatment, both the number of CWD pieces and percent cover of CWD were similar among treatments and controls (Table 1).

We captured 990 individuals of eight rodent species a total of 2823 times during the 3-year study period. White-footed mice composed >79% of all captures (787 individuals and 1634 recaptures). Other species included golden mice (*Ochrotomys nuttali*) (88), eastern chipumunks (*Tamias striatus*) (54), pine voles (*Pitymys pinetorum*) (23), eastern woodrats (*Neotoma floridana*) (17), southern flying squirrels (*Glaucomys volans*) (15), hispid cotton rats (*Sigmodon hispidus*) (5), and a single woodland jumping mouse (*Napaeozapus insignis*) (Table 2).

Populations in experimental units increased 228% on average between 2001 and 2002, but there was no evidence of an effect of the mechanical treatment ($F_{1,8} = 0.21$, $P = 0.658$) (Fig. 1). From 2002 to 2003, all units again showed an average increase in relative population size, but there was evidence of a differential effect among treatments ($F_{3,6} = 3.62$, $P = 0.084$). Specifically, the average percentage increases from 2002 to 2003 in the treatments were: C–C (7%), M–M (29%), C–B (53%), M–MB (82%) (Fig. 1). Tukey's procedure ($\alpha = 0.10$) indicated that the increase in the M–MB units was significantly greater than the increase in control units. Absolute population estimates and standard errors for each treatment and year are presented in Fig. 1 to facilitate interpretation.

Table 2
Mean (±S.E.) number of individual small mammals captured 2001–2003 in three fuel reduction treatments: burned (B), mechanical understory felling (M), mechanical understory felling followed by burning (MB), and controls (C) ($n = 3$ each), Green River Game Land, Polk County, NC

Species	Year	Treatment			
		B	C	M	MB
Southern flying squirrel (*Glaucomys volans*)	2001	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
	2002	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
	2003	2.3 ± 1.9	0.7 ± 0.7	0.3 ± 0.3	1.7 ± 1.7
Eastern woodrat (*Neotoma floridana*)	2001	1.0 ± 0.6	0.0 ± 0.0	0.0 ± 0.0	0.7 ± 0.7
	2002	1.3 ± 0.9	0.0 ± 0.0	0.7 ± 0.7	0.0 ± 0.0
	2003	0.7 ± 0.3	0.7 ± 0.7	0.3 ± 0.3	0.3 ± 0.3
Woodland jumping mouse (*Napaeozapus insignis*)	2001	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
	2002	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
	2003	0.0 ± 0.0	0.3 ± 0.3	0.0 ± 0.0	0.0 ± 0.0
Golden mouse (*Ochrotomys nuttali*)	2001	2.3 ± 1.5	2.3 ± 0.9	1.0 ± 1.0	2.3 ± 2.3
	2002	5.0 ± 1.2	2.0 ± 1.0	2.3 ± 1.2	4.0 ± 3.1
	2003	1.3 ± 0.9	2.3 ± 1.5	4.0 ± 2.1	0.3 ± 0.3
White-footed mouse (*Peromyscus leucopus*)	2001	8.7 ± 2.7	8.0 ± 2.3	13.0 ± 2.9	12.7 ± 5.8
	2002	23.7 ± 8.2	19.3 ± 3.8	21.3 ± 5.3	32.0 ± 7.0
	2003	27.7 ± 7.0	17.3 ± 2.9	23.7 ± 2.3	55.0 ± 10.1
Pine vole (*Pitymys pinetorum*)	2001	0.0 ± 0.0	0.7 ± 0.7	0.0 ± 0.0	0.0 ± 0.0
	2002	0.7 ± 0.3	1.0 ± 0.6	1.3 ± 0.9	0.3 ± 0.3
	2003	0.0 ± 0.0	0.0 ± 0.0	0.3 ± 0.3	3.3 ± 2.0
Cotton rat (*Sigmodon hispidus*)	2001	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
	2002	0.0 ± 0.0	0.0 ± 0.0	0.3 ± 0.3	0.3 ± 0.3
	2003	0.3 ± 0.3	0.0 ± 0.0	0.0 ± 0.0	0.7 ± 0.7
Eastern chipmunk (*Tamias striatus*)	2001	0.3 ± 0.3	1.3 ± 0.3	0.7 ± 0.7	0.0 ± 0.0
	2002	4.3 ± 1.3	2.0 ± 1.5	1.3 ± 0.9	3.0 ± 2.5
	2003	1.3 ± 0.9	1.3 ± 0.9	0.3 ± 0.3	2.0 ± 2.0

In 2001 no treatments had yet been implemented (pre-treatment); in 2002 mechanical understory reduction treatments had been implemented in M and MB only; in 2003 all treatments had been implemented. Numbers are not adjusted for the number of trap nights.

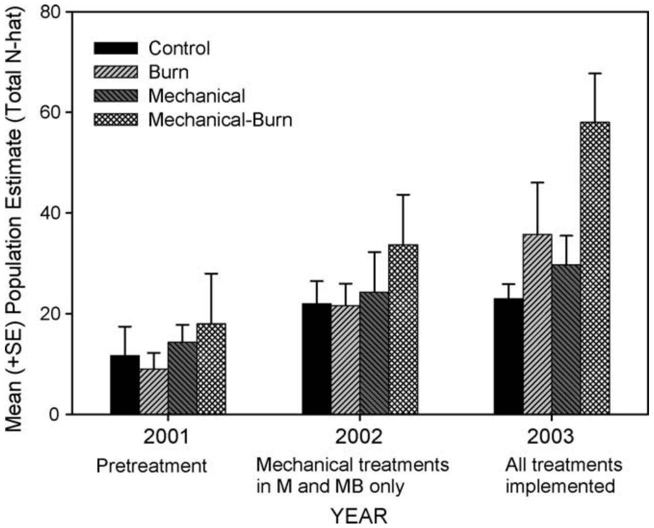

Fig. 1. Mean (±S.E.) population estimates of white-footed mice in three fuel reduction treatments: burned (B), mechanical understory felling (M), mechanical understory felling followed by burning (MB), and a control (C) ($n = 3$ each), Green River Game Land, Polk County, NC. Data for 2001 are pre-treatment; in 2002 only mechanical treatments had been implemented (in M and MB); 2003 after all treatments had been implemented.

We found no difference in white-footed mouse capture ratios (number of first captures:number of traps at a microsite) between the C and M treatments ($F_{1,18} = 1.99$, $P = 0.1756$) or between open and CWD microsites ($F_{1,18} = 0.10$, $P = 0.7575$), and no treatment × microsite interaction effect ($F_{1,18} = 0.09$, $P = 0.7637$) during 2002. In 2003 there was a significantly higher ratio of captures to the number of traps in MB than other three treatments ($F_{3,14} = 17.26$, $P < 0.0001$), but no effect of microsite ($F_{3,14} = 0.08$, $P = 0.7795$), and no treatment × microsite interaction effect ($F_{3,14} = 0.76$, $P = 0.5344$). A t-test using pooled data for both years and all treatments also indicated no effect of microsite on capture ratios (d.f. = 46, $t = 0.27$, $P = 0.7900$).

Body weights of adult male ($n = 409$; mean ± S.E., 19.4 ± 0.1 g) and female ($n = 245$; mean ± S.E., 19.0 ± 0.2 g) white-footed mice did not differ significantly (d.f. = 652; $t = -1.70$; $P = 0.0890$) (Table 3). Average adult body weight declined in 2002 from 2001 in both treatments, but declines were significantly greater in C–C than in C–M ($F_{1,8} = 9.14$, $P = 0.0165$). There was no evidence of a differential effect on adult body weight among treatments from 2002 to 2003 ($F_{3,6} = 2.20$, $P = 0.1889$) (Table 3).

The proportion of males captured was not affected by the M treatment alone (2001–2002) ($F_{1,8} = 0.57$, $P = 0.4703$), nor was a differential effect of the four treatments detected (2002–2003) ($F_{3,6} = 0.11$, $P = 0.9485$) (Table 3). Studywide, males outnumbered females 1.8–1 (d.f. = 70; $t = -8.25$; $P < 0.0001$).

The proportion of adult white-footed mice to juveniles was similar in all 3 years, and there was no evidence of an effect of the M treatment alone (2001–2002) ($F_{1,8} = 0.07$, $P = 0.9739$) or a differential effect of the four treatments (2002–2003) ($F_{3,6} = 0.57$, $P = 0.4703$) (Table 3). A t-test using data pooled across treatments and years indicated that adults composed the majority (mean ± S.E., 77.2% ± 2.2%) of captures (d.f. = 70; $t = 13.86$; $P < 0.0001$).

4. Discussion

Population estimates of white-footed mice generally increased during the experiment. Our results provide no evidence that mechanical treatment alone had any effect on population levels. However, the combination of burning and mechanical treatments caused a large proportional increase in population numbers. Burning of untreated units also caused a moderately large relative increase in numbers, but this average increase was not statistically significant. This suggests that the high tree mortality and associated increase in canopy openness in MB may have indirectly affected white-footed mouse populations.

Results of the few published studies that examine the effect of burns on *Peromyscus* spp. are inconsistent, likely due in part to high variability among sites and studies, and low treatment replication within most studies. Some studies report similar *Peromyscus* spp. abundance in both burned and unburned hardwood forest (Ford et al., 1999; Keyser et al., 2001). Kirkland et al. (1996) reported a lower abundance of

Table 3
Means (±S.E.) of adult body weight (g), age structure (percent adult), and adult sex ratio (percent male) of white-footed mice among three fuel reduction treatments: burned (B), mechanical understory felling (M), mechanical understory felling followed by burning (MB), and controls (C) ($n = 3$ each), Green River Game Land, Polk County, NC

Measurement	Year	Treatment			
		B	C	M	MB
Adult weight (g)	2001	20.8 ± 0.3	20.3 ± 0.9	20.0 ± 0.8	20.9 ± 0.7
	2002	18.0 ± 0.2	18.0 ± 0.3	18.5 ± 0.4	18.9 ± 0.3
	2003	19.4 ± 0.3	19.1 ± 0.3	20.4 ± 0.3	19.2 ± 0.2
Age structure (% adult)	2001	91.0 ± 5.9	93.1 ± 3.7	93.3 ± 3.6	100.0 ± 0.0
	2002	89.4 ± 5.6	93.0 ± 4.3	94.8 ± 3.8	91.0 ± 2.5
	2003	90.6 ± 5.2	93.5 ± 3.3	94.9 ± 5.1	86.9 ± 5.5
Adult sex ratio (% male)	2001	84.1 ± 9.6	68.3 ± 5.6	65.5 ± 7.8	84.8 ± 10.9
	2002	58.6 ± 2.3	61.7 ± 1.7	64.6 ± 3.5	66.0 ± 3.5
	2003	61.3 ± 3.1	64.3 ± 6.6	54.6 ± 10.2	58.5 ± 3.4

Data for 2001 are pre-treatment; in 2002 only mechanical treatments had been implemented (in M and MB); 2003 after all treatments had been implemented.

white-footed mice in a burned (with reduced shrub cover) than in an unburned deciduous forest in the central Appalachians during some months. In contrast, Krefting and Ahlgren (1974) reported that higher densities of deer mice in burned than unburned mixed conifer-hardwood forest sites. Higher *Peromyscus* spp. populations on burned sites have been attributed to better visibility and abundance of seed, a food source for the mice, after reductions in litter cover and depth (Tester, 1965; Ahlgren, 1966). In our study litter depth was reduced in both B and MB, which also had the largest relative increases in population numbers.

Other studies also have shown a positive response by *Peromyscus* spp. to silvicultural treatments that resulted in more canopy openness (Ford et al., 2000; Carey and Wilson, 2001; Fantz and Renken, 2005), possibly due to increased availability of fruit (Blake and Hoppes, 1986; Greenberg et al., in press) and flying/foliar arthropod (Whitehead, 2003) food resources. In our study, the heterogeneous forest canopy and other structural features in MB may have provided optimal habitat or increased fruit, seed, or arthropod food resources for white-footed mice.

Abundance is not necessarily a reflection of habitat quality (Van Horne, 1983). However, potential indicators of habitat quality, such as adult body weight, sex structure, and age structure of white-footed mice, were generally unaffected by the fuel reduction treatments. Adult body weight declined from 2001 to 2002, but more so in untreated units (C–C) than in C–M, where shrubs and small trees were mechanically felled. Possibly, conditions created by the M treatment promoted an increase in food resources such as fruit or arthropods. Overall, adult body weights were similar to those reported for white-footed mice in southern Virginia (Wolff, 1985). We found more males than females (1.8–1), but this difference was also unrelated to treatments. Wolff (1985) reported that males outnumbered females by 1.7–1 in southern Virginia.

In our study, capture rates for traps placed adjacent to CWD were similar to capture rates for traps placed in the open. In contrast, Greenberg (2002) found that white-footed mice preferentially used (as measured by capture rates) CWD, but relative densities were similar among sites with different levels of CWD loading. Other studies also suggest that white-footed mice use CWD preferentially for travel, orientation, foraging, nesting, and refuge sites (Kirkland, 1990; Tallmon and Mills, 1994; McCay, 2000). Because cover features were altered by all fuel reduction treatments, we might have expected to see greater use of CWD in some or all treatments, to avoid visual or auditory detection by predators (Barnum et al., 1992). However, an absence of a microsite or microsite x treatment interaction effect indicated that CWD was not an important factor governing the microdistribution of white-footed mice in our study. Further, white-footed mouse populations (and capture ratios) were higher in MB post-treatment, but CWD loadings were no greater in MB than in the other fuel reduction treatments. This further suggests that CWD was not a major influence on population differences among fuel reduction treatments in our study.

5. Conclusions

White-footed mouse population estimates increased significantly in response to mechanical understory felling followed by prescribed fire. There was no evidence of a response to mechanical understory felling alone, and some suggestion of increases due to prescribed fire alone, although this could not be strongly supported statistically. We acknowledge that the sensitivity of our experiment was reduced by a myriad of common sources of uncontrollable experimental and sampling error, but we believe the experiment provides strong support for a causal positive relationship between burning and small mammal population response. Whereas all fuel reduction treatments resulted in understory reductions, high tree mortality in MB resulted in other changes in habitat structure. These included higher snag density and canopy openness that may have contributed directly or indirectly (through changes in food supply) to the white-footed mouse response. We found no indication that treatments affected white-footed mouse fitness (body weight) or demography (age or sex ratios). Trap placement adjacent to CWD did not affect white-footed mouse capture rates within or among treatments, indicating that the mice did not use CWD preferentially or change their use patterns in response to fuel reduction treatments. Land managers should understand possible effects of different fuel reduction treatments on white-footed mouse populations, as these mice are an important component of the fauna and food chain of deciduous southern Appalachian forests.

Acknowledgements

This is contribution number 104 of the National Fire and Fire Surrogate Research (FFS) project. This research was funded by the USDA Forest Service through the National Fire Plan. Although the authors received no direct funding for this research from the U.S. Joint Fire Science Program (JFSP) the research was greatly facilitated by the JFSP support of existing FFS project sites. Special thanks to Aimee Tomcho, Joe Tomcho, Jill Baker, David Watts, Gabrielle Graeter, and many others for field assistance. Thanks also to Stanlee Miller for help with identifications, Dean Simon for overseeing treatment implementation, and Ross Phillips and others for collecting and providing habitat data. Mark Ford, Karen Kinkead, and two anonymous reviewers provided suggestions that improved earlier versions of this manuscript. We thank the North Carolina Wildlife Resources Commission for their support, for permitting this study to be conducted on state Game Lands, and for their assistance in conducting prescribed burns.

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