INITIAL RESPONSE OF WOODRATS TO PRESCRIBED BURNING IN OAK WOODLAND

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ABSTRACT: Despite increased incidence of wildfire in oak (*Quercus* spp.) woodland and the increased use of prescribed burning as a management tool, few empirical data are available on the effects of burning to wildlife in oak woodlands. During the fall of 1993 to the fall of 1997, we studied the effects of prescribed burning on woodrat (*Neotoma fuscipes*) relative abundance in oak woodland at Camp Roberts in coastal-central California. In October 1997, the California Department of Forestry and Fire Protection (CDF) conducted an experimental burn over 500 acres of oak woodland at Camp Roberts, including half of our 1-ha study plots. On average, nearly half of each plot was burned (range = 30-66%). CDF classified the burn as light to medium intensity. Of the 14 vegetation variables measured before and after the burn, 5 were reduced significantly: cover of grasses, green forbs, and shrubs, abundance of coarse woody debris, and number of woodrat houses. In the falls of 1993 to 1996 (pre-burn) and again in the fall of 1997 (post-burn), relative abundance of woodrats was measured by live trapping. Trap success and proportion of recaptures did not differ pre- vs. post-burn. Additionally, the sex ratio of woodrats was not different after the burn in fall 1997 between burned and unburned plots. Our results indicate that low-intensity prescribed burns in oak woodland have minimal short-term (<1 year) effects on resident woodrats. Furthermore, any potentially negative effects of low-intensity prescribed burns on woodrats may be mitigated by positive, longer-term effects, such as reduced competition for oak regeneration, rejuvenation of vegetation, and reduction of fuel load and the threat of catastrophic wildfire. We are continuing to monitor the woodrat abundance to evaluate responses over the longer term.

Keywords: dusky-footed woodrat, *Neotoma fuscipes*, oak woodland, prescribed fire, *Quercus*.

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Oak- (*Quercus* spp.) dominated woodlands occur over approximately 3 million ha of California. They encircle the Central Valley and extend south along the coast to the border with Mexico. Oak woodlands are used by more species of terrestrial vertebrates than any other vegetative complex in the state (Ohmann and Mayer 1987).

During most of this century, oaks were viewed as a hindrance to livestock production and their removal was encouraged for perceived benefits of forage production (Rossi 1979). During 1945-1985, for example, 1 million acres of oak woodlands statewide were thinned or cleared for rangeland improvement (Bolsinger 1988). Over 80% of oak woodlands today are privately owned and most are under no professional management (Griffin and Muick 1984). Concern has been growing over the long-term sustainability of oak woodlands. California, the most populous state in the nation, is adding 6 million people per decade; these people increasingly are moving into the oak woodland-urban interface. Oak woodland valleys and foothills are desirable places to live, and, since 1970, thousands of acres have been converted to suburban and semiurban developments (Bolsinger 1988). Demands on woodlands for intensive agriculture, recreation, and simply as viewsheds all have increased greatly.

Current uses and demands on woodlands create both a fear of and a need for fire in woodlands. The area burned by wildfire in California has increased in recent decades (California Department of Forestry and Fire Protection [CDF] 1996). The severity of several recent wildfires in California oak woodland has prompted interest in the use of prescribed burning to limit fuel accumulation. In addition, the use of prescribed burning for forage and shrub management is a common livestock management practice on oak rangelands. The California Fire Plan (CDF 1996) outlines the increased use of prescribed burning in oak woodlands to limit both fuel buildup and the severity of wildfires and as a management tool for livestock. For example, landowners and agencies annually conduct prescribed burns over 2,000-8,000 acres of oak woodland on California's central coast.

Fire in woodlands affects vegetation composition and structure (Tester 1965, Ffolliott and Bennett 1996), as well as wildlife populations. Numerous studies have investigated the effects of fire on plant germination and succession in California chaparral (e.g., Horton and

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Kraebel 1955, Hanes 1971, Keeley and Keeley 1981) and Australian ecosystems (e.g., Christensen and Kimber 1975, Fox 1982). Other researches have studied effects of fire on animals in chaparral (e.g., Lawrence 1966; Longhurst 1978; Quinn 1979, 1983; Wirtz 1979, 1982), but no studies have documented the effects of fire on vegetation structure in oak woodland and its effects on wildlife diversity and abundance. Our purpose was to document numerical responses of terrestrial vertebrates to experimental prescribed burning in oak woodland. Here, we report the change in selected vegetation components in response to an experimental prescribed burn in well-structured oak woodland at Camp Roberts, California, and the initial responses of the woodrats in the burned areas. The data are from fall 1993 to fall 1996, before the burn, and fall 1997, a few weeks after the prescribed fire was conducted.

STUDY AREA

Camp Roberts is a military facility of the Army National Guard and is located in northern San Luis Obispo County (the north portion of Camp Roberts is in Monterey County) 18 km north of Paso Robles, California (Fig. 1). The facility comprises 17,800 ha, of which roughly 7,200 ha is classified as oak woodland (Camp Roberts EMAP 1989). The dominant tree species of the oak woodland is blue oak (*Quercus douglasii*) with a variable contribution of coast live oak (*Q. agrifolia*). Where it occurs, understory is dominated by toyon (*Heteromeles arbutifolia*), redberry (*Rhamnus crocea*), bigberry manzanita (*Arctostaphylos glauca*), ceanothus (*Ceanothus* spp.), and poison oak (*Toxicodendron diversilobum*). Small ($\leq$0.25 ha) patches of chamise (*Adenostoma fasciculatum*) occur on two of the study plots. On the woodland floor, wild oats (*Avena* spp.), bromes (*Bromus* spp.), and fescues (*Festuca* spp.) predominate. Common forbs include deerweed (*Lotus scoparius*), filaree (*Erodium* spp), fiddlenecks (*Amsinckia* spp.), and hummingbird sage (*Salvia spathacea*).

METHODS

Experimental Design

In summer of 1993, using topographic maps and ground reconnaissance, we selected nine sites within the southern half of Camp Roberts where there was the least potential for interference with military activities and where the most heavily treed areas occur. Site selection criteria were: >16 ha in size; accessible by road; dominant vegetation of blue oak, and estimated canopy cover of >50%. Within each of the 9 sites, compass and tape measure were used to lay out a 17 X 17 sampling grid (289 intersections) with 15-m spacing. Each square sampling grid was 5.8 ha. During fall of 1993 to fall 1996, these 9 study plots were sampled to evaluate habitat relationships for birds, small mammals, and herpetiles and to establish a baseline for these taxa in relatively unperturbed, well-structured oak woodland. In the summer of 1996, we arranged with CDF to conduct experimental burning over some of the same areas where we had conducted sampling. In the winter of 1996, to increase the number of replicates of burned and unburned plots, we selected 10 smaller sampling grids using existing corners (8 X 8 sections) of six of the nine 17 X 17 grids established in 1993. We also established 12 additional 8 X 8 sampling grids for a total of 22 trapping grids, 11 in burned and 11 in unburned areas. Our criteria for establishing the 8 X 8 grids was that they be in areas of dense oak woodland and that they be at least 35 m from any other 8 X 8 grid to ensure reasonable independence of sampling plots. Our small mammal trapping data from 1993-1996 indicate that movements by woodrats of greater than 45 m were uncommon. Of the >8,000 captures of >2,000 adult and juvenile woodrats recorded during 1993-1996, <20 individuals were captured at locations $\geq$45 m apart (Tietje and Vreeland 1997, unpubl. data).

Field Sampling

Vegetation. Vegetation was sampled on 130, 10-m diameter circular plots (Fig. 2) in August to September of 1997 before the burn was conducted, and again after the burn in October to November of 1997. Vegetation sampling was conducted only in the burned areas; because we wanted to document reduction in vegetation characteristics due to the fire, we did not sample control plots.

Ground, shrub, and canopy cover were measured where each of the four cardinal directions (N, S, E, and W) intersecting the 10-m-diameter circle (Fig. 2). Ground cover was sampled by ocularly estimating the percent of a 1-m^2 frame occupied by grass and grass-like dried forbs (considered grasses, e.g., yellow starthistle [*Centaurea solstitialis*]), green forbs (e.g., hummingbird sage, deerweed), woody stems, woodrat houses, tree trunks, duff, moss, fine woody debris (FWD, 2.5 cm diameter and $\geq$1 cm diameter within the frame), rock ($\geq$10 cm diameter), and bare soil. Duff depth was taken with a ruler (nearest 0.5 cm) at the center of the square frame. Shrub cover was sampled using a 2.5-m shrub sampling pole (Griffith and Youtie 1988) (3.1 cm diameter) held vertically by an observer at each of the intersections of the 10-m-diameter circle (Fig. 2). A second observer stood at the center of the circle and counted the number of 20-cm-long alternating black and white rings within 5, 5-ring levels (0.5 m each level) on the pole that were at least 25% covered by vegetation. Canopy cover was measured at each of the intersections of the 10-m-diameter circle with a spherical, concave densiometer (Lemmon 1956) held by the observer at approximately waist height and facing the center of the 10 m circle (Fig. 2). Both shrub and tree canopy vegetation was included in canopy measurements.

Coarse woody debris (CWD, ≥10 cm diameter within the circle and 1 m long), snags (≥10 cm dbh and ≥1.5 m tall), and woodrat houses that occurred within the 10-m-diameter vegetation plots were counted (Fig. 2). To facilitate the relocation of these vegetation components after the burn, the approximate location by compass bearing and distance within the circle were recorded during pre-burn sampling.

Woodrats. We trapped woodrats on the 8 X 8 burned and unburned grids before the experimental burn during 4 Oct-18 Nov 1993, 10 Oct-11 Nov 1994, 9 Oct-10 Nov 1995, 14 Oct-15 Nov 1996, and after the burn during 6-31 Oct 1997. During fall 1993 to fall 1996, before the 12 new 8 X 8 grids were added, we trapped on only

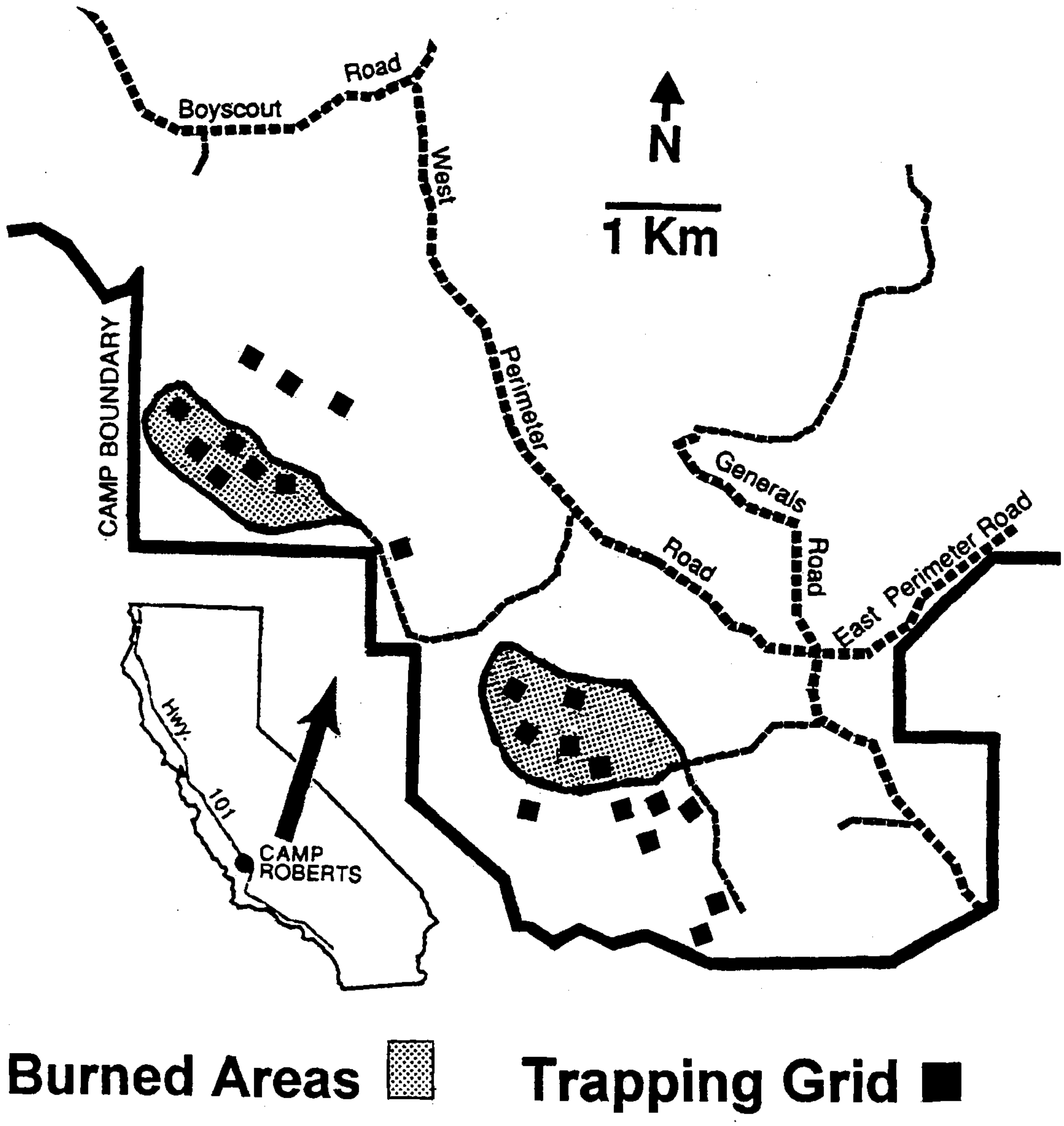

Figure 1. Camp Roberts study area showing the location of 22 small mammal trapping grids used to assess response of dusky-footed woodrats to prescribed burning in oak woodlands, Camp Roberts, California, fall 1993-1997. Trapping grids not drawn to scale.

6 burned and 4 unburned grids. During fall of 1993 to fall of 1997, a single Sherman live trap (7.6 X 9.5 X 30.5 cm) was placed at each of the 64 grid intersections. Traps were baited with "COB" horse feed (rolled corn, oats, and barley laced with molasses) and checked for five consecutive days. Captured animals were tagged and released. We recorded species, capture location, and sex. Total trapping effort during fall 1993-96 was 12,800 trap-nights pre-burn (10 plots) and 7,040 trap-nights (22 plots) post-burn during fall 1997. While conducting the post-burn trapping in fall 1997, we recorded whether burning had occurred within 1 m of each of the 64 intersections on each of the 11 burned grids. This information was used to assess the percent of burned plots that burned.

Data Analysis

Vegetation and trapping data were summarized at the plot level. We treated plots within the two burned areas as independent replicates. Burned and unburned plots were not paired. Because sample size was low ($n \leq$ 22) and data were non-normally distributed, we used non-parametric analyses. Changes in vegetation were analyzed using Wilcoxon paired-sample tests (Zar 1984). Abundance of vegetation elements was compared for burned plots before and after the burns. Number of captures per 100 trap-nights was analyzed by analyses of variance using Type III sums of squares on ranks (Conover and Iman 1981). Because we had pre-burn data, a year X treatment interaction effect in the ANOVA model was evidence for an effect caused by prescribed fire. We used the log-likelihood ratio, G, in contingency tables to test for differences in proportions of recaptures before and after the fire and for differences in proportions of male vs. female woodrats on burned and unburned plots. We used the convention of $\alpha = 0.05$ for statistical significance.

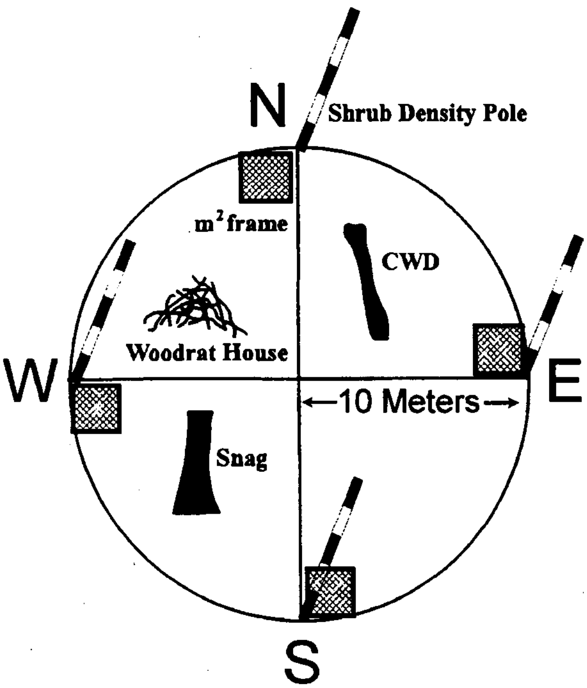

Figure 2. Schematic representation of a 10-m-diameter vegetation sampling plot on which vegetation variables were recorded to assess change after prescribed fire, Camp Roberts, California, August - November 1997. Hatched squares are 1 m^2 frames in which ground cover was ocularly estimated. Vertical (leaning) poles represent locations of shrub (understory) density and canopy cover estimates. Habitat elements in the circle (coarse woody debris [CWD], snags, woodrat houses) were counted. Figure not drawn to scale.

RESULTS

Burning Intensity

Flame length usually was <1 m. Exceptions occurred over small areas in dense, open grass, in two, ca. 0.25 ha patches of decadent chaparral, and in rare instances where individual, mature oak trees burned. In chaparral, flame height reached 20 m for several minutes. The burned area of the 1-ha plots ranged from 30% to 66% (mean = 46%). Almost no visible heating effects were observed on the soil surface, except for the areas around some larger logs and some woodrat houses that burned. Duff either was unburned or consisted of partially burned or charred pieces of twigs and leaves, mostly intact and recognizable. The fire only lightly affected the shrub layer and the tree canopy. CDF rated this fire intensity as low-to-medium range.

Vegetation Changes

Ground Cover. Of 13 ground cover variables measured (cover of grasses, duff, bare soil, FWD, green forbs, mosses, rock, woodrat houses, tree trunk, shrub stems, duff depth, number of woodrat houses, number of pieces of CWD), reduction was significant in four: grasses were reduced from 31% to 10% cover ($P = 0.001$); green forbs were reduced from 1.0 to 0.4% cover ($P = 0.020$); number of woodrat houses was reduced from 79 to 56 ($P = 0.004$); and the number of pieces of CWD was reduced from 237 to 153 ($P = 0.002$) (Figs. 3 and 4). Change in 3 other ground cover variables approached significance ($0.10 > P > 0.05$): FWD was reduced from 3 to 2% cover ($P = 0.069$); duff depth was reduced from 2.7 to 2.2 cm ($P = 0.067$); and the number of snags was reduced from 68 to 59 ($P = 0.062$) (Figs. 3 and 4). The area within the 1-m square frame that was burned averaged 28% and 5 pieces of new CWD were created by the fire on the 130, 10-m vegetation sampling plots (Figs. 3 and 4). These

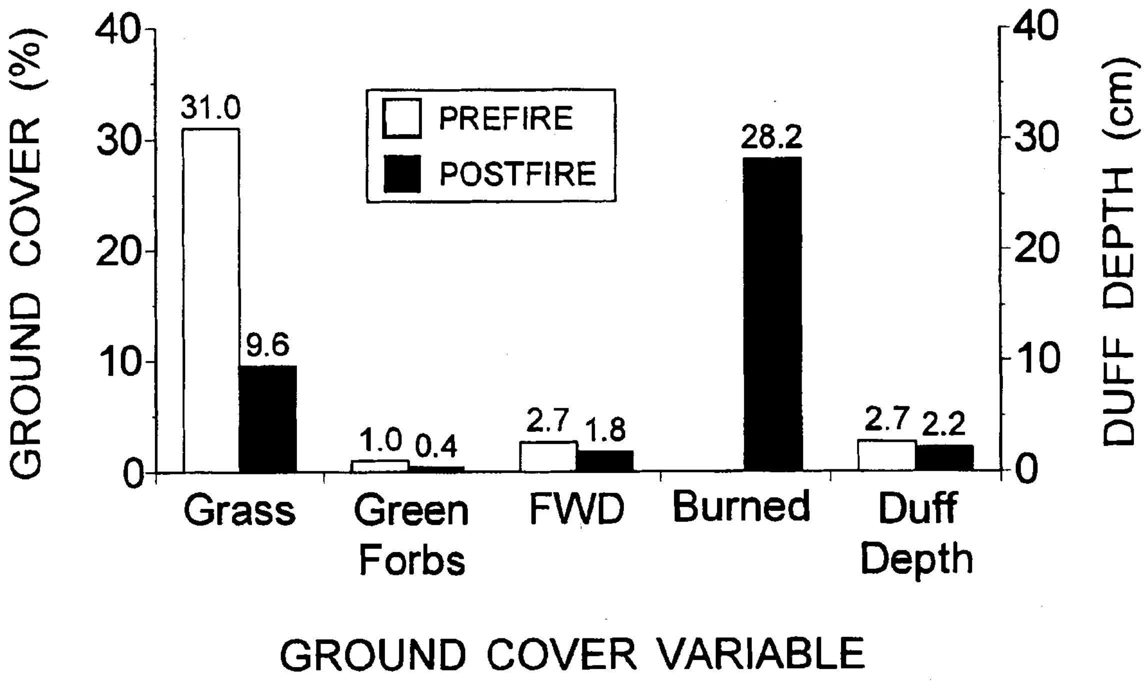

Figure 3. Response to prescribed fire of 5 ground cover variables ocularly estimated in 1 m² frames in oak woodlands, Camp Roberts, California, August - November 1997. FWD is fine woody debris ≥2.5 cm long and ≥1 cm diameter (e.g., small branches, twigs, and bark).

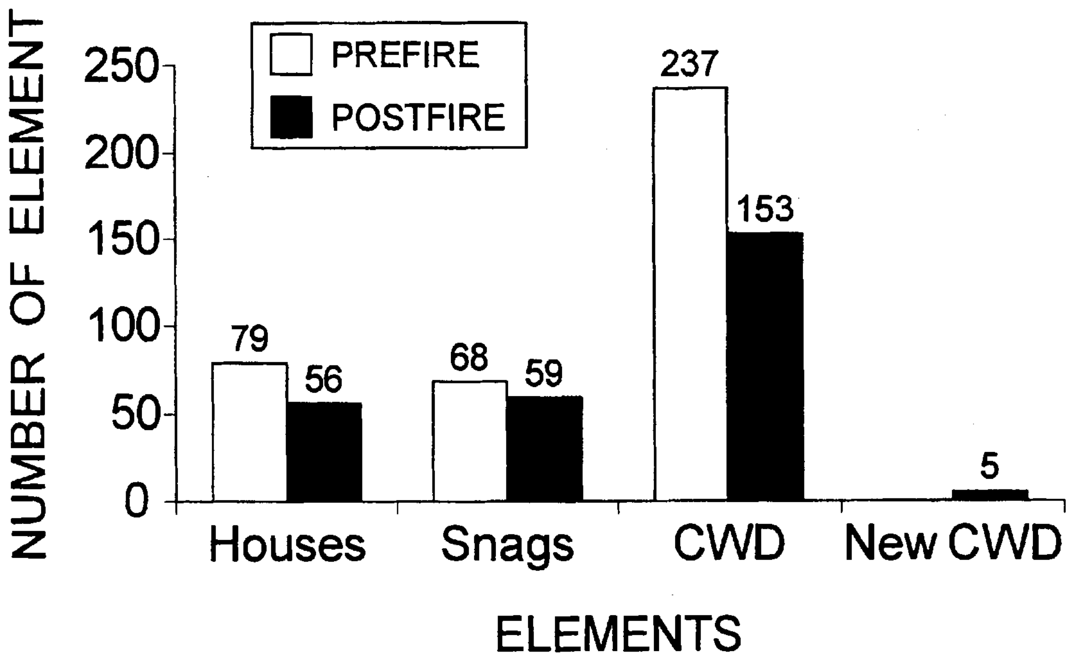

Figure 4. Response to prescribed fire of 4 habitat elements counted in 10-m radius vegetation sampling plots in oak woodlands, Camp Roberts, California, August - November 1997. CWD is coarse woody debris ≥10 cm diameter and ≥1 m long. New CWD is CWD created by fire (e.g., a mature, standing tree whose root wad was burned causing the tree to fall).

"created pieces" were large, usually an entire oak tree or a stem of a multi-stemmed oak tree that had fallen due to burning of the trunk or root wad.

Shrub and Canopy Cover. Shrub cover decreased significantly from 71 to 66% ($P = 0.003$; Fig. 5). Among the 5, 0.5-m levels on the 2.5-m shrub sampling pole, reduction was significant for the bottom of the pole (78 to 74% cover, $P = 0.027$) and the top of the pole (72 to 57% cover, $P = 0.004$). Canopy cover changed from 74 to 66%. This change approached significance ($P = 0.054$). Singeing occurred over 6% of the canopy (Fig. 5).

Woodrat Responses

Capture Success. Trap success (captures/100 trapnights) of woodrats captured on burn plots ($n = 6$) in the four trapping bouts before the prescribed burning (1993-96) ranged from 9.3 in the fall of 1993 to 24.4 in the fall of 1995 (Fig. 6). On control plots ($n = 4$), the range was 4.3 in fall of 1993 to 17.4 in the fall of 1995. In the fall of 1997, after the prescribed burns, trap success was 21.3 on the burned plots ($n = 11$) and 14.9 on the unburned plots ($n = 11$).

We detected no difference in capture success of woodrats on burned vs. unburned plots pre- vs. postburn (treatment X year interaction: $F_{4,80} = 0.12$, $P = 0.976$). Capture success differed among years (main effect of year: $F_{4,80} = 5.03$, $P = 0.001$; Figure 6), but no overall difference was detected between burned and unburned plots (main effect of treatment: $F_{1,80} = 3.04$, $P = 0.085$), indicating that woodrat relative abundance was similar on burned and unburned plots before and after the burn.

Recaptures. In our comparison of the proportion of woodrats recaptured on burn plots from the previous fall, we expected that more new individuals would be captured on burn plots after the burn compared to more individuals that were already tagged before the burn. However, we found no evidence that the proportion of recaptures was lower on burned plots after the burning treatment ($G_{10} = 14.521$, $P = 0.150$; Fig. 7).

Sex Ratio. To test if the burning treatment differentially affected male vs. female woodrats, we used postburn data (fall 1997) to examine the ratio of male to female woodrats captured on burn vs. control plots. The ratio of male to female woodrats on burned and control plots was similar in fall 1997 ($G_1 = 0.030$, $P = 0.862$; Fig. 8).

DISCUSSION

We were unable to document a change in relative abundance of woodrats immediately following a light- to moderate intensity prescribed fire in oak woodland. The low intensity of the fire probably was insufficient to

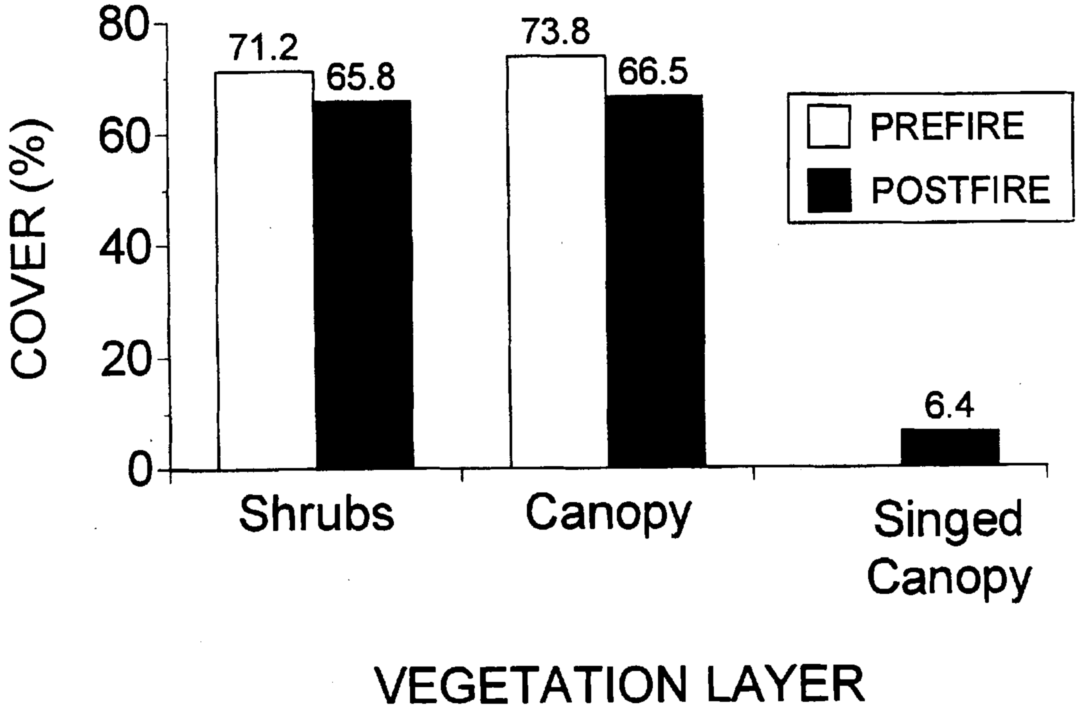

Figure 5. Response to prescribed fire of 3 habitat elements measured in 10-m radius vegetation sampling plots in oak woodlands, Camp Roberts, California, August - November 1997.

kill woodrats. In addition, the prescribed burn only moderately affected vegetation cover or structure and occurred over less than half of the two burn areas. On our study plots, woodrats were most abundant in the densest vegetation (Tietje et al. 1997, Tietje and Vreeland 1997), which was least affected by fire. Because grass cover was sparse, the fire generally did not carry into heavily shaded areas that lacked ground cover, but had well-developed shrub layers. The thick litter layer, shrub cover, or canopy in these areas also did not carry the fire. More intense fires may have burned portions of the denser vegetation, which could in turn have resulted in lower woodrat relative abundance in burned areas.

Although we observed statistically significant reductions in shrub cover and number of pieces of coarse woody debris, both important habitat elements for woodrats (Tietje et al. 1997, Tietje and Vreeland 1997), this reduction probably was not biologically significant for woodrats. Similarly, marginally significant reductions were observed in canopy cover and number of snags, but woodrat abundance was unaffected by these reductions. Lawrence (1966) also documented low mortality and loss of tree canopy cover after prescribed fire in chaparral, but documented high reductions in chaparral shrub cover, except for cover of poison oak, which increased. Poison oak was a major component of the shrub layer in the woodlands we surveyed at Camp Roberts and may display a similar trend in the future. Tester (1965) qualitatively noted that shrubs, young trees, and some large pin oaks (*Q. palustris*) were killed after fire in Minnesota oak savanna, but that mature bur oaks (*Q. macrocarpa*) were not killed. Conversely, Ffolliott and Bennett (1996) documented increasing mortality to oak trees (*Q. emoryi* and *Q. arizonica*) with increasing fire severity after natural wildfire where over 80% of trees on a high-severity fire site were root-killed.

The lack of studies against which to compare effects of prescribed burning on small mammals in oak wood-

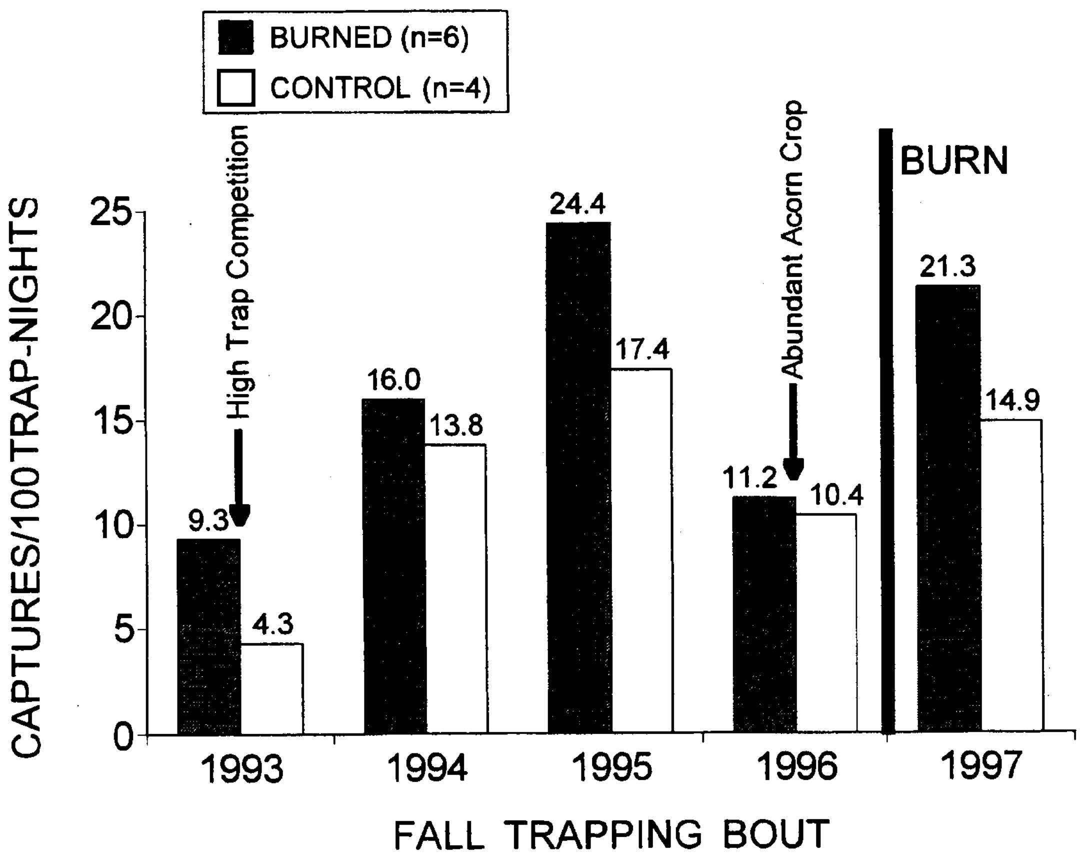

Figure 6. Relative abundance (trap success: captures/100 trap-nights) of dusky-footed woodrats before and after prescribed fire in oak woodlands, Camp Roberts, California, fall 1993-1997.

lands makes interpretation of some of our results difficult. Tester (1965) documented substantial post-fire increases in a rodent community in oak savanna in Minnesota. However, increases in abundance primarily were the result of immigration of new individuals into burned areas from surrounding, unburned vegetation. In addition, Tester (1965) concluded that the reduction in litter cover created favorable habitat for one abundant species (*Peromyscus maniculatus*) in his study area. Lawrence (1966) documented immediate post-fire reductions in small mammal abundance in chaparral, but abundance returned to pre-fire levels within a few months. Quinn (1983) observed significant decreases in captures of small mammals following prescribed fire in chaparral; one year after the fire, abundance was rebounding. Lawrence (1966) noted that chaparral-woodland species (*Peromyscus truei* and *P. californicus*) were unable to maintain their pre-burn numbers compared to grassland species (*P. maniculatus, Chaetodipus californicus, Reithrodontomys megalotis*) that recolonized burned areas. We recognize that responses of small mammal populations to fire may be different between oak woodlands and chaparral, because they have different structural characteristics. The small mammal community might also respond differently to fire.

Limitations of the Study

The lack of response of woodrats to prescribed fire in our study may have been the result of natural fluctuations in woodrat abundance (Fig. 6). Captures of woodrats varied from year to year (1993 to 1997). Significant differences in capture success of woodrats among years likely resulted from high trap competition during fall of 1993. Unusually high numbers of pocket mice (*Chaetodipus californicus*) and *Peromyscus* spp. were captured that year. (Tietje, unpubl. data) (Fig. 6). In addition, an unusually abundant acorn crop during fall of 1996 may have made the trapping bait less attractive to small mammals (Fig. 6). This pre-burn variability may have masked any negative response by woodrats to the fire treatment. However, captures of woodrats in fall 1997 were the second highest for any fall during our 5 years of trapping.

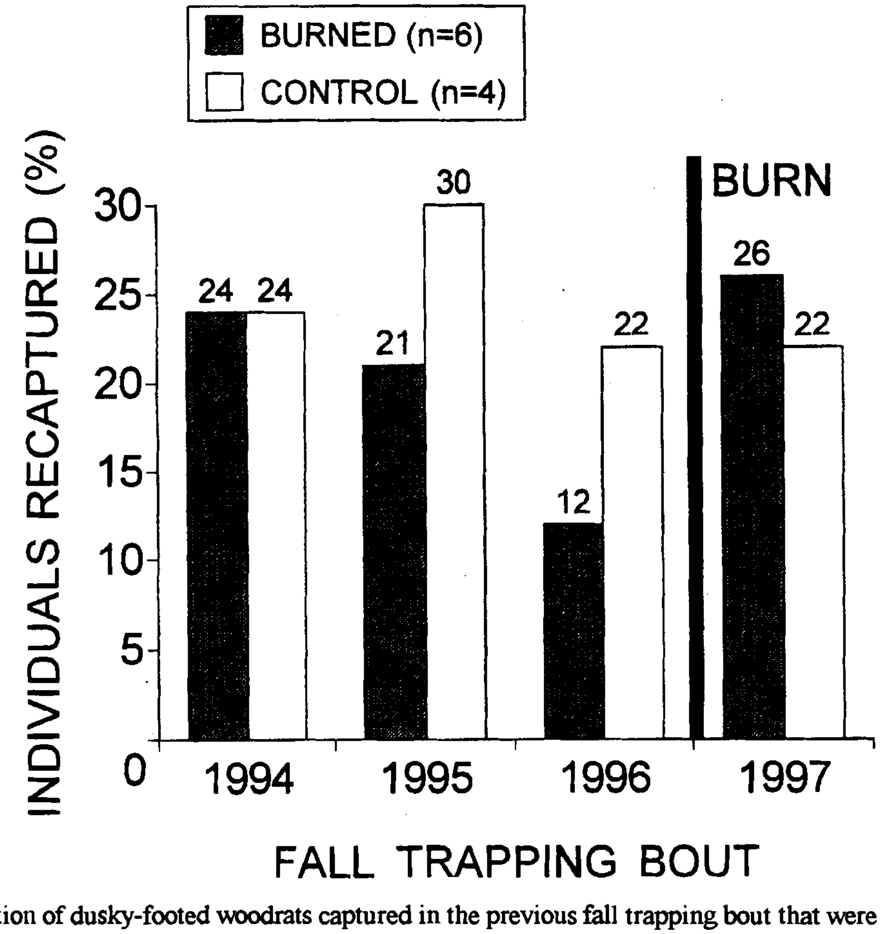

Figure 7. Proportion of dusky-footed woodrats captured in the previous fall trapping bout that were recaptured in the present fall trapping bout in oak woodlands, Camp Roberts, California, fall 1993-1997.

Our data were from only 1 post-fire trapping season. Despite the lack of immediate reduction in woodrat relative abundance after the fire, longer-term changes in woodrat abundance may result. For example, small mammals may be more vulnerable to predation after fires that reduce hiding cover (Tester 1965, Lawrence 1966) and that reduce fitness due to a loss in food supplies (Lawrence 1966). Alternatively, light- to moderate-intensity fires may rejuvenate shrubs and trees by reducing water competition from annual grasses (Douglas McCreary, Natural Resource Specialist, University of California Cooperative Extension, Marysville, Calif. pers. comm.). Shrubs and trees produce seeds (acorns, redberry and toyon berries) that are food sources for the frugivorous woodrat (Linsdale and Tevis 1951). However, incomplete or insubstantial loss of habitat components important to woodrats probably should not result in reduced woodrat abundance or individual fitness.

Management Implications

In our study, light- to moderate-intensity fires, which are typical in oak woodlands (Ben Parker, Forester, CDF, San Luis Obispo, Calif. pers. comm.), had no detectable short-term effects on woodrat populations. The use of fire as a management tool to limit fuel buildup and rejuvenate fire-adapted oak woodland should be encouraged. More intense prescribed burns and catastrophic wildfire, which remove larger amounts of downed wood, woodrat houses, shrub cover, and tree canopy, likely would affect populations. However, effects of intense fire on woodrat populations in oak woodlands remain unknown. Future studies of the effects of prescribed and natural fire on small mammals should experimentally investigate fire at different levels of intensity, over a larger scale, and for longer time periods. In addition, studies should investigate potential short-term changes in behavior, productivity, and increased susceptibility to predation as a result of loss of hiding cover, as well as long-term (>2 years) changes in population dynamics.

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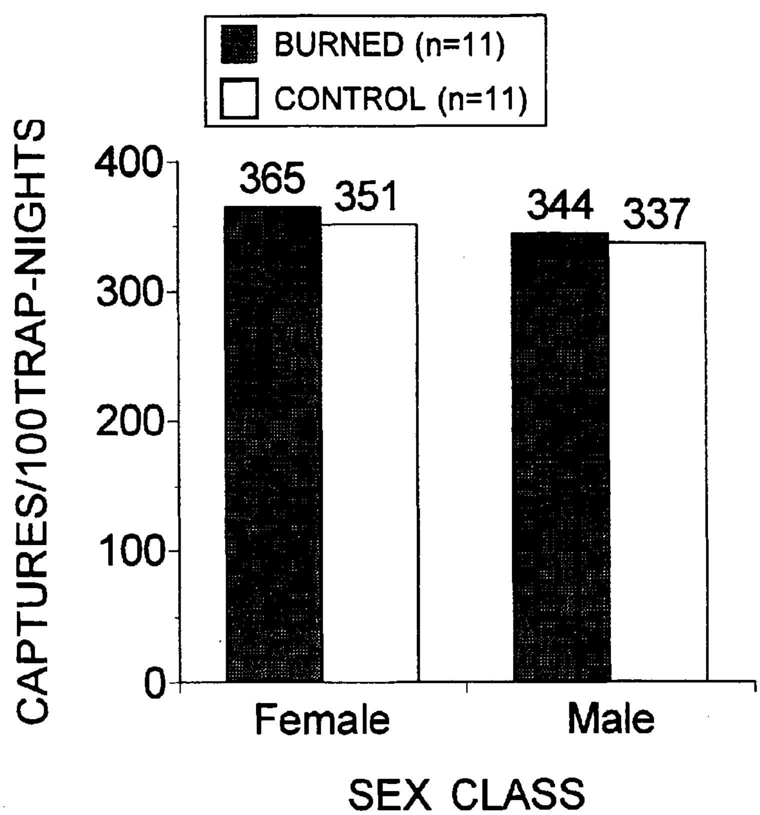

Figure 8. Relative abundance (captures/100 trap-nights) of female and male woodrats on burned and unburned trapping plots after prescribed fire in oak woodlands, Camp Roberts, California, fall 1997.

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