Numerical Response of Small Vertebrates to Prescribed Fire in California Oak Woodland

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Abstract.—Use of prescribed fire is increasing in California oak woodlands, but its effects on vertebrate wildlife are unknown. We conducted a light-intensity prescribed fire in mixed blue oak–coast live-oak woodlands in coastal-central California and assessed vegetation change and numerical response of small, nongame vertebrates to the fire. Four of 13 vegetation and habitat components that we measured were reduced significantly ($P < 0.05$) by the fire. We observed no change in relative abundance of small mammals, breeding birds, amphibians, or reptiles in response to the prescribed fire.

Introduction

Oak (*Quercus* spp.) woodlands are the most extensive vegetation type in California, covering approximately three million ha (Rossi 1979; Griffin and Muick 1984). In varying compositions, 10 native shrub and nine native tree species comprise numerous woodland habitat types. More vertebrate wildlife species use oak woodlands than any other vegetation type in California (Ohmann and Mayer 1987; Airola 1988).

California oak woodlands are fire-adapted, having evolved with fire during the past one million years. The dominant use of oak woodlands is rangeland for livestock production. Prescribed fire is used in oak woodlands as a livestock forage and fuel management tool (Griffin and Muick 1984). The California Fire Plan (California Department of Forestry and Fire Protection 1996) suggests increased use of prescribed fire in California's oak woodlands to reduce the severity of wildfires by limiting fuel accumulation and to manage livestock forage.

In the central coast region of California (roughly the area between Santa Barbara and San Francisco extending to the coast mountain ranges approximately 80 km inland), the California Department of Forestry and Fire Protection (CDF) conducts prescribed burns on 1,000–4,000 ha annually (Ben Parker, CDF, San Luis Obispo Ranger Unit, San Luis Obispo, CA, pers. comm.). The CDF presumes that prescribed fire benefits wildlife of oak woodland habitats, but no published research supports this assumption. Most research on effects of prescribed fire to wildlife and their habitats in California historically has been conducted in chaparral ecosystems (Lawrence 1966; Lillywhite 1974; Longhurst 1978; Quinn 1979, 1983; Wirtz 1979, 1982). Except for a manuscript from this project (Vreeland and Tietje 1998), we are aware of no published research on effects of prescribed fire to California oak woodland habitats or associated wildlife species. Our objective was to quantify vegetation change and numerical response of small, nongame vertebrates to a prescribed fire conducted in mixed oak woodlands of the central coast region of California.

Study Area

We conducted this study at Camp Roberts (CR), California, a facility of the California Army National Guard located approximately 30 km from the Pacific Ocean in northern San Luis Obispo and southern Monterey counties. Topography at CR varies from flat to gently rolling hills and steep (>45° slope) hills. The climate of the study area is Mediterranean, characterized by cool, wet winters and hot, dry summers. Annual precipitation averages 38 cm (66-year range=10.8–74.1 cm), falling almost exclusively as rain between November and March. Mean annual temperature averages 15.3°C. Summer high temperatures frequently approach 50°C; winter lows rarely fall below –6°C. Sheep grazing and military training occur in our study area, but are limited in extent, duration, and intensity. Fire has been excluded from the study area for >15 years (Brian Duke, CR Environmental Office, Camp Roberts, CA). Our study sites were on slopes <20°, on north- to east-facing aspects, and 300–500 m elevation.

Camp Roberts covers 17,000 ha, with 41% classified as oak woodland (Camp Roberts EMAP 1989). Three oak habitat types occur at CR: valley oak (*Q. lobata*), coast live-oak (*Q. agrifolia*), and blue oak (*Q. douglasii*), with considerable overlap between coast live-oak and blue oak types.

We used blue oak and mixed blue oak–coast live oak stands in the San Luis Obispo County portion of CR (Tietje et al. 1997; Tietje and Vreeland 1997*a*). Blue oak was the dominant canopy species with a variable contribution (0–45%) of coast live-oak. Where present, understory vegetation was composed primarily of toyon (*Heteromeles arbutifolia*), redberry (*Rhamnus crocea*), poison oak (*Toxicodendron diversilobum*), and bigberry manzanita (*Arctoctsphylos glauca*). Small (≤ 0.25 ha) patches of chaparral (*Ceonothus* spp. and *Adenostoma fasciculatum*) occurred on three study plots. Common forbs included hummingbird sage (*Salvia spathacea*), deerweed (*Lotus scoparius*), filaree (*Erodium* spp.), and fiddleneck (*Amsinckia* spp.). Wild oats (*Avena* spp.),

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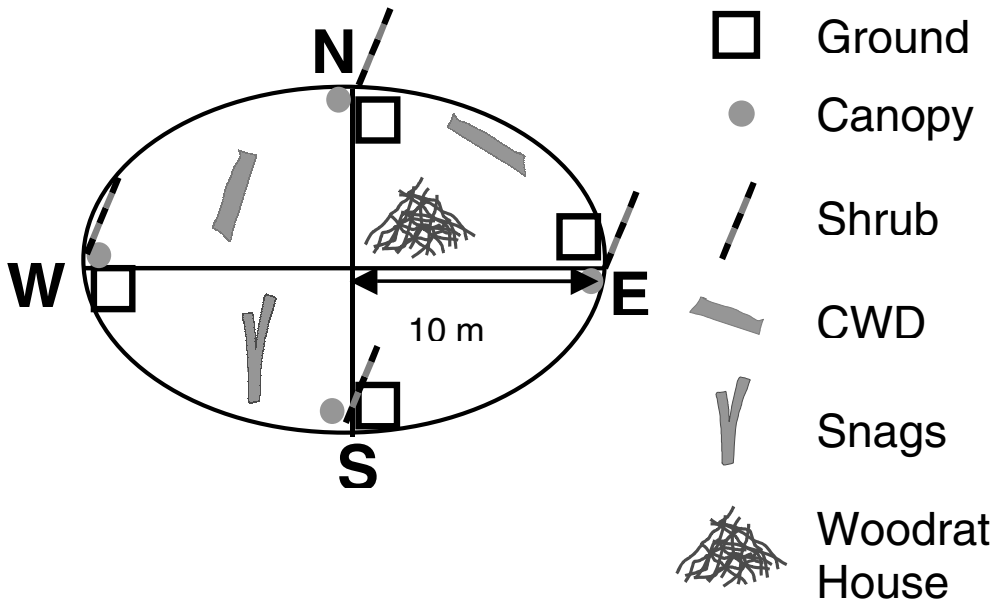

Figure 1.—Schematic layout of 10-m, circular plots used to sample vegetation characteristics in blue oak–coast live oak woodlands in San Luis Obispo County, California, 1997–1998. "Ground" represents ocular estimation of ground cover in 1 m^2 quadrats. "Canopy" represents measurement of canopy cover by concave spherical densiometer (Lemmon 1956). "Shrub" represents measurement of understory obstruction or shrub cover with a vegetation sampling pole (Griffith and Youtie 1988). "CWD," "Snags," and "Woodrat House" represent counting these elements within 10 m of plot centers. Not drawn to scale.

bromes (*Bromus* spp.), and fescues (*Festuca* spp.) dominated grassy openings of the woodland floor.

Blue oak sites were characterized by 40–60% canopy cover, <10% shrub cover, and >50% ground cover of exotic annual grasses. Mixed blue oak–coast live-oak sites were characterized by 60–90% canopy cover, 10–30% shrub cover, and a well-developed, thick leaf litter layer with abundant herbs, forbs, and <30% grass cover. Tietje et al. (1997) and Tietje and Vreeland (1997*a*) give detailed descriptions of vegetative and habitat characteristics of our study sites.

Methods

Prescribed Fire

We selected two areas to receive a prescribed fire treatment. One area (hereafter "Burn 1") was 80 ha, the other area (hereafter "Burn 2") was 120 ha. The California Department of Forestry and Fire Protection burned these areas on consecutive days in October 1997. The perimeter was burned first, then the interiors were ignited using drip torches, and accelerants delivered from helicopters. Surrounding unburned areas hereafter are referred to as "Ctrl" or "unburned areas."

Vegetation Sampling

We sampled vegetation in 11 1-ha plots throughout the burned areas within two months before and within 2–3 months after the prescribed fire. We randomly located five 10-m-radius sampling stations on each plot (Figure 1). At 10 m in the cardinal directions from the central point of each station, we measured canopy cover with a concave spherical densiometer (Lemmon 1956), understory obstruction (an index to shrub cover) with a vegetation pole (Griffith and Youtie 1988) graduated in five 0.5-m sections, and ocularly estimated ground cover within 1-m^2 frames. Within 10 m of the central point of each station, we counted the number of pieces of coarse woody debris (CWD, ³1 m long and maximum diameter ³10 cm), number of snags (³1.37 m tall and ³10 cm diameter at breast height), and number of dwellings (hereafter "houses") constructed by dusky-footed woodrats (*Neotoma fuscipes*).

Small Mammal Sampling

We live-trapped small mammals on 22 1-ha grids (8 x 8 grids, 15-m spacing) during May, 1997–1999. Plots were evenly divided between burned and unburned areas; six plots were in Burn 1 and five plots were in Burn 2. Plots were ³42 m apart. Spatial separation between plots was adequate because <4 animals moved among plots during the course of the study (W.D. Tietje, unpublished data). We trapped for five consecutive nights during 1997 and three consecutive nights during 1998 and 1999. We baited traps with horse feed ("COB": corn, oats, and barley laced with molasses), tagged animals with individually numbered ear tags, and released them

at site of capture. We handled animals in accordance with University of California, Berkeley Animal Use Protocol #R166-0199.

Breeding Bird Sampling

We conducted point-counts of breeding birds during March–April, 1997–1999. We used 86 50-m-radius point count stations located ³150 m apart. Stations were evenly divided between burned (13 stations were in Burn 1, 30 stations were in Burn 2) and unburned areas. We visited each station six times in each season. We rotated station visitations among start time, observer, and treatment. Two to four observers conducted counts in each year. We conducted counts for 10 minutes at each station between official sunrise and 1100 hours using standard breeding-bird survey protocols (Bibby et al. 1992).

Amphibian and Reptile Sampling

We counted amphibians and reptiles under 136 61-cm x 61-cm x 1.27-cm plywood coverboards (DeGraaf and Yamasaki 1992; Grant et al. 1992; Tietje and Vreeland 1997*b*) on each of nine 5.8-ha plots (8 x 17 grid, 30- x 15-m spacing). Five plots were in the unburned area, two plots each were in Burn 1 and Burn 2. We checked each coverboard once every 7–10 days during late January–early May in 1995–1999. We first deployed coverboards in 1994 to allow them to weather for 6–9 months, dissipating chemicals used in their manufacture that might affect their use by amphibians and reptiles (Grant et al. 1992). We had difficulty identifying slender salamanders (*Batrachoseps* spp.) (N. Scott, U.S. Geological Survey, Biological Resources Division, San Simeon, CA, pers. comm.). Pacific slender salamanders (*B. pacificus*) and black-bellied salamanders (*B. nigriventris*) potentially occur at CR, but cannot be distinguished in the field. Therefore we grouped them into one species category, hereafter "slender salamanders."

Analyses

We qualitatively described fire behavior and counted the number of grid intersections on small mammal trapping grids where fire burned to within 1 m to quantify areal coverage of the fire. In Vreeland and Tietje (1998), we conducted paired-sample *t*-tests to assess vegetation changes and considered differences significant at α=0.05. We repeat an abbreviated version of those vegetation results here, but direct the reader to Vreeland and Tietje (1998) for details on vegetation results. Because we were unable to randomly select burned areas and because treatment replication was low, our animal data cannot meet some assumptions of analyses of variance (ANOVA). Accordingly, rather than conduct inappropriate, low-power statistical tests, we used means and 95% confidence intervals (CI) to assess numerical response of vertebrates to the prescribed fire. We selected one species to represent each of the four taxa we monitored. We selected a representative species based on two characteristics: large relative abundance and potential for demonstrating change as a result of prescribed fire. True replication in our study is at the burn unit level. Relative abundance was calculated for each sampling unit (plot or point count station), then averaged for a treatment area (Burn 1, Burn 2, Ctrl). Because we have pre- and post-fire data on burned and unburned areas, pre-fire differences in relative abundance between burned and unburned areas are unimportant. The meaningful test of an effect of the fire is a comparison between relative abundance of a species on burned areas compared to unburned areas before and after the fire. This is analogous to the interaction term in a 2-way ANOVA. Differences between treatment means were considered significant if 95% of the CI did not overlap.

Results

Prescribed Fire

The California Department of Forestry and Fire Protection rated the prescribed fire as light- to moderate-intensity (3–4 on a 10-point scale). Flame height generally was <1 m except in a few areas of decadent grasses, chaparral, or dense accumulations of CWD. Few mature trees died. The fire was patchily distributed and carried better through blue oak stands with abundant grass cover than through mixed oak stands with thick leaf litter and dense canopy cover. Mineral soil was mostly unexposed except in small areas with high fuel load (dense CWD, chaparral). Area burned on 1-ha plots averaged 46% and ranged from 30 to 66%.

Vegetation

Understory obstruction and grass cover were reduced by 7% and 70%, respectively. Much of the reduction in understory obstruction was the result of reduction in grass cover that covered the lower 1 m of the vegetation pole. Excluding this grass cover probably would result in no statistical reduction in understory obstruction. Grass cover returned to pre-fire cover by one growing season after the fire. Canopy cover, number of snags, and leaf litter depth (an index to litter volume), did not change. Approximately six % of the canopy was singed. Number of woodrat houses was reduced by 30%. Number of pieces of CWD was reduced by 35%. We recorded five new pieces of CWD in the 55 10-m vegetation plots. These pieces of CWD were created when roots of mature oaks were burned during the fire, causing the bole to fall to the ground. Vreeland and Tietje (1998) report detailed vegetation responses to the prescribed fire.

Small Mammals

We captured nine species of small mammals during three years of live-trapping (Table 1). Woodrats, piñon

Table 1.—Relative abundance of small mammals, breeding birds with at least 100 detections, amphibians, and reptiles monitored in response to prescribed fire in blue oak–coast live oak woodlands in San Luis Obispo County, California, 1995–1999

Common name (scientific name)	Relative abundance[a]
Small Mammals	
Dusky-footed woodrat (*Neotoma fuscipes*)	26.49
Piñon mouse (*Peromyscus truei*)	6.61
Brush mouse (*Peromyscus boylii*)	4.40
California mouse (*Peromyscus californicus*)	3.06
California pocket mouse (*Perognathus californicus*)	1.34
California vole (*Microtus californicus*)	0.36
Merriam's chipmunk (*Tamias merriami*)	0.36
Deer mouse (*Peromyscus maniculatus*)	0.11
Heermann's kangaroo rat (*Dipodomys heermanni*)	0.01
Breeding Birds[b]	
Oak titmouse (*Parus inornatus*)	3,001
Dark-eyed junco (*Junco hyemalis*)	2,564
Lesser goldfinch (*Carduelis psaltria*)	1,842
Bushtit (*Psaltriparus minimus*)	1,529
Western scrub-jay (*Aphelocoma coerulescens*)	1,344
House finch (*Carpodacus mexicanus*)	1,336
Bewick's wren (*Thryomanes bewickii*)	1,221
Violet-green swallow (*Tachycineta thalassina*)	805
Spotted towhee (*Pipilo erythrophthalmus*)	643
Anna's hummingbird (*Calypte anna*)	517
California quail (*Callipepla californica*)	505
White-breasted nuthatch (*Sitta carolinensis*)	485
Hutton's vireo (*Vireo huttoni*)	480
Yellow-rumped warbler (*Dendroica coronota*)	480
Golden-crowned sparrow (*Zonotrichia atricapilla*)	459
Ash-throated flycatcher (*Myiarchus cinerascens*)	453
California towhee (*Pipilo crissalis*)	426
Western bluebird (*Sialia mexicana*)	405
Nuttall's woodpecker (*Picoides nutallii*)	394
Mourning dove (*Zenaida macroura*)	350
Ruby-crowned kinglet (*Regulus calendula*)	315
Blue-gray gnatcatcher (*Polioptila caerulea*)	285
Acorn woodpecker (*Melanerpes formicivorus*)	246
Orange-crowned warbler (*Vermivora celata*)	245
Wrentit (*Chamaea fasciata*)	204
Lawrence's goldfinch (*Carduelis lawrencei*)	177
Hermit thrush (*Catharus guttatus*)	141
Lark sparrow (*Chondestes grammacus*)	134
Northern flicker (*Colaptes auratus*)	121
Brown-headed cowbird (*Molothrus ater*)	111
Amphibians and Reptiles	
Western skink (*Eumeces skiltonianus*)	3.96
Western fence lizard (*Sceloporus occidentalis*)	2.19
Slender salamander (*Batrachoseps* sp.)	1.74
California legless lizard (*Anniella pulchra*)	0.47

continued

Table 1.—continued

Common name (scientific name)	Relative abundance[a]
Southern alligator lizard (*Gerrhonotus multicarinatus*)	0.35
Gopher snake (*Pituophis melanoleucus*)	0.22
California whipsnake (*Masticophis lateralis*)	0.22
Ring-necked snake (*Diadophis punctatus*)	0.14
Night snake (*Hypsiglena torquata*)	0.07
Common king snake (*Lampropeltis getulus*)	0.06
Chorus frog (*Hyla regilla*)	0.06
Side-blotched lizard (*Uta stansburiana*)	0.03
Western toad (*Bufo boreas*)	0.02
Common garter snake (*Thamnophis sirtalis*)	0.01
Monterey salamander (*Ensatina eschscholtzii*)	0.01

[a]Small mammals: captures/100 trap-nights; breeding birds: total observations during 3 years (86 total point count stations); amphibians and reptiles: observations/100 coverboard checks.
[b]Fifty-five species of birds were observed fewer than 100 times during 3 years and are not shown.

mice (*Peromyscus truei*), brush mice (*P. boylii*), and California mice (*P. californicus*) were the four most abundant species. Woodrats were the most frequently captured species (26.5 captures/100 trap-nights during 1997–1999). Relative abundance (captures/100 trap-nights) of woodrats on burned areas compared to the unburned areas did not change after the prescribed fire (Figure 2, Table 2). Relative abundance of woodrats was approximately 60% lower in 1998 and 1999 than in 1997, but this reduction was proportional among burned and unburned areas. Confidence intervals consistently overlapped treatment means and other CI within and among years.

Breeding Birds

We observed over 85 species of birds during the breeding season on all 86 point count stations (Table 1). Dark-eyed juncos (*Junco hyemalis*) were one of the two most frequently observed species (1.7 observations/point/visit during 1997–1999, respectively). Relative abundance (mean number of observations/point) of juncos on burned areas compared to unburned areas did not change after the prescribed fire (Figure 3, Table 2). We observed an approximately 80 % increase in juncos two years post-fire, but this increase was proportional among burned and unburned areas. Juncos consistently were more abundant on Burn 1 than Ctrl, but the difference remained consistent before and after the fire. Other confidence intervals were narrow, but consistently overlapped treatment means and other CI within and among years.

Amphibians and Reptiles

We observed four amphibian (two salamanders, one frog, one toad) and 11 reptile (five lizards, six snakes) species under plywood coverboards (Table 1). Among amphibians, only slender salamanders were observed in appreciable numbers (1.74 observations/100 coverboard visits during 1995–1999). Relative abundance (number of observations/100 board-visits) of slender salamanders was similar on burned areas compared to unburned areas before and after the prescribed fire (Table 2). Except during 1999, slender salamanders were more abundant on Burn 1 than Burn 2 or Ctrl, but this pattern of relative abundance remained consistent among years before and after the prescribed fire. Confidence intervals were broad, frequently included zero and other treatment means, and overlapped other CI within and among years.

Western skinks (*Eumeces skiltonianus*) were the most frequently observed reptile species (3.96 observations/100 coverboard visits during 1995–1999). Relative abundance (number of observations/100 board-visits) of western skinks varied among years and among treatment areas within and among years, but was similar on burned areas compared to unburned areas before and after the prescribed fire (Table 2). Skink observation rate appeared to be increasing on Ctrl, decreasing on Burn 2, and irregular on Burn 1, but these trends began 2–3 years before the fire, and were not caused by nor interrupted by the fire. Confidence intervals were broad, frequently included zero and other treatment means, and overlapped other CI within and among years.

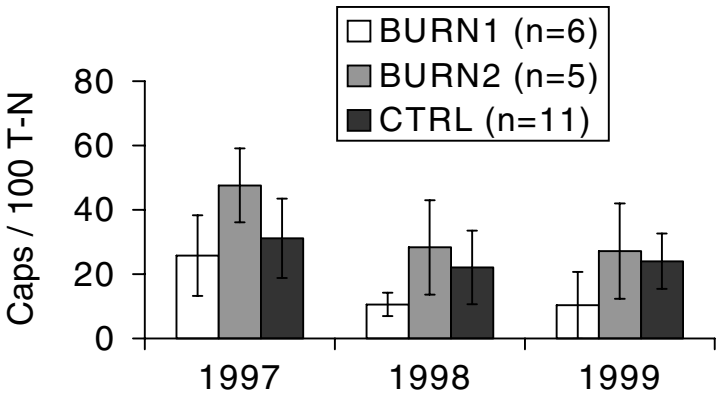

Figure 2.—Dusky-footed woodrat abundance (captures per 100 trap-nights) in May 1997–1999 before and after an October, 1997 prescribed fire in blue oak–coast live oak woodlands in San Luis Obispo County, California. Error bars are 95% confidence intervals.

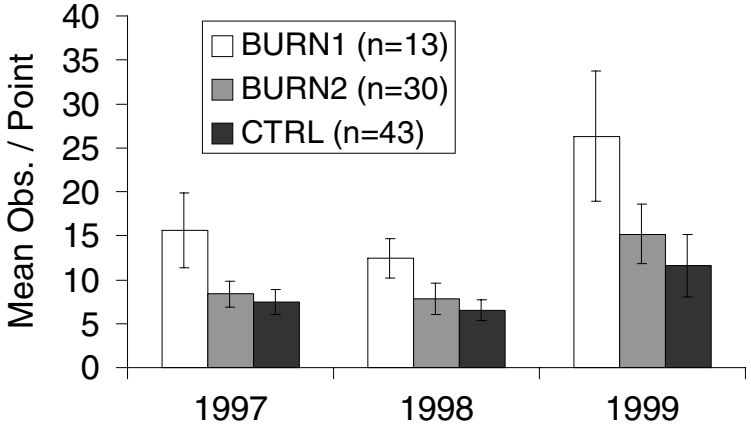

Figure 3.—Dark-eyed junco abundance (mean number of observations per point) in March and April, 1997–1999 before and after an October, 1997 prescribed fire in blue oak–coast live oak woodlands in San Luis Obispo County, California. Error bars are 95% confidence intervals.

Discussion

Conclusions

We detected no change in relative abundance of small mammals, breeding birds, amphibians, or reptiles after a light- to moderate-intensity prescribed fire in California blue oak–coast live oak woodlands. Intensity of the fire was limited by cool temperatures, moderately high relative humidity, low wind speed, and resulted in generally marginal or short-term changes to the vegetation characteristics we measured. Prescribed fire in oak woodlands may reduce resource competition from exotic annual grasses, stimulating shrub and tree health and vigor and, ultimately, mast production and overall habitat rejuvenation. Small vertebrates may respond to this habitat rejuvenation.

Coarse woody debris serves as hiding, breeding, and foraging habitat for small vertebrates. Reduction in CWD was substantial after the prescribed fire, but CWD may not be limiting for animals that use it. We counted 237 pieces of CWD on 1.73 ha of vegetation sampling plots (137 pieces/ha). Size criteria for CWD was appropriate to gauge use by most vertebrates but is a conservative estimate of CWD abundance, because small vertebrates will use smaller pieces of CWD. We observed slender salamanders using pieces of woody debris (e.g., dead coast live-oak bark) as small 20 cm x 20 cm x 3 cm, considerably smaller than the minimum size of CWD for this study.

Our casual observations detected only one instance (a woodrat) of direct mortality caused by the fire. Direct mortality likely was low, because most animals we monitored have subterranean access or are capable of other strategies that enable them to escape fire. Russell et al. (1999) reviewed over 12 published papers on effects of fire to herpetiles and concluded that direct mortality of these animals in fires is low.

No other published literature reports on the effects of prescribed fire to vegetation structure or vertebrate communities in California oak woodlands. Many researchers have compared burned and unburned areas after fire, but lack of pre-fire observations limits conclusions that can be drawn about effects of fire to animals. Some studies have combined other silvicultural treatments (e.g., thinning, forest harvesting) with burning. Distinguishing effects of burning from other treatments can be difficult, especially if treatments were not conducted under mutually exclusive study designs. Much study of prescribed fire and its effects on habitat

Table 2.—Relative abundance (mean, 95% CI) of dusky-footed woodrats, dark-eyed juncos, slender salamanders, and western skinks on burned and unburned areas before and after prescribed fire in blue oak–coast live oak woodlands in San Luis Obispo County, California, 1995–1999. The prescribed fire was conducted in October, 1997. Pre-fire years are 1995–1997 for salamanders and skinks, and 1997 for woodrats and juncos. Woodrats and juncos were not monitored in 1995 and 1996. See text for sample sizes

Year	Treatment	Dusky-footed woodrat	Dark-eyed junco	Slender salamander	Western skink
1995	Burn 1			8.6 (0–89.0)	2.6 (00.–13.3)
	Burn 2			0.8 (0–07.7)	4.0 (00.–08.3)
	Ctrl			0.6 (0–01.4)	3.3 (00.–06.7)
1996	Burn 1			2.0 (0–15.1)	2.0 (00.–12.3)
	Burn 2			1.0 (0–11.3)	3.4 (00.–10.4)
	Ctrl			0.3 (0–00.6)	3.5 (00.–07.0)
1997	Burn 1	25.8 (13.2–38.3)	15.6 (11.3–19.9)	2.9 (0–19.2)	1.7 (00.–07.7)
	Burn 2	47.6 (36.1–59.1)	08.4 (06.9–09.8)	2.4 (0–21.6)	2.7 (00.–09.7)
	Ctrl	31.2 (18.8–43.5)	07.4 (06.0–08.9)	1.0 (0–02.2)	4.8 (0.2–09.3)
1998	Burn 1	10.6 (07.0–14.2)	12.4 (10.1–14.6)	6.6 (0–52.4)	3.3 (00.–26.1)
	Burn 2	28.3 (13.6–43.0)	07.8 (06.0–09.6)	3.2 (0–32.1)	2.4 (00.–04.7)
	Ctrl	22.1 (10.6–33.6)	06.5 (05.4–07.6)	1.0 (0–02.2)	6.7 (0.3–13.0)
1999	Burn 1	10.3 (00.0–20.7)	26.3 (18.9–33.7)	1.8 (0–15.4)	2.8 (00.–18.9)
	Burn 2	27.2 (12.4–42.0)	15.9 (11.8–18.6)	2.8 (0–33.1)	2.4 (00.–15.6)
	Ctrl	24.0 (15.4–32.7)	11.6 (08.0–15.2)	0.4 (0–00.9)	6.7 (00.–12.7)

structure and wildlife have been conducted in other ecosystems (e.g., California chaparral, California dry pine, southeastern U.S. pine forests, prairie, northern temperate oak savannah, Appalachian hardwoods, and others). Species in different habitats and ecosystems likely respond differently to prescribed fire; therefore, comparisons of our results to other prescribed fire studies in other habitat types are hampered by these cross-habitat differences. Therefore, we restrict our comparisons with other prescribed fire studies to those that experimentally investigated effects of fire, rather than with studies that conducted post-fire observations on burned and unburned areas, on burned areas before and after fire without unburned controls, and wildfires that generally burn hotter and faster than prescribed fires.

Despite a 30% reduction in number of woodrat houses, we observed no change in relative abundance of woodrats because of the prescribed fire. Not all woodrat houses are occupied (Vreeland and Tietje 1999) and fire did not reach many houses in the densest habitats; therefore, few woodrats were forced to relocate to new areas or construct new houses after the fire. Tevis (1956) experimentally investigated effects of a site-preparation fire in logging debris (Pacific madrone [*Arbutus menziesii*] and tanoak [*Lithocarpus densiflorus*]) on numbers of small mammals in northern California Douglas-fir (*Pseudotseuga menziesii*) stands. Over 70 % of marked animals were not recaptured after the fire and probably died or emigrated from the burned area immediately after the fire. Most recaptured animals were caught at the periphery of the fire, suggesting they were not resident within the burn blocks before the fire or temporarily relocated after the fire. However, within three weeks after the fire, more new individuals were captured in the burned area than were captured before the fire. Tevis (1956) attributed the immigration by new individuals to the lack of resident mice created by the fire. We observed no such immigration to burned areas, probably because few or no residents were killed by the fire or emigrated from the burned areas, thereby leaving little unoccupied habitat.

Similarly, Tester (1965) trapped deer mice (*P. maniculatus* and *P. leucopus*) before and after a prescribed fire in burr oak (*Q. macrocarpa*)–northern pin oak (*Q. ellipsoidalis*) savanna in Minnesota and reported substantial immigration to burned areas within one month after the fire. Few animals captured before the fire were recaptured after the fire, but Tester (1965) could not speculate if absence of these animals suggested low survival during the fire, emigration, or high predation following the fire. Tester (1965) attributed increased use of burned areas to reduction in litter cover and depth, which were negatively correlated with presence of deer mice.

Masters et al. (1998) observed greater small mammal abundance, species richness, and diversity in Arkansas shortleaf pine (*Pinus echinata*) stands treated with prescribed fire than in unburned stands. Sullivan and Boateng (1996) observed dramatic decreases in deer mice and Oregon voles (*Microtus oregoni*) immediately following prescribed fire in coniferous forests of British Columbia, but animal abundance returned to pre-fire levels within four months following the fire. Red-backed voles (*Clethrionomys gapperi*) also increased substantially immediately following fire. Kaufman et al. (1988) observed a doubling of deer mouse abundance on burned areas 3–5 weeks following prescribed fire in Kansas tall-grass prairie. Ford et al. (1999) observed a decrease in abundance of pine voles (*Microtus pinetorum*) following prescribed burning in pitch pine (*Pinus rigida*) and oak stands in North Carolina, but abundance of nine other species did not change following fire. Lawrence (1966) trapped dusky-footed woodrats and observed immediate, but marginal short-term reductions in abundance following prescribed fire in chaparral and grasslands in California. Pattern of abundance between burned and unburned areas in both vegetation types remained similar for nearly three years following fire. Lawrence (1966) also observed decreased mass of piñon mice on burned areas following fire, but seasonal pattern of mass was similar between burned and unburned areas. Lawrence (1966) attributed decreased mass to 90% reduction in food abundance. Masters et al. (1998) and Kaufman et al. (1988) attributed increases in abundance to more favorable habitat or foraging conditions in burned areas.

Effects of prescribed fires conducted during fall may be less extensive or severe for breeding bird communities than for wintering bird communities in California oak woodlands, because breeding activity and behavior may commence during or after habitat rejuvenation initiates. Wilson et al. (1995) observed varied responses by breeding birds to prescribed fire in shortleaf pine–oak forests in Arkansas. Overall, bird densities and density of ground- and shrub-foraging and shrub-nesting species were greater in burned than in unburned stands following fire. Similarly, Salveter et al. (1996) observed no changes in abundance of neotropical migrant or resident songbird species following experimental prescribed fire in shortleaf pine and oak forests in Arkansas. Lawrence (1966) observed immediate reductions in abundance of chaparral birds following prescribed fire in California chaparral. Lower bird abundance persisted in burned areas compared to unburned areas through three years post-fire. Conversely, Lawrence (1966) observed increases in grassland birds and no significant change in abundance of oak woodland birds during the same fire in chaparral. Horton and Mannan (1988) observed a reduction in abundance of northern flickers (*Colaptes auratus*) and violet-green swallows (*Tachycineta thalassina*), an increase in mountain chickadees (*Parus gambeli*), and no change in 13 other species following a low-intensity prescribed fire in Arizona ponderosa pine (*Pinus ponderosa*) forests. We also monitored violet-green swallows and flickers in our oak-woodland study site, but observed no changes in their relative abundance, although detection rates were low. Salveter et al. (1996) suggested that some guilds (e.g., ground foragers, cavity nesters) may collectively demonstrate increases or decreases in abundance following prescribed fire. Reynolds and Krausman (1998) observed no reduction in relative abundance of breeding birds or wintering birds following experimental winter prescribed burning in Texas mesquite (*Prosopis glandulosa*) grasslands, and argued that species respond differently, potentially confounding guild analyses. Petersen and Best (1987) observed no immediate reduction in bird density following prescribed fire in sagebrush (*Artemisia* spp.) rangelands in Idaho, but documented increased species richness and density two years post-fire. Winter and Best (1985) observed a change in placement position of sage sparrow (*Amphispiza belli*) nests following prescribed burning in sagebrush (*Artimesia* spp.) rangelands in Idaho, suggesting functional responses to prescribed fire may be as important as changes in bird abundance.

Slender salamanders require moist microhabitats to maintain suitable respiratory conditions, and therefore are closely associated with habitat components (e.g., dense canopy cover, presence of CWD) conducive to maintaining these conditions. Western skinks are habitat generalists and ubiquitous, although they likely require CWD as refugia (Tietje et al. 1997; Tietje and Vreeland 1997*a*). Because CWD was abundant on our study plots, CWD probably is not limiting populations of slender salamanders, western skinks, or other herpetiles. Therefore, even large reductions in CWD may not result in detectable changes in relative abundance of small vertebrates.

We were unable to find experimental studies on effects of prescribed fire (without other silvicultural treatments) to amphibians and reptiles. Anecdotal and observational studies suggest that amphibians and reptiles may respond negatively (Enge and Marion 1986, McLeod and Gates 1998) or not at all (Enge and Marion 1986, Ford et al. 1999) to prescribed fire. Other observational studies suggest reptiles may exhibit different functional or behavioral activity in burned areas (Kahn 1960, Lillywhite and North 1974, Lillywhite et al. 1977). Observation rates of amphibians and reptiles typically are low in most studies, contributing to low measurement precision in response to treatments.

Because we did not mark amphibians and reptiles, we relied on observation rates to quantify their relative abundance. If herpetile numbers were reduced by the fire, these reductions could be masked by increased movement and subsequent increased observation rates, or increased use of coverboards with fire-induced loss of natural cover objects. Increased movement rates could result from changes in prey (e.g., small mammals,

invertebrates) abundance, foraging behavior, increased ease of mobility through ground vegetation reduced by fire, or genuine increases in animal abundance.

Caveats

Replication in our study was low because plots and point-count stations are pseudoreplicates of the burned (n=2) and unburned (n=1) areas. Therefore, true replication in our study was the burn blocks. Replicating prescribed fire studies at this scale is difficult for logistical reasons and budget constraints, but researchers should strive to replicate at the landscape level.

Sampling units were few for amphibian and reptile data (four burned plots, five unburned plots), contributing to low statistical power (broad CI) for these data. Furthermore, we did not mark individual birds, amphibians, or reptiles; therefore, our measures of relative abundance lack precision and may overestimate relative abundance. Although we individually marked small mammals, budget constraints restricted us to trapping only three nights during the two post-fire trapping sessions, which forced us to use capture rates rather than number of individuals as the metric for small mammal relative abundance. Data collected on number of individuals rather than capture rates has lower explanatory power.

Research on breeding birds suggests that bird density and relative abundance are poor measures of habitat quality and indicators for responses to treatments (Van Horne 1983, Vickery et al. 1992*a*,*b*). Engstrom et al. (1996) suggest that estimates of productivity and site fidelity would be better variables than density or relative abundance for demonstrating potential effects of prescribed fire to avian communities because relative abundance or density can be an inappropriate index to habitat quality.

Despite these shortcomings to our study, we believe our research represents important, initial information on effects of prescribed fire to blue oak–coast live-oak woodland habitat and associated small, nongame vertebrates. In addition, consistent responses to the prescribed fire by species in the four taxa we monitored suggest that our conclusions are appropriate.

We conducted the prescribed fire in October 1997, but assessed responses of small vertebrates 3–7 months after the fire. Monitoring animals immediately after the fire may have resulted in different effects to our study taxa. Our observations on the response to prescribed fire by small mammals during fall (October and November) and by wintering birds (January and February) followed similar patterns to the monitoring efforts we report here, however (Vreeland and Tietje 1998; W.D. Tietje, unpubl. data).

Research Implications

Studies of effects of prescribed fire must maintain an experimental design: before and after sampling on burned and unburned areas is required to document changes in animal numbers on burned areas compared to unburned controls (Russell et al. 1999). Replicating at landscape-level scales (>100 ha) is critical to assessing effects of prescribed fire, which necessarily occurs over large areas. Effects of prescribed fire, particularly for more intense fires, may last beyond two years post fire; long-term studies should evaluate effects of prescribed fire >5–20 years post-fire. Response of vegetation and animals may differ among fire intensity and among seasons in which prescribed fires are conducted (Engstrom et al. 1996, King et al. 1998). Further study of effect of season and intensity of fire is warranted. Researchers should mark individuals, preferably with individual marks, to precisely measure changes in numbers of individuals after prescribed fires. In addition to numerical responses, animals may demonstrate functional and behavioral responses to prescribed fire (Kahn 1960, Lillywhite and North 1974, Lillywhite et al. 1977). Researchers should assess functional response of animals and changes in behavior (foraging and nesting behavior, movement patterns, home range size) as well as numerical response (Winter and Best 1985, Engstrom et al. 1996). Estimates of reproductive potential (nest success [Engstrom et al. 1996], small mammal testes size), or individual status (health index [Lawrence 1966]) before and after prescribed fire would greatly enhance knowledge of effects of prescribed fire to small vertebrates (Engstrom et al. 1996).

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