

EFFECT OF FIREWOOD HARVESTING ON BIRDS IN A CALIFORNIA OAK-PINE WOODLAND

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Abstract: Despite a history of oak clearing and thinning in California, little is known about the effects of firewood harvesting on wildlife in oak woodlands. We studied the effect of firewood harvesting on population trends of birds during the breeding season in an oak-pine woodland in the foothills of the northern Sierra Nevada, California. During fall-winter of 1993-94, total tree basal area on 30 3.1-ha study plots was reduced by approximately 23% via removal of commercial-grade oaks, but all old-growth trees and trees with obvious nest cavities or woodpecker granaries were preserved. An additional 30 plots represented controls. We used point counts to derive a population index of birds 1 breeding season before and 2 breeding seasons after harvesting. We detected population shifts in 14 species (7 year-round residents, 4 breeding migrants, 1 migrant, 2 winter residents), of which 10 showed consistent population increases, 2 showed consistent population decreases, and 2 showed varying population changes in the 2 seasons after harvesting. Six of the species that increased on harvested plots appeared to respond to the presence of brush piles. Responses of other species were less clearly linked to a particular aspect of vegetation change. Although we found a negative effect of harvesting on only 2 species, the Pacific-slope flycatcher (*Empidonax difficilis*) and Hutton's vireo (*Vireo huttoni*), statistical power to detect anything less than total extirpation of many uncommon species was <0.8. Our results indicate that small-scale firewood harvests that reduce basal area by <25% and preserve nest cavities and granary trees have minimal negative short-term effects on most of the more common bird species present during the breeding season, but we caution that effects on uncommon species may have gone undetected. Further studies are necessary to evaluate the potential importance of site fidelity, brush-pile decay, and vegetation recovery in affecting species' responses over the long term.

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Oak woodlands cover almost 3 million ha in California and are distributed throughout the state (Allen et al. 1991). These woodlands encompass both structurally and floristically diverse vegetation types used by >110 bird species during all or part of the year (Verner 1980, Block 1989). Unlike California's coniferous forests that are largely under federal management, 84% of oak woodlands in California are privately owned and have received little professional management (Griffin and Muick 1990). Historically, owners of foothill oak-woodlands in California's Sierra Nevada and coastal mountain ranges have used their land for livestock grazing. Between 1945 and 1985, oaks were cleared from 480,000 ha in California, primarily to enhance forage production (Bolsinger 1988).

More recently, however, range clearing has declined, and urban or semiurban development has become the dominant cause of oak removal (Bolsinger 1988).

In addition to the complete loss of woodlands to urban conversion, firewood harvesting causes thinning of oak woodlands (Bolsinger 1988, Griffin and Muick 1990). Bolsinger (1988), for example, estimated that firewood harvesting in California caused the thinning of 25,000 ha of woodland annually. Firewood harvesting in oak woodlands will also directly affect vegetation structure and composition, which in turn affects the occupancy and resource use patterns of wildlife (Block and Morrison 1991, Block and Brennan 1993). Harvesting reduces canopy volume, changes the age and size distribution of trees, changes the relative frequency of tree species, and alters the understory by creating brush piles. In turn, vegetation changes may affect processes that influence community struc-

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ture (e.g., competition, predation, parasitism). These changes, in concert, likely have species-specific effects on habitat suitability for many birds (Block and Brennan 1993). For example, the creation of brush piles has been recommended as a management practice to provide cover and nesting habitat for game and nongame birds (Gorenzel et al. 1995). Conversely, fragmentation of woodlands has been associated with increased brood parasitism and declines of species dwelling in forest interiors (Robinson et al. 1995).

Although several descriptive studies (Manley 1988, Block 1989, Wilson et al. 1991, Wilson 1992) have examined variation in bird occupancy and resource use across existing vegetation gradients in California oak woodlands, no experimental studies have documented changes resulting from firewood harvesting. Our purpose was to document the population responses of species in the breeding-season bird community to an experimental firewood harvest in a Sierra foothill oak–pine woodland both 1 and 2 years after harvesting.

STUDY AREA

We conducted this study at the University of California, Sierra Foothill Research and Extension Center (SFREC), Yuba County, California, in the foothills of the northern Sierra Nevada range, about 25 km east of Marysville. Elevation on the 2,300-ha SFREC ranged from 75 to 625 m, with a general west to northwest aspect. Climate was Mediterranean, with mild, wet winters and hot, dry summers. Average annual precipitation during the study was 94 cm, most of which fell as rain between October and May (J. M. Connor, SFREC, unpublished data). Dominant trees were blue oak (*Quercus douglasii*), interior live oak (*Q. wislizenii*), and foothill pine (*Pinus sabiniana*), with fewer California black oak (*Q. kelloggii*), valley oak (*Q. lobata*), ponderosa pine (*P. ponderosa*), and California buckeye (*Aesculus californicus*). Most stands contained mixtures of the 3 dominant species, although some pure stands of blue oak or live oak were present. Dominant shrubs were poison oak (*Toxicodendron diversilobum*), coffeeberry (*Rhamnus californica*), toyon (*Heteromeles arbutifolia*), and buckbrush (*Ceanothus cuneatus*). A herbaceous understory consisted of annual and perennial grasses and forbs.

Historical land practices had modified the vegetation at SFREC. Except for 110 ha of

"natural area" fenced since 1971, SFREC was grazed by cattle. Since SFREC was purchased by the University of California in 1960, trees had been removed to increase forage production for cattle. Many areas had been partially thinned to varying percentages of residual canopy cover, and about 300 ha were completely denuded of woody vegetation. This study was confined to portions of SFREC with >10% canopy cover. We chose SFREC as a study site because it was among the few areas in California where a large-scale experimental harvest could be implemented, and because the vegetation and history of land use at SFREC were representative of many of California's privately owned woodlands.

METHODS

Experimental Design

We used a completely randomized design (Neter et al. 1990:36–37) with a treatment and control replicated 30 times. The treatment consisted of a reduction of approximately 23% of total tree basal area, achieved by uniformly thinning commercial-grade (minimum 15 cm root-collar diameter) blue oak and live oak. We chose this conservative harvest level because we knew that a heavy cut would affect many species, and we hoped to document a commercially viable harvest level that would affect fewer species. In addition, if we found no effect with this conservative harvest, we preserved the option of increasing the harvest in the future.

Experimental units were approximately 3.1-ha (100-m radius) circular plots centered on a single bird-counting station. In a previous study, Block (1989) established 105 counting stations at 300-m intervals along linear transects (900–4,200 m long) via a systematic random sampling design (Thompson 1992:119). Block (1989) chose this interval to ensure between-station independence of bird detections, over 90% of which were within 100 m of the count stations. We sampled vegetation at all 105 count stations during 1987–88 (Block et al. 1994). All trees and shrubs were measured on 3 circular 0.126-ha (20-m radius) plots placed within 80 m of each counting station via a systematic random sampling design. We used these vegetation data to rank counting stations by canopy cover, and then used only those stations with >10% canopy cover (99) to systematically select 30 treatment plots from the ranked list of stations (ev-

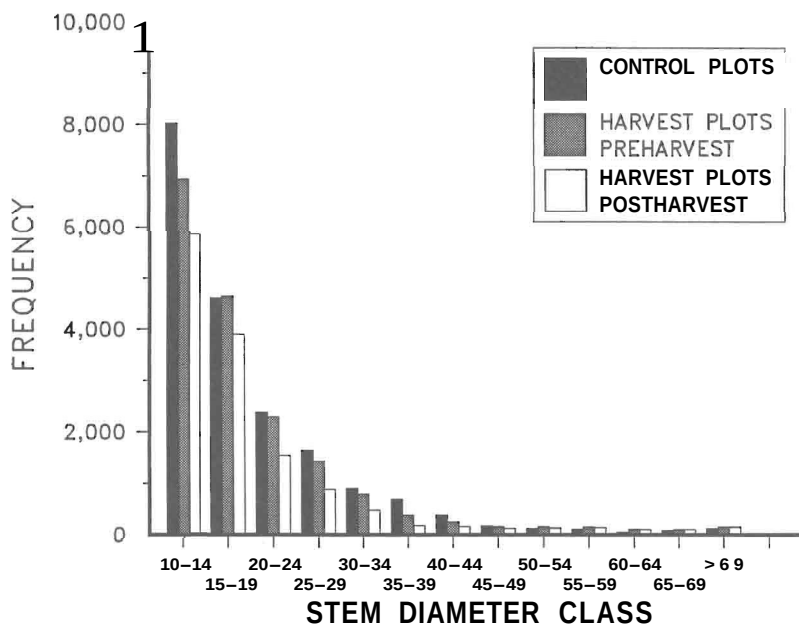


Fig. 1. Distributions of tree stems in control plots, harvest plots (preharvest), and harvest plots (postharvest) in a California oak-pine woodland, Yuba County, California, 1993-95.

ery third or fourth station, starting with a randomly selected station in the first group of 3). Our underlying goal in sampling systematically was to encompass the entire range of variation in bird density. We used canopy cover for ranking stations because we assumed it was the single measure of vegetation best correlated with the density of most bird species, and we intended to use it as a covariate in our analysis. Harvest levels were governed by a reduction of basal area rather than canopy cover because it was not feasible to estimate reduction of canopy cover when marking trees for harvesting.

After selecting treatment plots, the remaining 69 plots were initially considered controls. As pretreatment data collection progressed in 1993, it became apparent that numerous treatment plots could not be harvested because of difficult access, steep slopes, or overlap with other researchers' plots. Consequently, we had to make substitutions in the designation of treatment plots. To eliminate bias caused by these nonrandom substitutions, we evaluated each of the 69 control plots and removed from consideration 24 plots that could not have been thinned because of difficult access or steep slopes. Of the 45 remaining plots, we randomly selected 30 as controls to achieve a balanced design. We then assumed that the allocation of plots to treatments was reasonably random.

Mean preharvest canopy cover on these 60 experimental units was 35% (SD = 17), with a minimum of 12% and a maximum of 89%.

Harvesting occurred from mid-August 1993 to early March 1994. Commercial-grade blue and live oaks were harvested roughly in proportion to their occurrence (Fig. 1). Foothill pines and smaller oaks (<15 cm root-collar diameter) were only cut when required for access. Slash was piled and left at the sites, and stumps were allowed to resprout.

Marking and harvesting procedures were guided by standard operating practices for commercial harvests, as far as they could be deduced by discussions with SFREC personnel and local woodcutters. Harvesting also conformed to the recommendations for SFREC (Oak Management Recommendations for the Sierra Foothill Range Field Station, University of California, Davis, unpublished report). These guidelines prohibited harvesting on slopes >30%, within 30 m of the bottom of a draw or watercourse, on rocky outcrops, or within 15 m of a permanent road. They also specified the retention of all old-growth trees, acorn woodpecker (*Melanerpes formicivorus*) granary trees, snags, and trees with nest cavities. Old-growth trees were among the largest (by diameter at breast height) 5% of trees at the SFREC and usually had pronounced decadence and con-

tained many nest cavities. To conform with these guidelines, a uniform thinning was not always possible. On treatment plots with steep slopes and draws (10 plots), we had to thin more heavily in other areas of the plot to achieve an overall basal area reduction of 23%.

Bird Counts

Field Methods.—We used a fixed-radius, circular-plot technique (Verner 1985) to count birds on the study plots. Observers counted birds at 5–18 stations on a given morning, depending on the length and proximity of the various transects. Each station was visited 10 times at approximately equal intervals from 31 March to 18 July. We chose this sampling period as a compromise between allowing time for many visits and limiting our sampling to the period when populations of most species were stable and individuals were conspicuous. Although the long sampling period (10 visits) allowed for inclusion of early and late-season transients in the sample, sporadic detections of transients had little weight in our analyses relative to repeat detections of breeding individuals on successive visits to a count station.

Counts began within 0.5 hr before sunrise and ended within 4 hr after sunrise. Upon reaching a station, observers waited 1–2 min to allow birds to resume normal activities after being disturbed. During the next 5 min, observers recorded the species of each bird detected and whether the bird was ≤ 100 m from the count station. At most stations, observers remained at the count station for the duration of the count. However, when visual obstacles were adjacent to the count station, observers moved up to 15 m from the count station to get a more complete view of the plot. Because of potential biases in counting birds, we did not conduct counts when the wind exceeded approximately 25 km/hr or when more than a light rain fell.

Six observers conducted point counts, but only 1 observer (PAA) was constant through all 3 years of the study. Two observers conducted counts in 1993 and 1994, and 5 observers conducted counts in 1995. The second observer in 1994 also participated in counts in 1995. However, because observer bias in conducting point counts can be considerable (Scott *et al.* 1981, Verner 1988), all observers were trained. No observers had point count experience prior to participating in this study, but all observers had at least 1 month to familiarize themselves with

the birds at the study area before participating in counts. During this familiarization period, all observers conducted simultaneous trial counts and distance estimation exercises (Kepler and Scott 1981). By the end of the familiarization period, all observers could identify, by sight and sound, all species present at the study area during the period. To aid in distance estimation, distance markers were placed at several count stations with good visibility. To avoid confounding interobserver differences with treatment effects in our analyses, we alternated the set of stations visited by a given observer from 1 count to the next so that all stations received approximately the same number of visits by each observer in a year. We reduced potential bias from time of day variation in bird activity (Robbins 1981) by reversing the direction each transect was sampled from visit to visit.

Data Analysis.—We categorized species as year-round residents, winter residents, breeding migrants, migrants, or incidental via the observations of Block (1989) and the field notes of observers in this study. Incidental species were rarely detected and not considered a regular part of the avifauna. Detections for 2 migrants, dusky flycatcher (*Empidonax oberholseri*) and Hammond's flycatchers (*E. hammondi*), were pooled because of difficulty in distinguishing these species in the field. We excluded from analysis all incidental species, all other species with < 10 total detections, and all species mostly detected flying over plots.

To strive for independence of detections between adjacent count stations, we discarded all detections > 100 m from count stations. All remaining detections were used in our analyses. The population trend of each species was the response variable used in our analyses. We calculated population trend for each year postharvest as the difference between the number of birds counted each year postharvest and the number counted preharvest. Within a breeding season, we considered each visit a subsample and computed the mean population trend across all visits to a station. In estimating population trends, we used these simple counts rather than the adjusted counts provided by the models of Buckland *et al.* (1993) because density comparisons among species were unnecessary, and estimates from simple counts relied on fewer assumptions (Verner and Ritter 1985). In addition, we had insufficient detections for most

Table 1. Means and standard deviations of basal area (m^2/plot) and stem density (no. stems/ plot) for each of 4 tree species and for all trees combined on 30 control and 30 harvested plots before and after an experimental firewood harvest at the Sierra Foothill Research and Extension Center, California. Mean percent reductions due to harvesting follow pre- and postharvest values. Plots were 100-m-radius (3.1 ha) circles. Harvesting occurred during the fall–winter of 1993–94.

Tree species	Contml plots		Harvested plots				Mean % reduction ^c
			Preharvest ^a		Postharvest ^b		
	x	SD	x	SD	x	SD	
Blue oak							
Basal area (m ² /plot)	13.7	6.7	15.6	7.4	11.6	6.1	29
No. stems/plot	322	177	331	187	259	172	26
Interior live oak							
Basal area (m ² /plot)	6.5	7.7	4.9	5.1	3.6	4.1	34
No. stems/plot	279	415	223	273	184	230	24
California black oak							
Basal area (m ² /plot)	0.3	1.2	0.7	2.6	0.6	2.5	4
No. stems/plot	2	6	15	58	15	58	<0.5
Foothill pine							
Basal area (m ² /plot)	4.5	6.6	2.6	3.2	2.5	3.2	8
No. stems/plot	34	40	23	28	22	26	7
All trees							
Basal area (m ² /plot)	25.7	11.2	24.5	8.9	19.0	7.3	23
No. stems/plot	642	447	596	342	483	300	22

^a Pretreatment estimates are based on 3 20-m-radius subplots measured within each plot during 1987–88.

^b Posttreatment estimates are calculated by subtracting the basal area and number of stems harvested from pretreatment estimates.

^c Mean percent reduction is calculated by dividing the amount harvested by the pretreatment estimate and multiplying by 100.

species to generate the detectability profiles required for the models of Buckland et al. (1993).

For each regularly occurring species (except hawks and eagles), we examined treatment effects by comparing mean population trends between harvested and control plots. We tested the null hypothesis that mean population trends were equal both 1 and 2 years postharvesting. We evaluated the effect of harvesting on the population trend of each species via separate 2-factor (treatment, year) analyses of covariance with a repeated measure on year and the preharvest count as the covariate (RM-ANCOVA; Norušis 1992:123–159). A repeated measures model was necessary because we measured the population trend in each of 2 years postharvest, and these measurements were not independent. Use of the preharvest count as a covariate effectively reduced error variance because we found an inherent tendency for stations with high preharvest counts to have negative population trends, and stations with low or zero preharvest counts to have positive trends. With preharvest count as a covariate in the model, including preharvest canopy cover as a second covariate had little effect on error variance; thus, it was not used as a covariate, as originally planned.

We tested the assumption of homogeneity of variance with the Bartlett-Box test, and we test-

ed the assumption of normality of residuals by visual inspection of normal probability plots (Neter et al. 1990:608–609, 614–618). When these assumptions were not met, we added a constant and applied square-root or logarithmic transformations as appropriate. When transformations failed to remedy violations, we applied the RM-ANCOVA anyway, as balanced designs are robust to violations of normality and homogeneity of variance (Neter et al. 1990: 623–624). All RM-ANCOVA models were analyzed via SPSS/PC+ (Norušis 1992).

We used 2 separate methods of assessing the statistical significance of an effect by controlling or not controlling the Type I error rate. First, we made no effort to control the experimentwise Type I error rate. Within each RM-ANCOVA model, an effect was judged significant when $P \leq 0.05$. Second, because we used many separate RM-ANCOVA models, we assessed statistical significance while controlling the experimentwise error rate at 0.2. Effects of the same type in separate RM-ANCOVAs were judged in ascending order of probability against a variable α via a sequential Bonferroni adjustment (Rice 1989).

For all species for which we found no evidence of a treatment effect, we computed power ($1-\beta$) to detect an effect half as large as the mean preharvest count. For a hypothetical spe-

Table 2. Mean preharvest breeding-season bird counts (no. birds detected/100 5-min point-count stations) and mean postharvest population trends (differences in count [no. birds detected/100 5-min point-count stations]) computed in each of 2 breeding seasons after firewood harvesting on 30 control and 30 harvested plots at the Sierra Foothill Research and Extension Center, California, 1993–95. Harvesting occurred in the fall–winter of 1993–94.

Common name	Scientific name	Preharvest mean count		Postharvest mean population trend ^a				P-values for effect ^b			Minimum effect % ^d
		1993		1994		1995		Treatment	Treatment × year	Power ^c	
		Control	Cut	Control	Cut	Control	Cut				
Residents											
California quail	<i>Callipepla californica</i>	19.3	16.3	6.3	22.0	7.0	30.0	0.014	0.454		
Wild turkey	<i>Meleagris gallopavo</i>	0.3	1.3	0.0	0.0	1.0	−0.7	0.978	0.305	0.06	373
Mourning dove	<i>Zenaida macroura</i>	41.3	42.3	4.0	7.7	−18.7	−14.0	0.302	0.898	0.89	
Northern pygmy-owl	<i>Clauisidium gnoma</i>	0.3	0.3	0.0	0.7	0.3	2.3	0.039	0.346		
Anna's hummingbird	<i>Calypte anna</i>	2.0	0.7	−1.7	0.3	−0.3	1.3	0.326	0.795	0.09	218
Northern flicker	<i>Colaptes auratus</i>	4.3	4.0	−0.3	0.0	−2.3	−3.0	0.688	0.587	0.28	101
Acorn woodpecker	<i>Melanerpes formicivorus</i>	72.7	93.0	−14.3	−17.7	−23.0	−39.7	0.670	0.228	0.98	
Downy woodpecker	<i>Picoides pubescens</i>	1.0	0.7	1.3	1.7	0.7	0.7	0.828	0.868	0.05	422
Hairy woodpecker	<i>Picoides villosus</i>	4.3	1.7	−1.3	1.7	−3.3	−0.7	0.967	0.863	0.23	114
Nuttall's woodpecker ^e	<i>Picoides nuttallii</i>	18.3	20.3	10.7	12.0	−1.3	−4.0	0.882	0.483	0.43	78
Western scrub-jay	<i>Aphelocoma californica</i>	26.0	26.7	1.0	−1.7	0.3	−3.3	0.521	0.886	0.54	68
Wrentit	<i>Chamaea fasciata</i>	26.3	18.0	4.3	9.3	−1.7	1.7	0.576	0.779	0.37	86
Plain titmouse	<i>Baeolophus inornatus</i>	129.0	152.3	50.0	31.3	45.0	23.3	0.474	0.816	>0.999	
Bushtit ^e	<i>Psaltriparus minimus</i>	30.0	34.3	16.7	89.3	11.3	1.3	0.017	<0.001		
White-breasted nuthatch	<i>Sitta carolinensis</i>	46.0	48.3	22.7	32.0	4.0	0.0	0.564	0.163	0.78	52
Bewick's wren	<i>Thryomanes hewickii</i>	46.7	47.0	14.0	30.7	7.0	29.3	0.006	0.519		
Western bluebird	<i>Sialia mexicana</i>	20.0	23.3	14.3	10.3	−7.7	−0.3	0.525	0.221	0.31	95
American robin	<i>Turdus migratorius</i>	1.7	6.0	6.0	0.0	−0.7	−3.3	0.354	0.401	0.10	208
Phainopepla	<i>Phainopepla nitens</i>	1.7	0.3	1.0	3.7	−1.3	1.0	0.013	0.909		
European starling	<i>Sturnus vulgaris</i>	16.0	24.3	6.7	19.0	12.3	8.3	0.744	0.104	0.14	153
Hutton's vireo ^e	<i>Vireo huttoni</i>	9.0	8.0	12.0	3.0	10.0	2.7	0.014	0.747		
Spotted towhee	<i>Pipilo maculatus</i>	31.0	30.0	14.3	4.3	7.0	8.0	0.364	0.084	0.59	74
California towhee	<i>Pipilo crissalis</i>	12.0	7.7	1.3	11.0	6.3	23.0	0.001	0.393		
Lark sparrow	<i>Chondestes grammacus</i>	18.0	25.7	20.7	16.0	5.7	6.7	0.952	0.466	0.16	144
Rufous-crowned sparrow	<i>Aimophila ruficeps</i>	17.3	26.3	35.7	30.3	8.3	13.0	0.862	0.278	0.21	119
Western meadowlark	<i>Sturnella neglecta</i>	3.3	6.0	5.3	−0.3	4.7	3.3	0.285	0.185	0.09	229
Red-winged blackbird	<i>Agelaius phoeniceus</i>	0.3	1.3	2.3	1.3	0.3	−1.3	0.706	0.705	0.04	526
Brewer's blackbird ^e	<i>Euphagus cyanocephalus</i>	2.3	12.3	6.3	−4.7	−2.0	−12.0	0.953	0.912	0.07	263
Lesser goldfinch	<i>Carduelis psaltria</i>	100.7	82.3	32.7	39.7	3.7	32.0	0.343	0.105	0.66	59
Purple finch	<i>Carpodacus purpureus</i>	7.3	9.3	−4.0	−6.7	−5.3	−5.3	0.814	0.163	0.46	75
House finch	<i>Carpodacus mexicanus</i>	34.7	24.0	−2.7	14.0	5.3	10.3	0.184	0.195	0.37	85
Breeding migrants											
Western kingbird ^d	<i>Tyrannus verticalis</i>	12.7	14.3	−4.0	7.3	−1.0	9.0	0.029	0.696		
Ash-throated flycatcher	<i>Myiarchus cinerascens</i>	54.3	56.0	17.3	22.7	−4.0	9.0	0.106	0.263	0.91	
Pacific-slope flycatcher	<i>Empidonax difficilis</i>	1.7	2.3	3.0	−1.7	1.7	−0.7	0.038	0.131		
Tree swallow	<i>Tachycineta bicolor</i>	7.3	6.3	0.7	5.3	2.0	5.3	0.419	0.755	0.09	229

Table 2. Continued.

Common name	Scientific name	Preharvest mean count		Postharvest mean population trend ^a				P-values for effect ^b			Minimum effect % ^d
								Treatment	Treatment × year	Power ^c	
		1993	1994	1995	1996	1997					
Violet-green swallow	<i>Tachycineta thalassina</i>	22.3	19.3	11.3	10.3	29.3	27.3	0.627	0.923	0.18	131
House wren	<i>Troglodytes aedon</i>	26.0	18.3	-4.0	6.0	6.7	33.3	0.029	0.028		
Blue-gray gnatcatcher	<i>Poliophtila caerulea</i>	17.0	12.3	2.7	4.0	0.3	8.7	0.436	0.233	0.19	129
Solitary vireo ^g	<i>Vireo solitarius</i>	1.7	0.7	-0.3	0.3	1.7	2.0	0.859	0.875	0.08	371
Orange-crowned warbler	<i>Vermivora celata</i>	26.3	24.3	0.3	-1.0	8.0	0.7	0.361	0.226	0.43	78
Black-throated gray warblers	<i>Dendroica nigrescens</i>	0.3	1.7	0.3	2.7	0.7	-0.7	0.741	0.067	0.05	384
Yellow-breasted chat ^e	<i>Icteria virens</i>	2.7	1.0	2.3	-0.3	2.3	1.0	0.309	0.251	0.05	491
Black-headed grosbeak	<i>Pheucticus melanocephalus</i>	5.3	3.3	2.7	1.7	5.3	4.0	0.406	0.916	0.09	214
Lazuli bunting	<i>Passerina amoena</i>	22.7	23.3	-3.0	-0.3	2.0	-1.7	0.971	0.208	0.54	68
Chipping sparrow ^e	<i>Spizella passerina</i>	35.0	44.7	-1.3	-3.7	-3.7	-6.7	0.732	0.850	0.46	76
Brown-headed cowbird	<i>Molothrus ater</i>	17.3	18.3	6.7	3.7	11.0	11.7	0.859	0.528	0.21	119
Bullock's oriole	<i>Icterus bullockii</i>	18.7	12.7	-8.0	2.0	-3.7	7.3	0.030	0.837		
Migrants											
Western wood-pewee ^e	<i>Contopus sordidulus</i>	2.0	2.3	2.7	-0.7	0.0	1.3	0.580	0.028		
Dusky and Hammond's flycatchers	<i>Empidonax oberholseri</i> and <i>E. hammondi</i>	1.7	1.0	-0.3	0.0	7.0	4.7	0.246	0.286	0.05	417
Warbling vireo	<i>Vireo gilvus</i>	0.7	0.0	1.0	0.3	3.0	2.7	0.140	0.827	0.04	1,072
Nashville warbler	<i>Vermivora ruficapilla</i>	1.0	0.3	-0.7	1.0	1.3	0.7	0.793	0.163	0.04	515
Yellow-rumped warbler	<i>Dendroica coronata</i>	34.3	26.0	0.3	-4.3	-1.0	7.3	0.466	0.308	0.25	109
Townsend's warbler	<i>Dendroica townsendi</i>	1.0	2.3	0.7	0.7	-1.0	-0.3	0.095	0.754	0.08	246
Wilson's warbler	<i>Wilsonia pusilla</i>	6.0	7.3	0.3	2.0	-1.3	-1.3	0.276	0.582	0.28	101
Western tanager	<i>Piranga ludoviciana</i>	2.3	1.3	1.0	0.3	0.0	0.3	0.221	0.570	0.11	190
Winter residents											
Steller's jay	<i>Cyanocitta stelleri</i>	9.3	19.7	-9.3	-19.7	-9.0	-19.7	0.334	0.321	>0.999	
Ruby-crowned kinglet	<i>Regulus calendula</i>	4.7	6.0	0.7	2.0	11.0	6.7	0.949	0.111	0.15	151
Hermit thrush ^h	<i>Catharus guttatus</i>	0.0	0.0	0.7	0.0	2.0	1.0	0.259	0.732		
Dark-eyed junco	<i>Junco hyemalis</i>	1.3	1.7	-1.3	-1.7	2.7	5.0	0.389	0.405	0.05	428
White-crowned sparrow	<i>Zonotrichia leucophrys</i>	0.3	0.3	-0.3	2.0	0.3	3.3	0.002	0.697		
Golden-crowned sparrow ^e	<i>Zonotrichia atricapilla</i>	2.3	1.0	5.0	23.0	3.3	23.0	<0.001	0.806		

^a Postharvest mean population trend is computed as the mean of the differences in count between the postharvest year indicated in the column heading and the preharvest baseline year (1993).

^b P-values are for effects in a HM-ANCOVA computed independently for each species, with no adjustments made to control the experimentwise Type I error rate.

^c Power = $1 - \beta$, where β is the probability of failing to reject the null hypothesis of no treatment effect when it is false. Specified power is for a main effect of treatment on postharvest change in count equal to 50% of the mean preharvest count.

^d Minimum treatment effect size (expressed as a percentage of mean preharvest count) detectable with power = 0.8.

^e Log transformed for HM-ANCOVA, $y = \ln(x + c)$, where $c > |\text{minimum } x|$.

^f Square-root transformed for HM-ANCOVA, $y = (x + c)^{1/2}$, where $c > |\text{minimum } x|$.

^h Suspected but unconfirmed breeder.

Power not calculated because mean preharvest count = 0.

cies in which the mean population trend on the control plots was zero, this effect would represent a 50% change in mean count on harvested plots. If power was <0.8 , we also calculated the size of the effect that could be detected with power = 0.8. All power calculations were made with the computer program STPLAN (University of Texas System Cancer Center, Houston, Texas, USA). Because this program does not specifically provide for a covariate, actual error degrees of freedom were 1 less than used in the power calculations; thus, power estimates were slightly inflated.

RESULTS

Vegetation Changes

From the 30 harvested plots, we removed 3,405 individual stems totaling 164.2 m² of basal area, and we removed 331 cords of firewood (Table 1). The mean reduction in basal area per plot was 23% (minimum 16%, maximum 36%). Because we harvested primarily blue oaks and interior live oaks, the mean reduction in basal area for these species was higher (29% for blue oak, 34% for interior live oak) than the overall mean. The mean reduction in number of stems for blue oaks (26%) and live oaks (24%) was slightly lower than the reduction in basal area because we harvested few trees <15 cm root-collar diameter.

We did not measure changes in the understory, but shrubs in most plots were sparse enough that harvesting operations did not affect the existing shrub layer. Mean preharvesting shrub cover on the harvested plots was 6.4% (SD = 4.9), and 13 plots had an initial shrub cover of $<5\%$ (W. M. Block, unpublished data). Creation of brush piles clearly increased the effective shrub cover on the harvested plots and in some cases created a shrub layer where none had existed. Brush piles were not constructed to meet any particular criteria, but most were 3–4 m in diameter and 1.0–1.5 m in height.

The herbaceous layer on harvested plots was disturbed to varying degrees depending on the month that harvesting occurred and the equipment used. Disturbance was generally light and was most evident on plots harvested in winter and early spring where a bulldozer was used to remove wood. In the first breeding season post-harvest, these plots had exposed soil where the bulldozer had removed newly sprouted grasses

and forbs. By the second breeding season post-harvest, these tracks were no longer evident.

Bird Responses

We identified 99 species of birds within the study plots: 16 were incidentals, 15 had <10 total detections, and 6 were mostly detected flying over plots (Aigner 1996). Of the remaining 62 species included in our analysis, 31 were year-round residents, 16 were breeding migrants, 6 were winter residents, and 9 were migrants en route to breeding grounds (Table 2).

The effects of interest in the RM-ANCOVA model were treatment and the treatment $\times$ year interaction. The treatment effect was a direct measure of the difference in mean population trend between harvested and control plots. A statistically significant treatment $\times$ year interaction was evidence that the difference in population trend on harvested and control plots was not constant in the 2 years postharvesting. The effect of year on population trend, although a necessary component of the model, was not biologically meaningful, because it simply indicated the difference in the number of birds counted between 1993 and 1994 was not the same as the difference between 1993 and 1995. The year effect was also confounded with interobserver differences, which made interpretation even more difficult. Because each observer balanced counting effort between harvested and control plots (with 100% success in 1993, 99.7% success in 1994, and 99.3% success in 1995), interobserver differences were not confounded with the main effect of treatment.

Without controlling the experimentwise Type I error rate, we found evidence ($P \leq 0.05$ for effects in each RM-ANCOVA considered separately) that population trends of 14 species were affected by harvesting (Table 2). Eleven of these species had constant population responses in the 2 years postharvesting (no treatment $\times$ year interaction), whereas 3 showed evidence for the effect of an interaction between treatment $\times$ year. The California quail, northern pygmy-owl, western kingbird, Bewick's wren, phainopepla, California towhee, white-crowned sparrow, golden-crowned sparrow, and Bullock's oriole all showed consistently increasing population trends on the harvested plots relative to the controls. Evidence for a treatment $\times$ year interaction for the house wren indicated that a population increase on harvested plots increased from the first to the second season

postharvesting. Three of the species with large percent increases, the phainopepla, northern pygmy-owl, and white-crowned sparrow, were still relatively rare throughout the study, with 31, 16, and 22 total detections in all 3 years combined, respectively. By contrast, we had 193 total detections of the golden-crowned sparrow, which was the next rarest bird in this group.

Two species, the Pacific-slope flycatcher and Hutton's vireo, showed consistently negative population trends on harvested plots compared to control plots. Averaged over the 2 years postharvesting, flycatcher detections declined by 50% on the harvested plots but increased by 140% on the control plots. Postharvest detections of Hutton's vireo increased an average of 122% on control plots but increased only 35% on harvested plots. The Pacific-slope flycatcher was also consistently rare, with only 43 total detections over all 3 years, compared to 236 detections for the Hutton's vireo.

The bushtit and western wood-pewee had varying responses in the 2 years postharvesting. Bushtit detections increased 260% on the harvested plots in 1994 and then returned to their approximate preharvesting levels in 1995. On control plots, bushtit detections increased only 56% in 1994, but detections remained elevated 38% above their preharvest level in 1995. Detections of wood-pewees, relative to population trends on control plots, decreased on harvested plots in 1994 and then increased in 1995. In 1994, wood-pewee detections decreased 29% from preharvesting levels on harvested plots but increased 133% on control plots. In 1995, pewee detections increased 57% from preharvesting levels on harvested plots but returned to preharvesting levels on control plots.

When we controlled the experimentwise Type I error rate at 0.2 with a sequential Bonferroni adjustment, population shifts for bushtits, California towhees, white-crowned sparrows, and golden-crowned sparrows were the only responses that remained statistically significant. Although we recognize that evidence for the responses of some species may be spurious, we think this likelihood should be evaluated on a species-by-species basis with reference to existing knowledge about the habitat requirements of each species.

Among species for which we found no evidence of treatment effects, statistical power to detect a 50% change in count (assuming no change on control plots) varied (Table 2). We

had power >0.8 for only 5 species: mourning dove, acorn woodpecker, ash-throated flycatcher, plain titmouse, and Steller's jay. The power estimate for the Steller's jay is artificially inflated because many birds were detected in 1993, but almost none were detected in 1994 or 1995. Consequently, the population trends had almost zero variance. For all but 12 additional species, the size of the effect we could detect with power = 0.8 was $\geq 100\%$ of the mean preharvest count. For these species, complete extirpation from the harvested plots was the only negative response we could detect with power = 0.8. In general, power was greatest for ubiquitous and abundant year-round residents and breeding migrants and was consistently low for non-breeding migrants.

DISCUSSION

The experimental firewood harvest affected vegetation primarily by reducing basal area and canopy cover and increasing the proportion of pines and trees in very large (old growth) or very small (noncommercial) size classes. The existing understory was little affected, except that brush piles augmented the existing shrub layer or perhaps provided a functional shrub layer where none had existed previously.

Bird species affected by the vegetation change included residents, breeding migrants, winter residents, and possibly 1 migrant. Few overall patterns emerged. Population responses were species-specific and were only sometimes consistent with existing knowledge about the ecology and habitat requirements of a species.

The most prominent pattern was 6 of the 10 species that responded positively to harvesting were regularly observed using brush piles. When detected on harvested plots, California quail, golden-crowned sparrows, and white-crowned sparrows were almost always in or near brush piles. House wrens, Bewick's wrens, and California towhees were also regularly seen in brush piles. All of these species, with the exception of the house wren, were among the 6 species most frequently observed using brush piles by Gorenzel *et al.* (1995) in a portion of our study area. They also found spotted towhees frequently used brush piles. Although we occasionally observed spotted towhees in brush piles, we found no evidence that spotted towhees increased on harvested plots, despite relatively good power to detect an effect ($1 - \beta = 0.8$, for a 74% change in mean count). These

discrepancies in brush pile use for the house wren and spotted towhee may be due to seasonal differences between the 2 studies. Gorenzel et al. (1995) made most of their observations from late fall to early spring, whereas ours were made in the spring and early summer. Spotted towhees were present at the study site year-round but may have used different habitats at different times of the year. House wrens were only present at the study site from late March to early July.

Among other species for which we found evidence of harvesting effects, responses were less clearly linked to such a distinct aspect of vegetation change. Population responses for 4 species were consistent with existing knowledge of their habitat requirements, but exact mechanisms for the shifts were unclear. The western kingbird and Bullock's oriole, both of which showed consistent population increases on harvested plots, are associated with more open stands of oaks (Block 1989). The Hutton's vireo and Pacific-slope flycatcher, both of which showed consistent population decreases on harvested plots, are associated with dense canopy cover (Johnson 1980; W. M. Block, unpublished data). The flycatcher reaches its highest density in moist, coastal coniferous forests (Johnson 1980), and even undisturbed woodlands at our study site appeared to provide only marginal habitat for this species (it was detected on only 19 of the study plots).

For 2 species, the northern pygmy-owl and the bushtit, population responses were not clearly consistent with existing knowledge of habitat requirements. Somewhat contrary to our results, Wilson et al. (1991) found that pygmy-owls were associated with high tree density in a coast-range oak woodland. Similarly, in an earlier study at our site, Block (unpublished data) found that increased canopy closure by trees, particularly live oaks, best discriminated habitat used by bushtits from available habitat. Owls may have increased on harvested plots because the small openings created in denser woodland may have provided increased foraging opportunities (Bent 1938). We cannot think of an obvious reason for the increase in bushtits on harvested plots in the first year after harvesting, but we predict that bushtits should eventually experience a net population decline on harvested plots. Regardless of eventual outcome, the variable population response of bushtits in the 2 years after harvesting emphasizes that harvest-

ing does not always result in stable shifts in population trend, at least in the short term.

Population responses of the phainopepla and western wood-pewee were difficult to interpret. Habitat requirements for the phainopepla in oak woodlands are poorly known; thus, their apparent increase on harvested plots, if not spurious, remains unexplained. The varying response of the western wood-pewee was probably spurious, because, like other uncommon nonbreeding migrants, detections were sporadic in space and time.

Although our results indicate few negative effects of a 23% reduction of basal area on the bird community, our study had several limitations. First, study plots were undoubtedly smaller than individual home ranges for some species and were probably small enough that plots could continue to be used by individuals relying on suitable habitat in surrounding woodland. Noon et al. (Silvicultural options in managed oak woodlands to benefit breeding birds, unpublished report. California Department of Forestry and Fire Protection, Sacramento, California, USA) suggested that many oak-woodland species respond to variation in their environment at spatial scales much greater than 5 ha. For many species, effects of harvesting are therefore likely to increase with the spatial scale of the harvest.

Second, our experiment had low statistical power to detect even large effects for many less-common species. For 29 species, we had power <0.8 to detect any population decline short of complete extirpation on the harvested plots. Given that some of these species were uncommon because they were habitat specialists, and that habitat specialists tend to be more sensitive to habitat change than generalists (Tellería and Santos 1995), we may have had low power to detect responses in some of the species most likely to be negatively affected by thinning.

Third, responses of some species may be delayed by site fidelity (Wiens et al. 1986, Knopf and Sedgwick 1987). We monitored responses for 2 years postharvest and anticipated a possible year lag in the response of species. Interestingly, we found little evidence for such lags. In the RM-ANCOVA models, response lags would have been indicated by a statistically significant interaction between treatment and year, with a population shift increasing from the first to the second season postharvesting. Only pop-

ulation shifts for the house wren showed evidence for such a pattern. The northern pygmy-owl did show a larger population increase on harvested plots in 1995 relative to 1994, but the interaction was not statistically significant. For most species in which we observed a response to harvesting, the difference in population trend between harvested and control plots was about equal in the 2 years postharvest.

Although we observed only 1 definite response lag, lags for some species could conceivably be as long as the lifetimes of the individual birds affected (Wiens et al. 1986). A further complication of response lags is a reduction of statistical power in the RM-ANCOVA. Our power calculations were based on the assumption of an average response of 50%. If no response occurred in the first year postharvesting, then the response in the second year postharvesting would have to be 100% for the power estimates to hold.

A final caveat is the likelihood that population responses of some species will change or emerge as brush piles decay and vegetation recovers. Brush piles are a temporary habitat feature, and their use may decline as they age and compress (Gorenzel et al. 1995). Also, vegetation change on the harvested plots is likely to be accelerated compared to natural succession on the control plots. For example, many stumps, particularly of live oaks, resprouted shortly after harvesting. Such resprouting leads to a multistemmed, shrub-like growth form that may further augment the effective shrub layer on harvested plots.

MANAGEMENT IMPLICATIONS

Our results indicated that small-scale firewood harvests that reduce basal area by <25% while preserving old-growth trees, trees with nest cavities, and granary trees have little negative short-term (<2 yr) effects on population trends of most of the more common bird species present during the breeding season. In addition, a small group of species that use brush piles for foraging, cover, and possibly nesting may be benefitted as long as brush piles remain usable. However, this conclusion comes with caveats. First, we emphasize that we implemented what we view as a conservative harvest. Harvesting was light in both the amount of basal area removed (23%) and in the area of the individual harvests (3.1 ha). Second, we emphasize that harvests should preserve old-growth

trees, trees with nest cavities, and acorn woodpecker granary trees to reduce negative effects on cavity-nesting species. Third, the limited temporal and spatial scale of this experiment, as well as the low statistical power to detect responses of many less common species, requires the conclusion that some effects of harvesting remain unknown.

Conclusions about the effects of firewood harvesting on wildlife in oak woodlands are dependent on the spatial and temporal scale at which we record our measurements (Block and Morrison 1991). In this study, we sacrificed a large spatial scale for the strong inference of a designed experiment. Future studies should examine effects at a larger scale (e.g., watershed). Furthermore, long-term studies are necessary to evaluate the importance of site fidelity in obscuring effects in the short term and to describe the persistence of the effects found here.

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