

LESSONS LEARNED FROM PRESCRIBED FIRE IN PONDEROSA PINE FORESTS OF THE SOUTHERN SIERRA NEVADA

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Abstract. Prescribed fire is a commonly used management tool in fire-suppressed ponderosa pine (*Pinus ponderosa*) forests, but effects of these fires on birds are largely unstudied. We investigated both direct and indirect impacts on breeding birds in ponderosa pine forests of the southern Sierra Nevada where fires were applied in the spring. Following prescribed fire, we found that the largest losses of snags, including the large ponderosa pine snags preferred for nesting, were during the first application of fire after a long fire free interval, while net losses after the second fire application were similar to those in unburned areas. Hardwood trees, primarily oaks, co-occur with ponderosa pines in these forests and were preferred as nesting substrates. Burning may increase recruitment of shade intolerant oaks (*Quercus* spp.) and pines, which is low in the current closed canopy conditions. Burning may also lead to post-fire changes in habitat use, but we found little apparent response in territory placement to prescribed burning by a resident species. During spring fire applications, direct mortality of eggs and nestlings is of concern. We documented low nest mortality and observed continued breeding activities during burning even when up to 85% of the area surrounding the nest was burned. Based on 1600 nests observed over nine years, May and June were the months when most nesting activity occurred. Overall, we found few negative impacts and note that essential habitat components, such as oaks and large ponderosa pines, may depend on reintroducing fire. Importantly, managers can reduce negative impacts by protecting preferred nesting snags and adjusting timing in relation to breeding activities.

Key Words: birds, Hutton's Vireo, prescribed fire, Sierra Nevada, snags.

LECCIONES APRENDIDAS DE QUEMA PLANEADA EN BOSQUES DE PINO PONDEROSA DEL SUR DE SIERRA NEVADA

Resumen. Las quemas planeadas son comúnmente utilizados como herramienta de manejo en bosques de pino ponderosa (*Pinus ponderosa*) donde fuegos han sido suprimidos, pero los efectos de estos incendios no se han investigado en las aves en gran parte. Investigamos tanto los impactos directos e indirectos en las aves nidando en bosques de pino ponderosa del sur de Sierra Nevada donde los incendios se han aplicado en la primavera. Siguiendo quemas planeadas, encontramos que la mayor pérdida de tocones, incluido los tocones de pino ponderosa grandes que fueron preferidos para anidación, ocurrieron durante la primera solicitud de fuego después de un largo intervalo libre de fuego, mientras que las pérdidas netas después de la segunda solicitud de fuego fueron similares a las de zonas sin quemar. Árboles de madera dura, principalmente encinos (*Quercus* spp.), ocurren con pinos ponderosa en estos bosques y fueron preferidos como sustratos de anidación. Incendios pueden aumentar el reclutamiento de sombra intolerante encinos y pinos, que es baja en el actual las condiciones de dosel cerrado. Los incendios también pueden dar lugar a post-incendio cambios en el uso del hábitat, pero encontramos poca respuesta aparente a quemas planeadas en la colocación de territorios de una especie residente. Durante la primavera aplicaciones de fuego pueden causar la mortalidad de huevos y polluelos y es motivo de preocupación. Documentamos baja mortalidad de nidos y observamos que las actividades de reproducción continúan durante el incendio, aun cuando hasta el 85% del área que rodea el nido fue quemado. Sobre la base de 1600 nidos observado durante nueve años, la mayoría de anidando ocurrió en los meses de mayo y junio. En totalidad, encontramos pocos impactos negativos y notamos que los componentes esenciales del hábitat, tales como encinos y pinos ponderosa grandes, pueden depender de la reintroducción de fuego. Es importante destacar que los gerentes pueden reducir los efectos negativos por la protección de tocones preferidos para anidación y ajustar el calendario en relación con las actividades de cría.

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INTRODUCTION

Ponderosa pine (*Pinus ponderosa*) forests are common throughout the western United States, and in the Sierra Nevada occur along the western slope at elevations ranging from 600 to 1500 m in the southern part of the range (Holland 1986). Ponderosa pine forests in the Sierra Nevada support a variety of other conifer and hardwood tree species (Mayer and Laudenslayer 1988). One important difference between these forests and interior ponderosa pine forests is the presence of tree species restricted to the Pacific states, including sugar pine (*Pinus lambertiana*), incense cedar (*Calocedrus decurrens*), and California black oak (*Quercus kelloggii*).

Ponderosa pine forests are fire adapted though natural fire regimes vary throughout the range (Rundel 1977, Miller 2000, Fitzgerald 2005). In the Sierra Nevada there is a long history of fire suppression which, along with extensive grazing and logging, has contributed to an altered forest structure and composition (Skinner and Chang 1996). As a result, stands have become dense, woody debris has accumulated, and shade tolerant species have increased (van Wagtenonk 1985, Minnich et al. 1995). Land managers are using prescribed fire to reduce forest fuels and wildfire severity, but prescribed burning also helps return an important ecological process to the landscape (Husari and McKelvey 1996, Kauffman 2004).

Attributes of fire such as timing, severity, and extent will influence its effects on wildlife (Lyon et al. 2000a). Prescribed fire attributes often differ from those of natural fires, especially within the context of altered fire regimes (McKenzie et al. 2004). In particular, prescribed fire is often applied outside of the historic fire season because of issues related to fuel loads or air quality. These fire application periods include spring and coincide with avian breeding activities. Prescribed fire may then result in direct mortality of nests, eggs and nestlings, in addition to indirect effects related to habitat alteration, but little research has occurred on these topics (Hejl 1994, Lyon et al. 2000b, Kotliar et al. 2002). Although adult birds are not threatened by prescribed fire, nests are vulnerable (Lyon et al. 2000b). Direct mortality of bird nests from prescribed fire has been recorded, primarily in grassland habitats (Grange 1948, Leedy 1950, Moyle 1964, Erwin and Stasiak 1979, Kruse and Piehl 1986).

Habitat components that provide nesting and foraging habitat for birds are not equally vulnerable to burning. Conifer species such as ponderosa pine have adaptations that reduce

fire mortality (Flint 1925), but other species, such as oaks, are easily top-killed during burning (McDonald and Huber 1995). Trees that are killed by fire but not consumed become snags, an important nesting substrate. But existing snags are also highly flammable (Horton and Mannan 1988).

In most passerine species, nesting and/or foraging resources are defended within an established territory (Brown 1969, Best 1977). Alteration of territory resources may lead to use of non-preferred habitats, presumably with lower fitness, or emigration to higher quality habitats (Brown 1969, Stacey and Ligon 1987). Established territories subject to low severity fires, which create a mosaic of burned and unburned patches, may remain occupied if they retain essential habitat elements or, alternatively, individuals may emigrate to unburned locations with potential fitness consequences (Petersen and Best 1987).

Because timing and location can be controlled to a certain extent, management burning can be used to examine the effects of fire within a rigorous study design. In this paper we focus on lessons learned from our research that relate to recommendations for application of prescribed fire. To summarize results of our research on prescribed fire effects, we used data from multiple studies from 1995 to 2003 in the southern Sierra Nevada, including snag dynamics, nest tree selection, territory placement of a resident bird species and direct nest mortality. We followed and measured snags before and after fire, and these changes were examined in relation to snags predicted use for nesting. We also examined the choice of tree species for nesting. Territories of Hutton's Vireo (*Vireo huttoni*) were mapped and related to average fire severity measures over time, assuming that habitat use would shift to avoid burned areas if burning substantially decreased habitat suitability. Finally we present data on direct losses of nests due to burning during the nesting period, and predict potential nest mortality based on the timing of the nesting period.

METHODS

STUDY AREA

The study area was located on the west slope of the southern Sierra Nevada in the High Sierra Ranger District of the Sierra National Forest, Fresno County, California (37°02'N, 199°15'W). In 1995, four 40-ha study plots were established in ponderosa pine-dominated forests (Fig. 1). While grazing, recreational, and vegetation

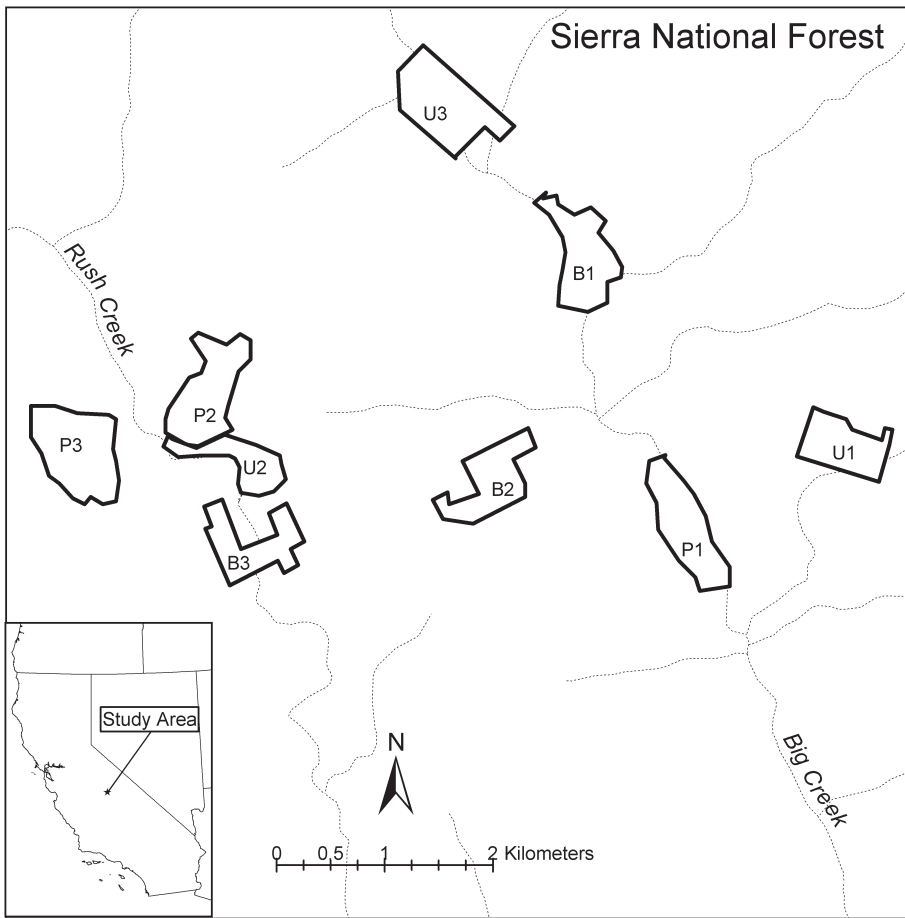


FIGURE 1. Study plot locations and prescribed fire applications. “B” designates plots burned in 2002. “P” designates plots burned in 1997/98 and 2003 except P3 which only had fire applied in 1998. “U” indicates plots that remained unburned. U1, U3, B2, and B3 plots were established in 1995. The remaining plots were established in 2001.

management activities occurred in the area, none of the plots had been recently logged. Forested areas were interspersed with granitic outcrops, creeks, and shrub fields dominated by whiteleaf manzanita (*Arctostaphylos viscida*). Tree species included ponderosa pine, sugar pine, incense cedar, white fir (*Abies concolor*), California black oak, and canyon live-oak (*Quercus chrysolepis*). Mountain misery (*Chamaebatia foliolosa*) was the dominant ground cover species. In 2001, five additional 40-ha plots were established (Fig. 1). Three of these plots had fire applied in 1997 or 1998 as part of a fuel reduction program (Fig. 1, P1-P3). All fires were set using drip torches with ignition occurring throughout the plots and surrounding areas. Previous to these prescribed fire applications, no fires were recorded on any plots after 1947.

SNAGS

All nine plots were used to track snag populations from 2001 to 2003. Fire was applied on three plots in 2002 (Fig. 1, B1-B3) and on two plots in 2003 that were burned previously in 1997 or 1998 (Fig. 1, P2-P3). A third plot, P1, also burned previously, was scheduled for burning in 2003 but was not burned due to weather constraints. This plot was grouped with unburned plots when considering snag changes over time. No fire was applied to the remaining three plots (U1-U3).

Belt transects (50 m wide) for sampling snags were established on each plot (see details in Bagne et al. 2008). Each transect was surveyed for snags greater than 20 cm diameter at breast height (dbh), and thus potentially

useable as a nesting substrate, until a minimum of 35 snags were recorded. Snags were tagged and measured before and after burning. Data recorded included height, dbh, tree species, and decay (Cline et al. 1980). Snags on unburned plots were measured during the same period. In addition, we examined snag fate in relation to snags preferred as nesting substrates. Probability of use of snags was related to diameter and tree species based on a discriminant function from the same study area (Bagne et al. 2008).

OAKS

Nests were located and monitored during nest searching efforts from 1995–2003 on four plots and 2001–2003 on five plots. We recorded substrate type (e.g., tree, snag, shrub, etc.) and species for each nest. In addition, 35 random sites were located on each of the four plots established in 1995 to assess availability of various habitat components. All trees and snags within a 0.04 radius circle were measured and recorded. Nesting substrates used were compared with available species using a Chi-square test with a continuity correction for a 2×2 table. When nests in the same location were reused, we used data from only one nest.

HUTTON'S VIREO TERRITORIES

To examine territory use we chose a resident species, because territories would be maintained throughout the time period when fire could be applied and allow us to locate pairs early in the year. Locations and movements of individual Hutton's Vireos were recorded during 2002 and 2003 on two plots, B2 and B3 (Fig. 1). Territory mapping began February 26, prior to burning in April 2002, and continued throughout the nesting season with a total of 15 days spent on B2 and 12 days on B3. Most pairs were in the process of nest building or had an active nest when fire was applied in April 2002. Mapping continued in 2003, one year post-burn, with a total of 8 days on B2 and 7 days on B3. Knowledge of territory locations from 2002 aided location of pairs in the subsequent year. While mapping effort varied, it exceeded the five days recommended to confirm territories (Bibby et al. 1992). Few individuals were banded, thus territories were based on uncontested presence in repeat locations and boundary conflicts. We used passive observations as well as playbacks of males singing to encourage movement and response (Falls 1981). Nests were also located for most pairs and aided territory mapping.

Fire severity was categorized on the two 40-ha plots where territory mapping was done, B2 and B3 (Fig. 1). Fire severity was classified into four categories based on the amount of area burned (0%, <50%, ≥50%, and 100%). Where burn extent was 100%, there were also indicators of high heat such as white ash and consumed logs. Each 40-ha plot was gridded at 50 m intervals. Each 50m square was visually divided into four 25×25 m quadrants and each quadrant was scored as one of the four fire severity categories.

Mapped locations and movements were transferred to ArcGIS (ESRI 2006) and minimum convex polygons were used to define territories (Beyer 2004). Territories were then overlaid on the fire severity grids to assess the severity of burning within each territory for 2002 and 2003. Fire severity scores that fell within the territory polygons were averaged for each year on the two plots. Only territories that substantially overlapped (≥50%) with scored area were used.

NEST MORTALITY

Nests were located and monitored during planned ignition periods. Nest searches focused on those species most likely to be affected by burning (i.e., ground, shrub, and snag nests). Fire-related nest mortality was evaluated on two plots where fire was applied between 4 April and 11 April 2002, and two plots where fire was applied between 10 June and 20 June 2003. The third plot where fire was applied in April 2002 (B1) was excluded because mechanical treatment had occurred on a portion of the plot previous to burning thus altering available nesting habitat. Nests were visited immediately before and after burning on the same day as ignition.

Because fire was applied outside of the peak nesting season, few monitored nests were active. To assess risk by nest timing, we compiled nest date information for all nests found in ponderosa pine forests. The date the first egg was laid, either observed or estimated, was used along with typical nesting period for each species to record all possible days when nests were active. We did not include nest fate information and assumed all nests survived the entire nesting period to better identify the activity period. Activity is thus somewhat inflated because it includes re-nesting attempts following failure.

At open-cup nests active during burning we estimated the percentage burned of the nest site area (11.3-m radius circle around the nest) and of the substrate supporting or, for ground nests, directly covering the nest.

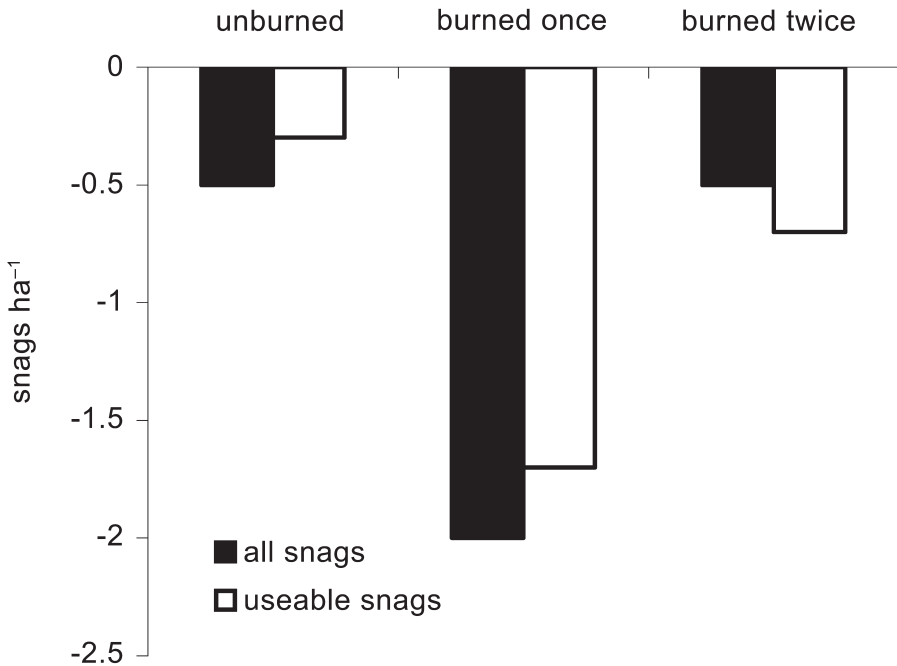


FIGURE 2. Net changes, including fallen and new snags, in snags ha⁻¹ following a first application of fire in 2002 on three plots (burned once), and a second application of fire in 2003 on two plots previously burned in 1997/98 (burned twice). Snag changes on unburned plots were measured during the same time period. No fire was applied to one plot burned in 1998 and was included with unburned plots. Useable snags are based on classification by the discriminant function based on snag selection by cavity nesting species (Bagne et al. 2008).

RESULTS

SNAGS

New snags replaced fallen or burned snags to some extent. The highest net losses of snags were during the first application of fire (2.0 snags ha⁻¹ or 12%, Fig. 2). Snags on plots burned twice, in 1997/98 and then 2003, fell at similar rates to those on unburned plots (0.5 snags ha⁻¹ or 3%, Fig. 2).

Snags preferred by birds were proportionally more likely to fall or burn than other snag types, but the pattern as related to burn history was similar. Net loss of these preferred or useable snags on plots following the first application of fire was 1.7 snags ha⁻¹ (34%). Net losses of preferred snags on plots burned twice were 0.7 snags ha⁻¹ (15%) while net loss of snags on unburned plots was 0.3 snags ha⁻¹ or 2% (Fig. 2).

OAKS

We located 628 nests of 34 avian species in trees from 1995 to 2003. Twenty-two species (56%) nested in hardwoods in ponderosa pine

forests. Although hardwoods and conifers were used almost equally, hardwoods were strongly selected for compared to available trees. Hardwoods comprised only 17% of the available trees, but 50% of all nests in trees were in hardwoods ($X^2 = 79.8$, $P < 0.001$; Fig. 3). Oak species comprised 92% of the hardwood trees used for nesting. The remaining hardwood species were white alder (*Alnus rhombifolia*), Oregon ash (*Fraxinus latifolia*), and Pacific dogwood (*Cornus nuttallii*), all of which were rare.

Hardwood and oak snags were not important as nesting substrates. Of 238 nests found in snags, only 17 were located in hardwood snags (7%) and only four were in oak snags.

HUTTON'S VIREO TERRITORIES

Twenty individual territories were mapped on the two plots in 2002 and 19 were mapped again in 2003. Hutton's Vireos were actively defending territories and involved in nesting activities when fire was applied in April 2002. No territories were deserted immediately following burning. Nine of 39 territories mapped in the two years had <50% overlap with the

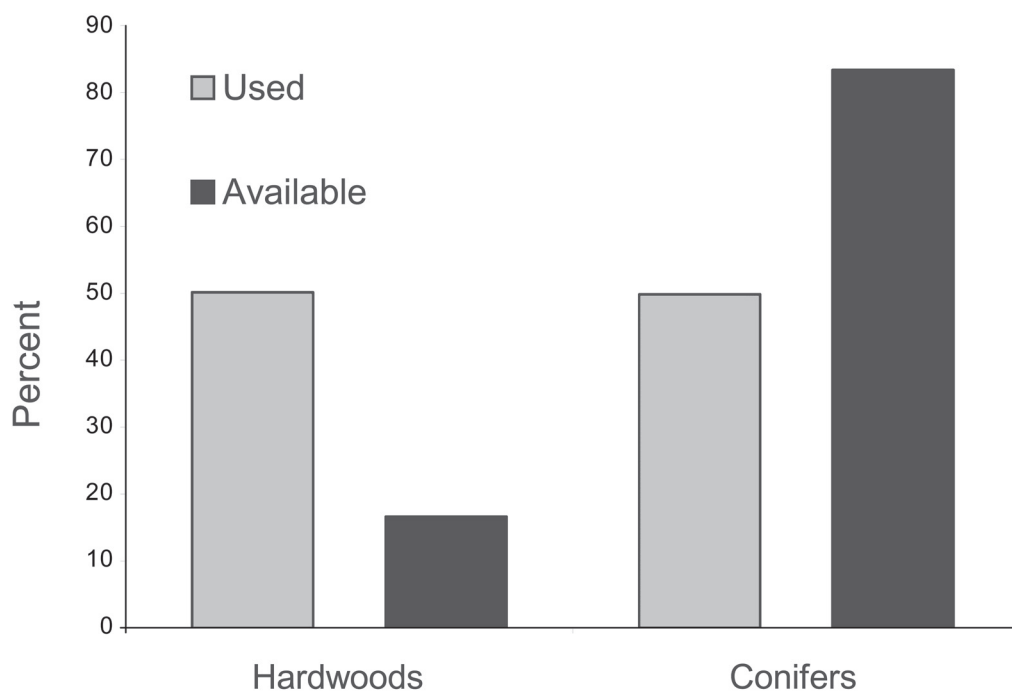


FIGURE 3. Percent conifer and hardwood tree species used as nesting substrates as compared to available trees at random plots.

fire severity grid and were excluded from the analysis. Territory locations and average fire severities were similar between 2002 and 2003, suggesting that overall Hutton's Vireos did not shift territory locations to avoid burned areas (Fig. 4). Mean fire severities within vireo territories were similar in 2002 and 2003 (B2: $\bar{x} = 1.10$, $SD = 0.92$ for 2002, $\bar{x} = 1.06$, $SD = 0.88$ for 2003; B3: $\bar{x} = 1.01$, $SD = 0.76$ for 2002, $\bar{x} = 1.04$, $SD = 0.75$ for 2003). A fire severity of 1 indicates the 25-m square was less than 50% burned.

NEST MORTALITY

Twenty-three nests of 10 species (Red-breasted Sapsucker, White-headed Woodpecker, Northern Flicker, Cassin's Vireo, Hutton's Vireo, Red-breasted Nuthatch, Brown Creeper, Spotted Towhee, Dark-eyed Junco, and Black-headed Grosbeak; see Appendix A for scientific names) were active during burning on four plots over two years. Four nests were destroyed during burning, including one cavity nest (Brown Creeper) and three open-cup nests (1 Spotted Towhee and 2 Dark-eyed Juncos) (Table 1). For open-cup nests, the nesting substrate was 100% burned for destroyed nests and ranged from 0 to 95%

for surviving nests. The surrounding nest site at all active open-cup nests was burned to some extent. Fifty to 90% of the nest site was burned for destroyed nests. For nests that survived fire, the amount of the nest site burned ranged from 15 to 85%.

From 1995 to 2003 we collected information on 1600 nests of 49 species (Appendix A). Extrapolating to the full nesting period, we created a dataset of 45 443 active nest days. Across all years, nests were active from 24 March to 20 August and concentrated around the peak at 29 May (Fig. 5). May and June were the primary nesting months (Table 2). We found little evidence that particular species or groups of species nested either predominantly early or late, compared to the breeding bird community as a whole. The earliest 5% of active nest days (before 28 April) were from 24 species and the latest 5% of active nest days (after 2 July) were from 30 species. Fifteen species had extended nesting seasons with active nests in both the first and last 5% of active periods. The majority of early nesters were resident or short distance migrants, but two species of long distance migrants, Pacific-slope Flycatcher and Black-headed Grosbeak, also nested during the period prior to 28 April.

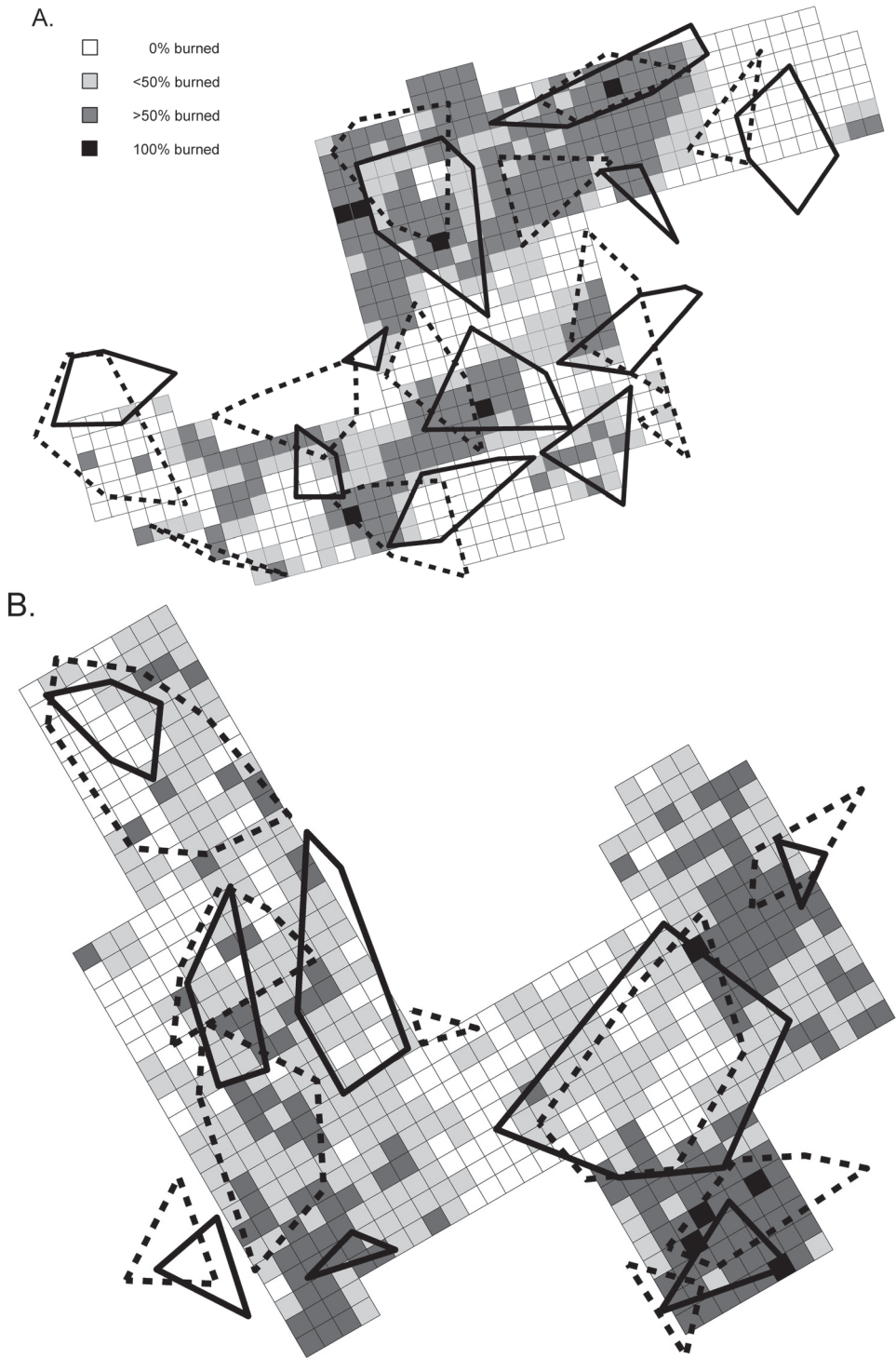


FIGURE 4. Burn severity and Hutton's Vireo territories for plots B2 (A) and B3 (B). Each square is 25 m × 25 m. Dashed polygons are for 2002 and solid polygons are for 2003. Polygons that were not primarily ($\geq 50\%$) within gridded area were not used to calculate average territory burn severities.

TABLE 1. FATE AND BURNING EXTENT FOR NESTS ACTIVE DURING PRESCRIBED FIRE TREATMENTS IN APRIL 2002 AND JUNE 2003. EXTENT OF BURNING WAS RECORDED AT OPEN-CUP NESTS (2 SURVIVING NESTS MISSING DATA). SUBSTRATE WAS THE PLANT SUPPORTING OR COVERING THE NEST. NEST SITE WAS A 0.04-HA CIRCLE CENTERED ON THE NEST.

Nest fate during fire	Cavity nests		Open-cup nests	
	<i>n</i>	<i>n</i>	% substrate burned (SD)	% nest site burned (SD)
Destroyed	1	3	100 (0)	67 (21)
Survived	7	12	11 (31)	38 (23)

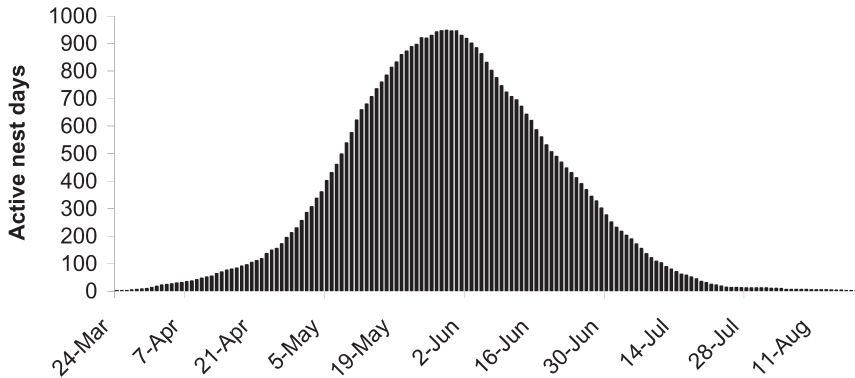


FIGURE 5. Nest activity by day compiled for all nests 1995 to 2003. Nests were considered to be active for the entire nest period (i.e., all successful).

DISCUSSION

Some negative impacts were found from prescribed fire in the spring, but the extent of the effects was surprisingly limited considering the high potential for impacts on breeding activities.

We found that hardwoods, primarily oaks, were a preferred nesting substrate and, in earlier work, we found that large diameter and ponderosa pine snags were preferred for nesting by snag nesting birds (Bagne et al. 2008). Regeneration and sustainability of these two important nesting substrates are dependent on fire. Fire prepares the seed bed and allows sufficient sunlight for regeneration of ponderosa pine (van Wagendonk 1985, Fitzgerald 2005). Once they become snags, however, we found that large ponderosa pine snags were vulnerable to burning, particularly during the first fire application after a long fire-free interval. These snags

represent only a fraction of the snags on the landscape thus protection of these snags during burning should be feasible. Raking flammable material from the base of preferred snag types is recommended (Horton and Mannan 1988).

Hardwood ecosystems in the Sierra Nevada have received little attention from land managers (McDonald and Huber 1995). The importance of oaks for nesting expands their value for wildlife beyond the established link with acorn production (Grinnell 1936, Goodrum et al. 1971, Rodewald 2003). Oaks are declining in many areas because recruitment is insufficient to offset mortality of older trees (McDonald and Tappeiner 1996, Garrison et al. 2002). Disturbances such as fire are needed because oak seedlings and saplings are outcompeted by the faster growing conifers (McDonald and Tappeiner 1996). Prescribed fire can be used to promote seedling establishment, eliminate leaf

TABLE 2. NUMBER OF ACTIVE NEST DAYS BY MONTH FOR 1600 NESTS OF 49 SPECIES MONITORED 1995 TO 2003.

Month	Active nest days (%)	Number of species
March	45 (0.1)	5
April	2803 (6.2)	26
May	21 787 (47.9)	43
June	18 141 (39.9)	42
July	2560 (5.6)	31
August	107 (0.2)	9

litter, and reduce competing vegetation. Careful consideration of the timing of prescribed fire is critical. While fire kills competing conifers, oaks are also damaged, although they are capable of rapidly resprouting following fire (McDonald and Tappeiner 1996). Burning in the fall before acorn drop and the loss of deciduous leaves may be optimal (Garrison et al. 2002).

We found no disruption in nesting activities of Hutton's Vireo during burning, nor did we find evidence of avoidance of burned areas, even within territories, in the year following fire. Though territories were occupied in both years, we do not know if these were the same individuals as few birds were banded. A resident species like Hutton's Vireo may be expected to have strong site fidelity due to high investment in the site. The retention of territories does, however, indicate that birds can be tolerant of changes due to prescribed fire. Retention of territories may also mean alternative territories are unavailable. More study is needed to understand the effect of fire on territory quality.

Few nests were destroyed during burning, but spring burning in our study also occurred outside the peak nesting season. Based on this small sample, approximately 17% of active nests suffered mortality. This is less than the 31% nest mortality reported for a mixed-grass prairie in North Dakota by Kruse and Piehl (1986) though in their study all were ground nests and burning was more extensive. In addition, we targeted nests of species that were potentially vulnerable; inclusion of all nests would have resulted in even lower overall nest mortality. Nesting birds were tolerant of burning activities and we saw no abandonment following burning despite active burning in close proximity to nests. Perhaps the most surprising result was the continued attendance of nests within actively burning areas. This tenacity has been observed by others (Leedy 1950, Moyle 1964, Fritzell 1975, Kruse and Piehl 1986) and suggests that birds are resilient to low severity fires even during breeding activities.

Compilation of the large nesting dataset indicated that breeding is compressed within a relatively short timeframe. Risk to nesting birds can be substantially reduced by adjusting timing of burns. In our study, 90% of all nesting activity occurred between 28 April and 2 July, with the majority in May and June. No particular patterns of nest activity by species were apparent and a diversity of species nested throughout the breeding season (Table 2). Thus, it appears that no group of species, such as residents or long-distance migrants, is especially vulnerable to fire during the nesting season, even at the margins. In the southern Sierra Nevada and similar

climates, burning before May or after June will avoid the majority of breeding activity. Other regions likely also have compressed nesting seasons that can aid avoidance of fire impacts.

We found few negative impacts from fire on avian species despite the fact that burning occurred in the spring when it has the potential to significantly influence breeding activities. Planning and implementation of management burns can and should include strategies that will lessen impacts on breeding birds. We encourage managers to consider not only short-term impacts of burning, but also long-term maintenance of important forest structures and species composition, as well as the potential of prescribed fire to reduce the risk of severe stand replacing fire. Management in relation to fire should take a broad view that can incorporate a landscape of burned patches of various ages rather than a myopic view of individual burns (Rotenberry et al. 1995).

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APPENDIX A. AVIAN SPECIES AND NUMBER OF MONITORED NESTS FROM 1995 TO 2003 USED TO COMPILE NEST ACTIVITY.

Species	Number of nests	Species	Number of nests
Mountain Quail <i>Oreortyx pictus</i>	4	Hutton's Vireo <i>Vireo huttoni</i>	160
Cooper's Hawk <i>Accipiter cooperii</i>	7	Warbling Vireo <i>Vireo gilvus</i>	14
Red-tailed Hawk <i>Buteo jamaicensis</i>	1	Steller's Jay <i>Cyanocitta stelleri</i>	34
Band-tailed Pigeon <i>Patagioenas fasciata</i>	2	Mountain Chickadee <i>Poecile gambeli</i>	7
Mourning Dove <i>Zenaidura macroura</i>	10	Bushtit <i>Psaltriparus minimus</i>	18
Northern Pygmy-Owl <i>Glaucidium gnoma</i>	2	Red-breasted Nuthatch <i>Sitta canadensis</i>	79
Great Gray Owl <i>Strix nebulosa</i>	1	Brown Creeper <i>Certhia americana</i>	66
Northern Saw-whet Owl <i>Aegolius acadicus</i>	2	Winter Wren <i>Troglodytes troglodytes</i>	11
Vaux's Swift <i>Chaetura vauxi</i>	1	American Dipper <i>Cinclus mexicanus</i>	3
Anna's Hummingbird <i>Calypte anna</i>	52	Golden-crowned Kinglet <i>Regulus satrapa</i>	2
Belted Kingfisher <i>Megaceryle alcyon</i>	2	Blue-gray Gnatcatcher <i>Poliophtila caerulea</i>	10
Acorn Woodpecker <i>Melanerpes formicivorus</i>	28	American Robin <i>Turdus migratorius</i>	83
Red-breasted Sapsucker <i>Sphyrapicus ruber</i>	8	Wrentit <i>Chamaea fasciata</i>	8
Downy Woodpecker <i>Picoides pubescens</i>	14	Nashville Warbler <i>Vermivora ruficapilla</i>	15
Hairy Woodpecker <i>Picoides villosus</i>	13	Yellow-rumped Warbler <i>Dendroica coronata</i>	4
White-headed Woodpecker <i>Picoides albolarvatus</i>	40	Black-throated Gray Warbler <i>Dendroica nigrescens</i>	24
Northern Flicker <i>Colaptes auratus</i>	39	Hermit Warbler <i>Dendroica occidentalis</i>	6
Pileated Woodpecker <i>Dryocopus pileatus</i>	1	Western Tanager <i>Piranga ludoviciana</i>	23
Olive-sided Flycatcher <i>Contopus cooperi</i>	2	Spotted Towhee <i>Pipilo maculatus</i>	163
Western Wood-Pewee <i>Contopus sordidulus</i>	48	Chipping Sparrow <i>Spizella passerina</i>	23
Hammond's Flycatcher <i>Empidonax hammondi</i>	32	Dark-eyed Junco <i>Junco hyemalis</i>	199
Dusky Flycatcher <i>Empidonax oberholseri</i>	1	Black-headed Grosbeak <i>Pheucticus melanocephalus</i>	140
Pacific-slope Flycatcher <i>Empidonax difficilis</i>	31	Purple Finch <i>Carpodacus purpureus</i>	32
Black Phoebe <i>Sayornis nigricans</i>	3	Lesser Goldfinch <i>Carduelis psaltria</i>	4
Cassin's Vireo <i>Vireo cassinii</i>	128		