CHAPTER 15.

Resiliency of Ponderosa Pine Forests to Bark Beetle Infestations Following Fuel-Reduction and Forest-Restoration Treatments

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INTRODUCTION

About 10 million ha of forests are classified as having moderate to high fire hazards in the Western United States (Stephens and Ruth 2005), many of which are characterized by ponderosa pine (*Pinus ponderosa*), an integral component of three forest cover types and a major component of >65 percent of all forests in the Western United States (Burns and Honkala 1990). Prior to Euro-American settlement, many ponderosa pine forests were open and parklike, as frequent thinning of small-diameter [<19 cm diameter at breast height (d.b.h.)] and fire-intolerant trees by low-intensity surface fires and competitive exclusion of tree seedlings by understory grasses maintained such conditions (Covington and Moore 1994). Today, these forests tend to be denser, have more small trees and fewer large trees, and are dominated by more shade-tolerant and fire-intolerant tree species such as white fir (*Abies concolor*). Consequently, fuel-reduction and forest-restoration treatments have been widely promoted to reduce the intensity and severity of future wildfires and to increase resilience to a multitude of other disturbances. When properly applied, prescribed fire, mechanical thinning, and their combination are effective for increasing residual vegetative resilience to wildfire (Agee and Skinner 2005, Stephens and others 2012). For example, Ritchie and others (2007) studied the effects of fuel-reduction and forest-restoration treatments at Blacks Mountain Experimental Forest, California on observed fire severity between treated and untreated stands impacted by a wildfire. Tree survival was highest in areas that were both thinned and prescribed burned. Survival in thinned-only areas was significantly greater than in untreated areas but less than in areas that had received the combined treatment.

Following prescribed fire, tree mortality may be immediate due to consumption of living tissue and heating of critical plant tissues, or can be delayed, occurring over the course of several years as a result of fire injuries to the crown, bole, or roots. Levels of delayed tree mortality are difficult to predict, and depend on numerous factors including tree species, tree size, phenology, degree of fire-related injuries, initial and postfire levels of tree vigor, the postfire environment, and the frequency and severity of other predisposing, inciting, and contributing factors (Stephens and others 2012). The propensity for bark beetles to attack fire-injured trees has led to questions regarding how the amount and distribution of bark beetle-caused tree mortality may negatively impact efforts to restore fire-adapted forest ecosystems with prescribed fire. Furthermore, fuel-reduction and forest-restoration treatments have functionally different effects on the structure and composition of residual forests and their resiliency to bark beetles. In particular, factors such as stand density, species composition, host density, average tree diameter, and average stand age are consistently identified as primary attributes associated with the severity of bark beetle infestations (Fettig and others 2007).

In recent years, numerous studies have examined the effects of fuel-reduction and forest-restoration treatments on the amount, distribution, and causes of tree mortality (Stephens and others 2012). Most were conducted on small experimental plots (<4 ha) for a relatively short period of time (1 to 5 years). The primary objective of this project was to evaluate the resiliency of different stand structures, created by applications of fuel-reduction and forest-restoration treatments, to bark beetle infestations at large spatial scales representing reasonable management scenarios in ponderosa pine forests a decade following prescribed burns.

METHODS

The project was executed at two locations: (1) Goosenest Adaptive Management Area, Klamath National Forest (41°30′ N., 121°52′ W.; 1500–1780 m elevation), in association with the Fire and Fire Surrogate Study (McIver and others 2013); and (2) Blacks Mountain Experimental Forest, Lassen National Forest (40°40′ N., 121°10′ W.; 1700–2100 m elevation), in association with the Blacks Mountain Experimental Forest Ecological Research Team (Oliver 2000). Due to page limitations, this summary focuses on Blacks Mountain Experimental Forest and expands on earlier research conducted there that details results 1 to 2 years after prescribed burns (Fettig and others 2008) and 3 to 5 years after prescribed burns (Fettig and McKelvey 2010).

Study Location

Blacks Mountain Experimental Forest was established in 1934 as a research facility for the study of forest management in interior ponderosa pine. The climate is characterized by hot, dry summers and cold, moist winters (Oliver 2000). Prior to treatment, stands were dominated by two age cohorts consisting of 300- to 500-year-old ponderosa and Jeffrey pines (*Pinus jeffreyi*) and incense cedar (*Calocedrus decurrens*) with a dense understory and midstory of 50- to 100-year-old pines, white fir, and incense cedar. There were 32.4 ± 1.5 (mean ± SEM, standard error of the mean) m^2/ha of basal area and 871 ± 58 trees/ha (Zhang and others 2008).

Treatments

Twelve experimental plots, 77 to 144 ha (mean = 111 ha), were established to create two distinct forest structural types: mid-seral stage (low structural diversity, LoD) and late-seral stage (high structural diversity, HiD). Each structure was randomly assigned to two plots within each of three blocks. Blocking allowed for allocating variation to differences in tree species composition associated with elevational gradients and year of treatment (see Fettig and McKelvey 2014 for schedule). LoD was created by removing large overstory trees and small understory trees leaving only trees of intermediate size, while HiD was attained by thinning smaller trees and retaining larger trees (Oliver 2000). Following harvest, half of each plot was prescribed burned (B). Burns were implemented using head, backing, and spot fires in fall. Hourly fuel moistures averaged 10.5 percent for 6- to 25-mm fuels (block 1) to 39.2 percent for duff (<20 mm) (blocks 1 and 2) (Fettig and others 2008). Following treatments, LoD and HiD averaged 10 and 25 m^2/ha of basal area and 282 and 513 trees/ha, respectively (Zhang and others 2008).

Data Collection and Analyses

A 100-percent cruise (census) was conducted on each plot to locate dead and dying pines and firs 2 (Fettig and others 2008), 5 (Fettig and McKelvey 2010), and 10 years after prescribed burns. While both incense cedar and western juniper (*Juniperus occidentalis*) are minor components of Blacks Mountain Experiment Forest (<8 percent of trees), these species are rarely attacked and killed by bark beetles and were ignored. All recently killed pines and firs >19 cm d.b.h. were tallied and the causal agent of mortality was identified. Tree species; d.b.h. (later placed into five diameter classes: 19.0–29.2, 29.3–39.3, 39.4–49.5, 49.6–59.7, and >59.7 cm); crown color; colonizing bark beetle species; presence of wood borers; and ranking of fire severity (1 to 4, based on external measures of bole char and bark consumption) were recorded (Fettig and others 2008). A section of bark ~625 cm^2 was removed on each recently killed tree with a hatchet at ~2 m in height on at least two aspects to determine if any bark beetle galleries were present in the phloem or cambium. The shape, distribution, and orientation of galleries were used to distinguish among bark beetle species. Bark removal also served as a means of separating mortality tallied during each of the three sample periods. This summary focuses on the mean percentage of trees killed by all causes, and by all bark beetle species, across all tree species. Responses for individual bark beetles species and host tree species can be found in Fettig and McKelvey (2014).

The experimental design was a randomized complete block with split plots with three blocks, two treatments (HiD and LoD), and two replicates per treatment. Due to an imbalance in the number of plots (i.e., because of one plot being impacted by mixed-severity wildfire in September 2002, necessitating its removal from the study), the Satterthwaite approximation method was used to estimate the appropriate degrees of freedom. An analysis of variance was performed on each response variable at $\alpha = 0.05$. If a significant treatment effect was detected, Tukey's multiple comparison test (Tukey's HSD) was used for separation of treatment means.

RESULTS AND DISCUSSION

A total of 188,793 pines and firs were monitored for mortality during the 10-year period. Of these, 106,314 were ponderosa pine, 63,636 were white fir, and 18,843 were Jeffrey pine.

Overall Tree Mortality

A total of 16,473 trees (8.7 percent of all trees) died during the 10-year period, of which 42.1, 5.2, and 52.7 percent were ponderosa pine, Jeffrey pine, and white fir, respectively.

The highest levels of tree mortality were observed during the initial sample period (1 to 2 years) (Fettig and others 2008) followed by the second (3 to 5 years) (Fettig and McKelvey 2010) and third (6 to 10 years) sample periods. This was expected, as fire-susceptible trees are often directly killed by the immediate effects of prescribed fire. Overall, tree mortality was concentrated (10,320 trees) in the smallest diameter class (24.1 cm), while the 54.7-cm diameter class had the lowest levels of tree mortality (252 trees).

Higher levels of tree mortality occurred on LoD + B (18.8 percent) compared to HiD (5.7 percent) and LoD (4.6 percent) ($p = 0.017$; all trees). Higher levels of tree mortality were also observed on LoD + B in the two smallest diameter classes ($p < 0.02$; 24.1 and 34.3), but in the two largest diameter classes, higher levels were observed in HiD + B ($p < 0.05$; 54.7 and >59.7). Interestingly, no significant treatment effect was observed for the 44.5-cm diameter classes ($p = 0.90$) (fig. 15.1). Fettig and others (2010) examined the effects of prescribed fire season (spring, fall, and none) on levels of tree mortality in ponderosa and Jeffrey pine forests in the central Sierra Nevada, California, and also reported few significant treatment effects in the intermediate diameter classes.

Bark Beetle-caused Tree Mortality

Western pine beetle and mountain pine beetle (*Dendroctonus ponderosae*) were observed colonizing ponderosa pine, Jeffrey pine beetle (*Dendroctonus jeffreyi*) was observed colonizing

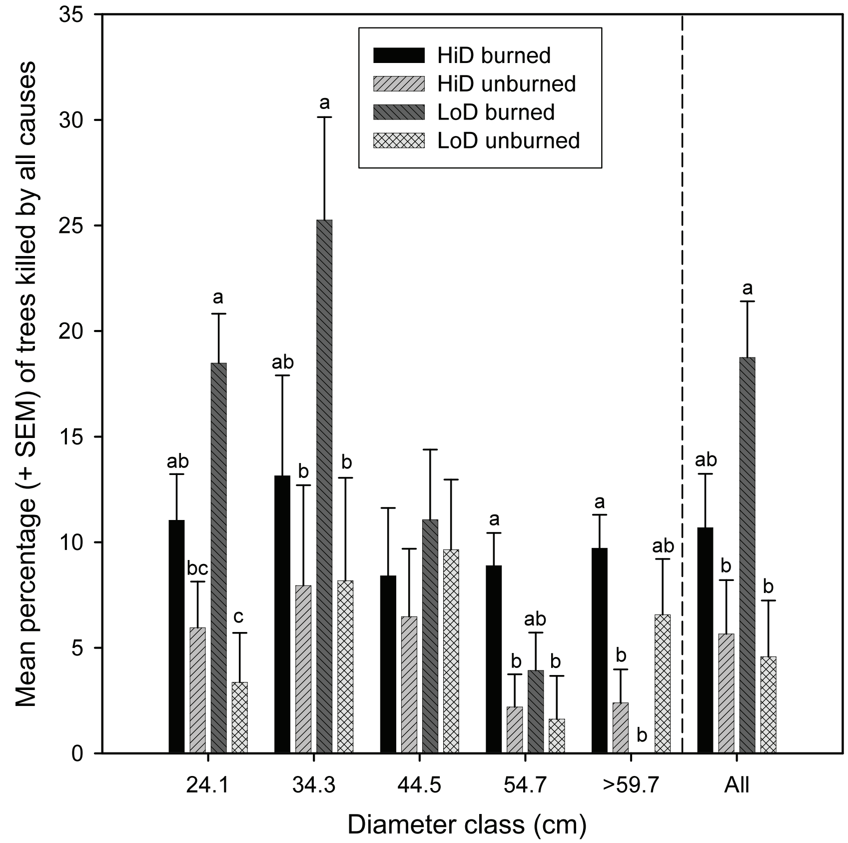

Figure 15.1—Mean percentage of trees killed by all sources by diameter class (midpoint of 10-cm diameter classes) and treatment (LoD = low structural diversity; HiD = high structural diversity) 10 years following prescribed burns, Blacks Mountain Experimental Forest, California. Means (+ SEM, standard error of the mean) followed by the same letter within groups are not significantly different (Tukey's HSD; p > 0.05) (from Fettig and McKelvey 2014).

Jeffrey pine, and fir engraver (*Scolytus ventralis*) was observed colonizing white fir. We also found pine engraver (*Ips pini*) and, to a much lesser extent, emarginate ips (*Ips emarginatus*) and *Ips latidens* (=*Orthotomicus)* colonizing ponderosa and Jeffrey pines. *Hylastes* spp., primarily *H. macer*; *Hylurgops* spp., primarily *H. subcostulatus*; and *Pseudohylesinus* spp. were occasionally observed but are not considered sources of tree mortality. Red turpentine beetle (*Dendroctonus valens*) was found colonizing many pines on burned-split plots (Fettig and others 2008), but their activity was largely limited to the 2 years following prescribed burns. Attacks by red turpentine beetle are usually confined to basal portions of stressed, weakened, or dead and dying pines, and are typically not considered a significant threat to tree health.

A total of 10,655 trees (5.6 percent of all trees) were killed by bark beetles (all bark beetle species combined) during the 10-year period (table 15.1). The highest levels of bark beetle-caused tree mortality were observed during the second sample period (4,193 trees), followed by the first (2,684 trees) and third (3,778) sample periods (table 15.2). Overall, bark beetle-caused tree mortality was concentrated (6,141 trees) in the smallest diameter class (24.1 cm), while the 54.7-cm diameter class had the lowest levels (178 trees) (table 15.1). Higher levels of bark beetle-caused tree mortality were observed on LoD + B than LoD in the 34.3-cm diameter class (15.4 and 7.3 percent, respectively) and for all trees (8.7 and 4.2 percent, respectively) ($p < 0.05$). HiD + B (6.4 percent) exhibited higher levels of bark beetle-caused tree mortality than HiD (1.7 percent) in the 54.7-cm diameter class ($p < 0.05$). HiD + B (6.6 percent) also exhibited higher levels of bark beetle-caused tree mortality than LoD + B (0 percent) in the >59.7-cm diameter class ($p = 0.02$) (fig. 15.2). No other significant differences were observed. Approximately 60.5 percent of all bark beetle-caused tree mortality occurred on burned-split plots.

CONCLUSIONS

During the 10-year period, 8.7 percent of all trees died, most of which was attributed to bark beetles (64.7 percent), primarily fir engraver, mountain pine beetle, and western pine beetle. Bark beetles killed trees of all ages and size classes, but mortality was generally concentrated on HiD (64.3 percent), on burned-split plots (60.5 percent), within the two smallest diameter classes (87 percent), and 3 to 5 years after prescribed burns. These observations were consistent for all bark beetle species, with few exceptions (Fettig and McKelvey 2014). The observation concerning bark beetle-caused tree mortality concentrated 3 to 5 years after prescribed burns differs from many other similar studies that reported mortality was concentrated during the first year or two (Stephens and others 2012). However, many of those studies were of limited duration (<3 years). Relatedly, it is important to note that the treatment effects observed in our study varied by sample period (Fettig and McKelvey 2010, 2014; Fettig

Table 15.1—Numbers of trees killed by bark beetles 10 years following prescribed burns, Blacks Mountain Experimental Forest, California (from Fettig and McKelvey 2014)

Treatment[a]	d.b.h. class[b]	*Dendroctonus brevicomis*	*Dendroctonus ponderosae*	*Dendroctonus jeffreyi*	*Ips* spp.	*Scolytus ventralis*	Total
HiD + B	24.1	190	448	7	129	1,354	2,128
	34.3	198	227	11	28	537	1,001
	44.5	102	59	2	1	157	321
	54.7	66	10	2	2	27	107
	>59.7	250	28	2	1	14	295
	All	806	772	24	161	2,089	3,852
HiD	24.1	119	450	1	2	1,071	1,643
	34.3	89	175	1	0	626	891
	44.5	33	37	2	0	200	272
	54.7	27	10	0	0	29	66
	>59.7	100	13	0	0	17	130
	All	368	685	4	2	1,943	3,002
LoD + B	24.1	65	221	5	252	1,151	1,694
	34.3	86	104	3	60	549	802
	44.5	18	13	1	2	66	100
	54.7	0	4	0	0	—	4
	>59.7	—	—	0	—	0	0
	All	169	342	9	314	1,766	2,600
LoD	24.1	12	68	4	6	586	676
	34.3	27	30	1	1	381	440
	44.5	11	3	0	0	67	81
	54.7	1	0	0	0	—	1
	>59.7	0	2	0	0	1	3
	All	51	103	5	7	1,035	1,201
Total		**1,394**	**1,902**	**42**	**484**	**6,833**	**10,655**

— = No hosts present.

[a] LoD, low structural diversity; HiD, high structural diversity. LoD was created by removing large overstory trees and small understory trees leaving only trees of intermediate size, while HiD was attained by thinning smaller trees and retaining larger trees. Following harvest, half of each plot was prescribed burned (+ B).

[b] d.b.h. = diameter at breast height in cm; value is the midpoint of the size class (except for the largest class).

Table 15.2—Trees killed by bark beetles by sample period and overall, Blacks Mountain Experimental Forest, California (from Fettig and McKelvey 2014)

Sample period	Interval (years)	*Dendroctonus brevicomis*	*Dendroctonus ponderosae*	*Dendroctonus jeffreyi*	*Ips* spp.	*Scolytus ventralis*	Total
1	1 to 2	442	468	18	456[a]	1,300	2,684
2	3 to 5	747	947	17	15	2,467	4,193
3	6 to 10	205	487	7	13	3,066	3,778
Total		**1,394**	**1,902**	**42**	**484**	**6,833**	**10,655**

[a] Fettig and others (2008) erroneously reported this value as 494.

and others 2008), further emphasizing the importance of long-term monitoring.

Oliver (1995) reported maximum stand density index (SDI) for even-aged ponderosa pine stands in northern California was regulated by bark beetle infestations. An SDI value of 230 defined a threshold for a zone of imminent bark beetle-caused tree mortality within which endemic populations kill a few trees but net growth is positive. Maximum SDI was defined at 365. While the SDI relationships described by Oliver (1995) are a tenuous fit for HiD, which does not represent an even-aged structure, it is quite appropriate for LoD. LoD plots averaged SDI values of 118 and 124 for the unburned and burned-split plots, respectively (Fettig and others 2008). These values are much less than the threshold SDI value of 230. As such, higher levels of bark beetle-caused tree mortality should be expected following similar treatments that retain higher residual stand densities, independent of the confounding effects of prescribed fire. No significant differences were observed between HiD and LoD for any of the variables analyzed within any diameter class (data not shown), suggesting that over the 10-year period, these structures were of similar resilience to bark beetle infestations and other disturbances.

Concerns about maintaining large-diameter trees, particularly pines on HiD + B, have been expressed by members of the interdisciplinary team at Blacks Mountain Experimental Forest (Ritchie and others 2008). The majority (77.9 percent) of mortality in the largest diameter class occurred during the first 5 years following prescribed burns. During this time, significantly higher levels of tree mortality were observed on HiD + B (8.4 percent) compared to HiD (1.2 percent) (Fettig and McKelvey 2010); however, during the second 5 years, no significant difference was observed between these treatments. There are insecticide- and semiochemical-based tools available that could be selectively used to protect trees most susceptible to colonization by bark beetles (Fettig and Hilszczański 2014). Furthermore, methods such as raking of litter and duff a short distance from the base of large-diameter ponderosa pines have been shown effective for reducing fire severity and subsequent levels of tree mortality when applied prior to burning (Fowler and others 2010). Such techniques might be considered in the future for reducing levels of tree mortality in the large-tree component following similar treatments.

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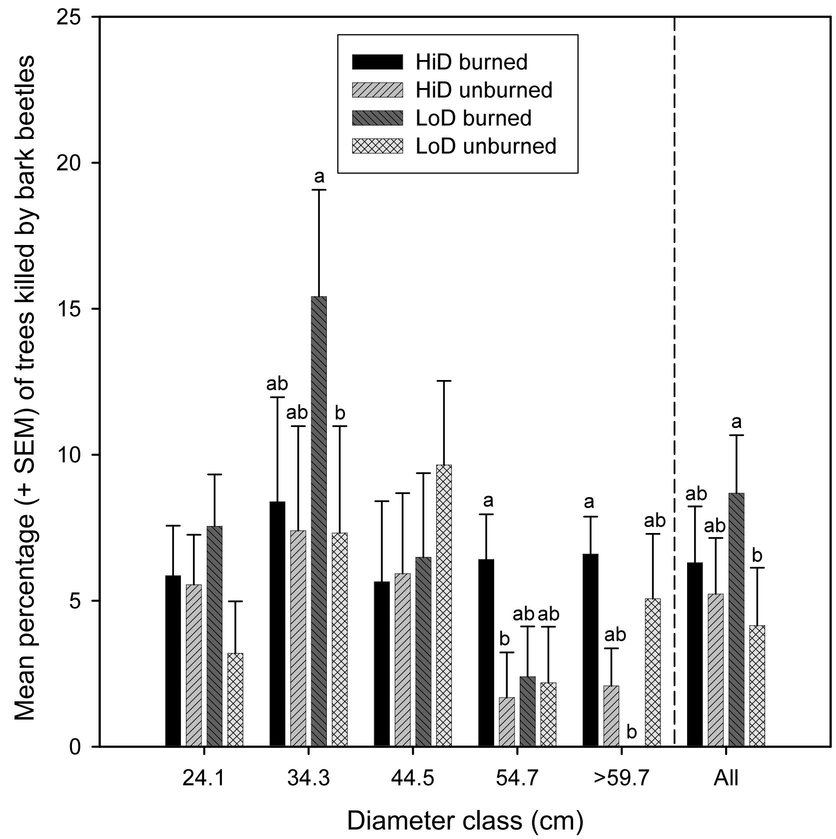

Figure 15.2—Mean percentage of trees killed by bark beetles (all species) by diameter class (midpoint of 10-cm diameter classes) and treatment (LoD = low structural diversity; HiD = high structural diversity) 10 years following prescribed burns, Blacks Mountain Experimental Forest, California. Means (+ SEM, standard error of the mean) followed by the same letter within groups are not significantly different (Tukey's HSD; $p > 0.05$*) (from Fettig and McKelvey 2014).*

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