Changes in Downed Wood and Forest Structure After Prescribed Fire in Ponderosa Pine Forests

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Abstract—*Most prescribed fire plans focus on reducing wildfire hazards with little consideration given to effects on wildlife populations and their habitats. To evaluate effectiveness of prescribed burning in reducing fuels and to assess effects of fuels reduction on wildlife, we began a large-scale study known as the Birds and Burns Network in 2002. In this paper we analyze changes in downed wood and forest structure (trees and snags) measured within one year after prescribed fire treatments that were completed in ponderosa pine (Pinus ponderosa) forests in Arizona and New Mexico (Southwest region), and Idaho and Washington (Northwest region). Apparent reductions in downed wood and trees were observed in both regions. However, statistically significant reductions of downed wood were found primarily in the Northwest ($p < 0.001$), whereas significant reductions of trees were reported only for the Southwest ($p = 0.03$). No significant post-treatment changes were detected in snag densities, although we observed a pattern of non-significant increases in all size classes. Additional fire treatments are likely needed to meet fuels reduction goals. Results of this study are intended to assist managers with developing scientifically sound and legally defensible prescribed fire projects that will reduce fuels and concurrently enhance wildlife habitat.*

Introduction

Fire regimes of lower elevation forests, particularly ponderosa pine (*Pinus ponderosa*) of the Interior Western United States, have been altered since Euro-American settlement (Agee 1993; Schoennagel and others 2004). Alterations in fire regimes and subsequent changes in forest structure and composition stem primarily from fire suppression, logging, and livestock grazing (Allen and others 2002; Schoennagel and others 2004; Veblen 2000). After decades of fire suppression, elevated fuel loads in many ponderosa pine forests have increased the likelihood of unusually large and severe fires (Arno and Brown 1991; Covington and Moore 1994), and the area burned annually has increased (Grissino-Mayer and Swetnam 2000; Keane and others 2002).

In an effort to restore ponderosa pine forest ecosystems, land managers have increasingly relied on prescribed burning (Horton and Mannan 1998; Arno 2000; Machmer 2002; Carey and Schumann 2003). Most prescribed fire plans focus on reducing the intensity of wildfire, with little consideration given to effects on wildlife populations and their habitats. Strategies for fire management should not only reduce fire risk but also maintain habitat for wildlife and other components of biodiversity (Saab and others 2005).

Ponderosa pine trees, snags and downed wood are among the most valuable habitat components for wildlife species in western North American forests (Balda 1975; Bull and others 1997; Hall and others 1997; Szaro and others

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1988). Large-diameter ponderosa pine snags, trees with decay, and downed logs are relatively easy to excavate by woodpeckers and provide roosting, nesting, and foraging habitat for a variety of wildlife (Bull and others 1997; Hall and others 1997; Szaro and others 1988; Scott 1979).

Many cavity-nesting birds depend on fire-disturbed landscapes for breeding, dispersal, and other portions of their life history (Saab and others 2004). Several cavity nesters are designated by state, federal, and provincial governments as species at-risk because they are responsive to fire and timber management activities. Stand-replacement fires in conifer forests are particularly important to breeding and wintering cavity-nesting birds (Blackford 1955; Raphael and White 1984; Saab and Dudley 1998; Kriesel and Stein 1999; Hannon and Drapeau 2005). Little is known, however, about bird population responses to prescribed fire, particularly in the Intermountain region (Bock and Block 2005; Saab and others 2005). In 2002, we began a regional study to evaluate effectiveness of prescribed fire in reducing fuels and to assess the effects of fuels reduction on habitats and populations of birds in ponderosa pine forests throughout the Interior West. Our study is known as the Birds and Burns Network (BBN) (see web page http://www.rmrs.nau.edu/lab/4251/birdsnburns/), with study areas located in seven states encompassing much of the range of ponderosa pine in the United States (Arizona, Colorado, Idaho, Montana, New Mexico, Oregon, and Washington). As of 2005, study areas in Arizona and New Mexico (Southwest region; SW), and Idaho and Washington (Northwest region; NW) have received prescribed fire treatments.

In this paper, our objective was to evaluate the magnitude of change in the quantities of downed wood, dead stems (hereafter termed snags), and live stems (hereafter termed trees) measured within one year after prescribed fire treatments. Based on previous studies (Horton and Mannan 1988; Machmer 2002; McHugh and Kolb 2003; Raymond and Peterson 2005), we hypothesized that downed wood of all sizes, large snags (≥ 9 inch diameter breast height [d.b.h.]), and smaller trees (< 9 inch d.b.h.) would be reduced as a result of prescribed burning, whereas we expected smaller snags (< 9 inch d.b.h.) to increase after fire treatments. Results of this study are intended to help managers develop scientifically sound and legally defensible prescribed fire projects that will reduce fuels and concurrently maintain and enhance wildlife habitat.

Study Area and Methods

Study areas were located in forests dominated by ponderosa pine, where prescribed fire treatments were implemented by the USDA National Forests. On each study area, a single treatment unit ranged in size from 500 to 1000 acres and was paired with an unburned control unit of equivalent area. As of 2005, prescribed fire treatments were completed at seven study units in four states and data from these units were used in the analyses presented in this paper. General objectives of these "low-intensity" fire treatments included fuels reduction, fire threat mitigation, and forest restoration.

Pre-treatment data were collected during the summers of 2002 and 2003. Four units were treated with fire in the SW on USDA National Forests (NF); two units during fall 2003 in Arizona (Apache-Sitgreaves and Coconino NFs), and two units that were initiated in fall 2003 and completed during spring 2004 in New Mexico and Arizona (Gila and Kaibab NFs, respectively). Three units were treated in the NW during spring 2004, one unit in Idaho

(Payette NF) and two units in Washington (Okanogan and Wenatchee NFs). Post-treatment data were collected one growing season after fire treatments during the summers of 2004 or 2005.

Overstory vegetation (trees ≥ 9 inches d.b.h.) on all units in both regions was dominated by ponderosa pine. For trees ≥ 20 inch d.b.h. or larger, ponderosa pine was also the dominant tree species for all locations except for the Gila NF, where alligatorbark juniper (*Juniperus deppeana*) had higher densities.

In Arizona, common understory vegetation included green rabbitbrush (*Chrysothamnus viscidiflorus*) and Fendler rose (*Rosa woodsii*), whereas gambel oak dominated the understory in New Mexico. Arizona fescue (*Festuca arizonica*) and blue gramma (*Bouteloua gracilis*) were the most common grass species throughout the SW. Elevations in the SW region ranged from 6800 feet on the Coconino NF to nearly 8200 feet on the Gila NF.

The understory vegetation in the NW was comprised of various species, including snowberry (*Symphoricarpos albus)*, spirea (*Spirea* spp.), serviceberry (*Amelanchier alnifolia*), and chokecherry (*Prunus* spp.). Bluebunch wheatgrass (*Pseudoroegenaria spicatus*) and Idaho Fescue (*Festuca idahoensis*) were the common grass species. Elevations ranged from 2200 feet in Washington to 6500 feet in Idaho.

Within each unit we established 20 to 40 permanently marked 1-acre random plots to measure fuel and vegetative characteristics. All plots centers were at least 820 feet apart (Dudley and Saab 2003). To determine the effects of prescribed fire on downed wood, snags, and trees, we measured these forest components at each plot before (pre) and after (post) prescribed fire. We considered the difference in pre and post values by plot to be a measure of the treatment effect size.

Measurements were nested within the 1-acre plot configured as two 66 x 326 feet rectangles that crossed in the center (that is, a rectangular cross plot). Tree and snag measurements followed methods outlined by Bate and others (1999). Snags ≥ 9 inches d.b.h. were counted within 33 feet of the centerline in the rectangular cross plot. Trees ≥ 9 inches d.b.h. were counted within 16.5 feet of the centerline in the SW and within 9.8 feet in the NW. Plot widths for trees and snags were based on power analyses of pilot data from each location to maximize efficiency of data collection (Bate and others 1999). For trees and snags < 9 inches, we counted within 6.6 feet of the centerline in the SW and within 3.3 feet in the NW.

In this paper we present preliminary results for both snags and trees in four categories: (1) < 3 inch; (2) ≥ 3 to 9 inch; (3) ≥ 9 inch; and (4) total density of all stems (snags or trees). Snags and trees in the ≥ 9 inch d.b.h. category were of special interest to us because they commonly represent the smallest size class that woodpeckers use for nesting (for example, Saab and others 2004) and the smallest sized trees harvested for timber values (USDA 1996).

We measured the weight (tons per acre) of downed wood following Brown's (1974) protocol. Downed wood is defined as the "… dead twigs, branches, stems, and boles of trees and brush that have fallen and lie on or above the ground" (Brown 1974, page 1). Downed wood pieces less than 1 inch diameter (1- and 10-hour fuels) were sampled along 41 feet of transect in two directions (north and south) from the plot center. Material in the ≥ 1 to 3 inch size class (100-hour fuels) was measured in the same two directions but along twice the length (82 feet). For coarse wood ≥ 3 inches (≥ 1000-hour fuels), we recorded the intersection diameter of each woody piece along 164 feet in each of the four cardinal directions originating from the plot center (total of 656 feet sampled). Downed wood pieces ≥ 3 inch were classified as either sound or rotten and we used the specific gravities provided by Brown (1974) to obtain a weight estimate for each condition class. That is, we used 24.96 lbs/ft^3 and 18.72 lbs/ft^3 for sound and rotten wood, respectively, relative to the density of water (62.4 lbs/ft^3) (Brown and See 1981). Here, we present results for downed wood in four size categories: (1) < 3 inch; (2) ≥ 3 inch; (3) ≥ 9 inch; and (4) total weight of all downed wood. Weights calculated for the ≥ 9 inch category were based on the large-end diameter (LED), whereas weights of other size classes were based on the intersect diameter.

We calculated a response to the prescribed fire as an "effect size" on each plot, which represented the change in fuels attributable to the prescribed fire. The effect size was measured by subtracting pre-fire fuel quantities from post-fire fuel quantities. We then computed least-squares means (PROC MIXED SAS Institute 2003) to test whether the effect size was significantly different from zero for weight of downed wood, snag densities, and tree densities. We accepted $p \leq 0.05$ as the observed probability level for Type I error in hypothesis tests. We used a nested analysis with plots nested within units, and units nested within regions. Results are reported for the mean effect size of stems per acre (± 1 standard error [SE]) and tons per acre (± 1 SE) at the regional level. A likelihood ratio test was computed to compare a model with a pooled estimate of variance across regions to a model with a separate variance estimate for each region. Generally, the model with separate variance estimates had significantly better goodness-of-fit and was used for the least-squares means analysis. Pooled variance results are reported only for trees ≥ 9 inch d.b.h. and for total trees.

Results

Weight of downed wood in all size classes decreased after prescribed fire treatments (Table 1), however most of the statistically significant differences were measured in the NW (Table 2). Downed wood was reduced by 25 to 43 percent in the SW and by 29 to 58 percent in the NW. Total weight of downed wood was reduced by nearly half in the NW region, where most of the downed material was comprised of large logs ≥ 9 inches LED (Table 1). In contrast to the NW, pre-fire weight of downed wood in the SW region was composed almost exclusively of small diameter material < 9 inches LED (Table 1).

Our hypothesis about reductions of small diameter (< 9 inches d.b.h.) trees (seedlings, saplings, and poles) was generally supported by the data; however, trees of all diameter classes in the SW region also decreased significantly after fire treatments (Table 2). Trees were reduced by 19 to 74 percent in the SW and 0 to 39 percent in the NW (Table 1). Stems in the smallest size class (<3 inches) contributed the most to changes in tree densities, whereas large tree (≥ 9 inches) densities changed the least.

We hypothesized that snags of the smaller size classes (< 9 inches d.b.h.) would increase and that large snags (≥ 9 inches d.b.h.) would decrease after fire treatments. Our results indicated no significant post-treatment changes in snag densities (Table 2), although we observed a pattern of non-significant increases in all size classes (Table 1). Increases in snags ranged from 30 to 72 percent in the SW and 29 to 229 percent in the NW. Large snags (≥ 9 inches d.b.h.) contributed to the greatest changes in dead stems in the SW, whereas smaller diameter stems (≥ 3 – 9 inches d.b.h.) contributed most to snag changes in the NW.

Table 1—Means, standard errors (SE), and percent change for downed wood (DW; mean tons per acre), and trees, and snags (mean stems per acre) measured pre- and post-fire treatment by region (Southwest [SW] and Northwest [NW]) in the Birds and Burns Network during 2002-2005. Downed wood was measured at large end diameter (LED) and stems were measured at diameter breast height (d.b.h.).

		SW [n = 134]			NW [n = 60]		
	Size class	Pre-fire mean	Post-fire mean	Percent change	Pre-fire mean	Post-fire mean	Percent change
	(inches)	*(SE)*			*(SE)*		
DW	< 3	2.0 (0.2)	1.5 (0.1)	–25	1.8 (0.2)	1.3 (0.1)	–27.8
(tons/ac)	≥ 3	2.3 (0.2)	1.3 (0.1)	–43.5	7.6 (0.8)	3.8 (0.5)	–50
	≥ 9	0.7 (0.1)	0.4 (0.1)	–42.9	6.3 (0.8)	2.6 (0.4)	–58.7
	Total	4.3 (0.3)	2.8 (0.2)	–34.9	9.4 (0.9)	5.1 (0.5)	–46
Trees	< 3	256 (26.5)	66.3 (12.6)	–74.1	234 (29.4)	144 (26.6)	–38.5
(stems/ac)	≥ 3 to 9	124 (9.5)	72.6 (8.4)	–41.5	191 (23.6)	133 (17.1)	–30.4
	≥ 9	52.2 (2.5)	42.2 (2.9)	–19.2	45.5 (2.4)	48.1 (2.9)	+.057
	Total	432 (33.2)	181 (18.2)	–58.1	470 (46.8)	324 (36.2)	–31.1
Snags	< 3	28.6 (3.9)	44.6 (5.6)	+55.9	62.2 (10.2)	110 (16.7)	+76.8
(stems/ac)	≥ 3 to 9	15 (2)	19.5 (2.4)	+30	12.7 (2.1)	41.8 (7.9)	+229
	≥ 9	2.5 (0.3)	4.3 (0.7)	+72	2.8 (0.4)	3.6 (0.5)	+28.6
	Total	46 (5.4)	68.4(7.5)	+48.7	77.6 (11.5)	156 (23.5)	+101

Table 2—Results of least–square means analysis to test for statistical differences from zero, or no change in the quantity of downed wood (DW; tons per acre), trees (stems per acre), and snags (stems per acre) measured pre- and post-prescribed fire in western ponderosa pine forests. Mean estimate of the effect size, standard error of the estimate (SE), t-value, *p*-value, and sample size [n] are reported for each size class by region (Southwest [SW] and Northwest [NW]) in the Birds and Burns Network during 2002-2005.

		SW [n =134]				NW [n = 60]			
	Size class	Estimate (effect size)	SE	t-value	*p*-value	Estimate (effect size)	SE	t-value	*p*-value
	(inches)								
DW	< 3	–0.49	0.32	–1.53	0.19	–0.46	0.43	–1.06	0.34
(Δ in tons/ac)	≥ 3	–1.15	0.43	–2.68	0.04	–3.9	0.58	–6.71	0.001
	≥ 9	–0.4	0.18	–2.11	0.09	–3.7	0.54	–6.83	0.001
	Total	–1.66	0.73	–2.28	0.07	–4.3	0.36	–11.97	<0.001
Trees	< 3	–212.6	28.6	–2.7	0.04	–107.1	47.5	–2.26	0.07
(Δ in stems/ac)	≥ 3 to 9	–60.9	29.5	–2.07	0.09	–61.1	35.2	–1.74	0.14
	≥ 9	–13.3	8.06	–1.65	0.16	2.9	9.38	0.31	0.77
	Total	–287.1	99.9	–2.37	0.03	–165.2	117.3	–1.41	0.22
Snags	< 3	19.4	13.7	1.42	0.21	47.9	19.5	2.45	0.06
(Δ in stems/ac)	≥ 3 to 9	5.4	4.2	1.29	0.25	29.2	17.6	1.65	0.16
	≥ 9	1.86	1.06	1.75	0.14	0.79	0.55	1.43	0.21
	Total	26.6	14.8	1.80	0.13	77.9	37.2	2.09	0.09

Discussion

Decreases in downed wood and trees supported our hypotheses regarding changes in these forest components after prescribed fire treatments. While we expected only the smaller size classes of snags (< 9 inch d.b.h.) to increase after prescribed fire, we observed a pattern of non-significant increases in large snag (≥ 9 inch d.b.h.) densities as well. Apparently, prescribed fire treatments were severe enough to kill trees of all size classes, particularly in the SW where this result was statistically significant.

Nearly half of large downed wood (≥ 9 inch LED) was consumed by prescribed fire in both regions. Drought conditions, followed by low wood moistures prior to fire treatments, may have contributed to the large loss of downed wood. When moisture contents are less than 15 percent, fire generally consumes about half of large downed woody materials (Brown and others 1985). Efforts to retain these large structures may require seasonal adjustments for burning times when moisture contents are higher and fire severity effects are lower (Thies and others 2005). Maintenance of large, downed wood is important ecologically because these structures provide foraging habitat, thermal cover, and concealment for many sensitive wildlife taxa (Bull and others 1997; Szaro and others 1988), although logs may have been a limited resource in low-severity fire regimes (Agee 2002).

Overall tree densities in the SW were significantly reduced after fire treatments. Although we observed a pattern of decreased tree densities in the NW, no statistical differences were detected in densities measured before and after prescribed fire. We think, however, that all observed changes in tree densities were important ecologically. For example, in both regions we observed the greatest reduction of tree densities in the smallest size class (< 3 inches d.b.h.), followed by reductions in the medium size class (≥ 3 to < 9 inch), with little change in large (≥ 9 inches d.b.h) tree densities. Small diameter trees function as ladder fuels in dense stands by carrying flames into the crowns of mature trees, where the potential for larger tree mortality increases (Pollet and Omi 2002). Indeed, prescribed fire programs that remove small diameter trees can reduce the likelihood and cost of stand-replacing fires (Arno 1980; Fernandes and Botelho 2003; Pollet and Omi 2002).

We observed relatively little change in densities of large trees ≥ 9 inch d.b.h. This result was not surprising because the thick bark of ponderosa pine is fire-resistant, improving tree survival during low to moderate severity burns (Agee 1993). Historically, large-diameter ponderosa pines were harvested because of their high timber and fuelwood values (Agee 1993). These same trees are also among the most valuable for many wildlife species of conservation concern (Bull and others 1997; Lehmkuhl and others 2003; Saab and others 2004). Retention of large-diameter snags and decayed trees, particularly ponderosa pine, can provide vital nesting and roosting habitat for a variety of wildlife species (Bull and others 1997; Martin and Eadie 1999). For example, the sapwood of ponderosa pine is relatively thick compared to other conifers and exceptionally valuable for the excavation of nesting and roosting tree cavities (Bull and others 1997).

We observed apparent increases in snag densities, including the large diameter size class in both regions. While this pattern was not statistically significant, the result has implications for the creation of wildlife habitat. Maintenance and recruitment of larger diameter snags is particularly important because large snags have greater longevity and provide wildlife habitat for a longer period of time than smaller snags (Raphael and Morrison 1987; Everett and others 1999; Saab and others 2004). Additional tree mortality is expected two to three years after fire, because time allows for crown scorch and consumption to cause further tree death (McHugh and Kolb 2003).

In contrast to our results that suggest increased densities of large snags after fire, Horton and Mannan (1988) reported that large ponderosa pine snags were reduced by about 50 percent within the first year after a moderately-intense prescribed fire. Detrimental effects of prescribed fire on suitable nesting snags were also reported in ponderosa pine forests of Canada, where burning caused heavy scorching of large snags (Machmer 2002). Differences in fire severity among studies likely contributed to the opposing results of snag changes after prescribed fire.

Several authors suggest protecting nest trees by removing combustible materials around their base prior to burning to reduce losses of suitable nest/roost snags (Horton and Mannan 1988; Machmer 2002; Tiedemann and others 2000). Specifically, Horton and Mannan (1988) recommend protecting large (>50 cm [20 inch] d.b.h.) snags and logs with moderate decay. Tiedemann and others (2000) recommended removing combustible material around snags > 30 cm (12 inch) d.b.h. These methods are labor intensive and cost prohibitive for large-scale prescribed fire programs, unless snag protection is required for Threatened and Endangered species. While prescribed fire consumed some wildlife snags, burning also recruited snags (Table 1). Direct effects of prescribed burning on wildlife should also be considered. For example, prescribed fires conducted during spring or early summer may cause direct mortality to nestlings and fledglings (Lyon and others 2000).

Smaller snag (< 9 inch d.b.h.) densities increased 30 to 60 percent in the SW and two to four times that amount in the NW region. While still standing, these dead trees contribute to increased risk of spot fires (Stephens and Moghaddas 2005). With time, these stems create ground fuels and increase the likelihood of higher fire intensities (Reinhardt and Ryan 1998). Such fuel accumulations can limit the effectiveness of prescribed fire programs to a relatively short period of time such as two to four years (Fernades and Botelho 2003). Studies suggest that relatively frequent, natural fires are necessary to maintain ponderosa pine forests in a diverse landscape mosaic more common to historical conditions (Brown and Cook 2006; Fry and Stephens 2006) that existed just prior to European settlement. Similarly, prescribed fires also have the potential to mitigate the likelihood of severe crown fires (Fernandes and Botelho 2003; Finney and others 2005; Pollet and Omi 2002; Raymond and Peterson 2005), which were once rare but regular events in ponderosa pine forests (Shinneman and Baker 1998).

Few of our results were statistically significant at $p \leq 0.05$. Managers willing to take more acceptable risk can interpret our results as being more definitive by using a significance level of $p \leq 0.10$ (Zar 1999). Inherent differences in pre-treatment forest structure existed in our ponderosa pine forests, which possibly influenced fire behavior and resulted in high variability in the effectiveness of fuels reduction. The power to detect statistically significant changes is low without large numbers of replicates. However, long-term prescribed fire programs can still play an important role in reducing fire hazard potential (Fernandes and Botelho 2003), suggesting that our study areas may require multiple fire treatments to reach fuels reduction and restoration goals. In addition, wildland fire can also be used to effectively reduce fuels and to closely mimic past disturbance regimes in ponderosa pine forests (Baker and Ehle 2001).

In this paper we did not evaluate the influence of fire severity on changes in fuels and other vegetation after fire. In the future, we plan to incorporate fire severity data to help with understanding the influences of severity on vegetation mortality, and wildlife populations and their habitat (Saab and Powell 2005). Also, we recommend monitoring vegetation and wildlife populations for several years after prescribed burning because of changes in vegetation and wildlife responses with time since fire (Hannon and Drapeau 2005; McHugh and Kolb 2003; Reinhardt and Ryan 1998; Saab and others 2004). Severity information and monitoring for multiple years after fire will help in developing guidelines for prescribed fire projects that will reduce fuels and concurrently create wildlife habitat.

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