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Snag and woody debris dynamics following severe wildfires in northern Arizona ponderosa pine forests

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Abstract

Following severe wildfires in southwestern ponderosa pine forests, dead trees remain on the landscape and eventually fall, but relatively little is known about the quantity and quality of post-wildfire coarse woody debris (CWD). To describe post-fire conditions, we measured snags, CWD, and fine woody debris and forest floor depth on seven fires in a chronosequence from 3 to 27 years old in northern Arizona. Snags declined in density with increasing time since fire and generally followed expected patterns of decay, except that few snags stood long enough to reach a clean-bark state. The mean biomass of the surface CWD ranged from as low of 3.3 Mg ha^{-1} to a high of 41.3 Mg ha^{-1}. Total CWD biomass in the surface fuel load remained roughly comparable from 8–9-year-old fires to a 27-year-old fire but the state of the CWD changed from sound to rotten. The change to a rotten condition suggests an increase in ignitability of the post-fire fuel load, but fine fuels that could support high fireline intensity were relatively low. The number of "jackstraws," points where intersecting downed logs could create a hot spot if reburned, was slightly higher in the oldest fire. Few fire-created snags remained by the 27th year post-fire. Management options to reduce fuels after severe wildfire, such as salvage logging, prescribed burning, or passive management, should be addressed in a broad ecological context.

Keywords: Chronosequence; Coarse woody debris; Crown fire; Fuel load; Fire hazard

1. Introduction

Following severe wildfires, dead trees remain on the landscape and eventually fall. In some circumstances, the accumulation of post-wildfire coarse woody debris (CWD) may support relatively severe reburning. For example, after the initial Tillamook fire in 1933 burned 121,200 ha of Douglas-fir forest in Oregon, the same area burned again in 1939 and 1945 (Pyne, 1997). The reburns were difficult to suppress, burning two-thirds of the original acreage and destroying regeneration that had established after the first fire (Kemp, 1967). More recently, Odion et al. (2004), in mixed conifer forests of the Klamath-Siskiyou Mountains of California, found that there was three times more high severity fire in reburn areas than those that had not burned in at least 84 years.

Ponderosa pine (*Pinus ponderosa* P. & C. Lawson) forests in the southwestern United States have undergone changes in structure and function in the last 100 years, making them more vulnerable to severe wildfire (Dahms and Geils, 1997). Compared to historically open forest conditions, modern forests have more young and small trees, and heavier fuel loads (Cooper, 1960; Covington et al., 1994). Concurrent with this change is the increase in the number and size of wildfires in the western United States (Swetnam et al., 1999). Management of post-wildfire fuel complexes, and concern over the prospect of reburning, have thus increasingly come to the forefront of southwestern forest management. Post-fire management is often contentious, especially when salvage harvesting is suggested (McIver and Starr, 2001). Reasons given for salvage logging include reduced erosion (McIver and Starr, 2001), insect outbreak (Simon et al., 1994), and lower intensity and severity of future fires (USDA, 2004), though McIver and Starr (2001) found little objective evidence that a reduction in fire intensity occurred after an area was salvage logged.

Our goal in the present study is to investigate post-fire fuel dynamics, the decay and fall of standing snags, and the accumulation of fallen coarse woody debris on the surface, in order to help inform decisions about post-fire management, especially with respect to potential future fire behavior and effects. Both standing and fallen dead wood can be referred to

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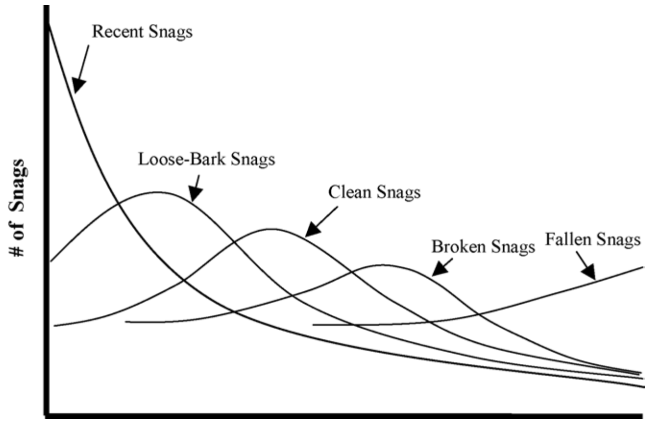

Fig. 1. Hypothesized changes in snag decay classes over time following a severe wildfire. The decay classes are from Thomas et al. (1979).

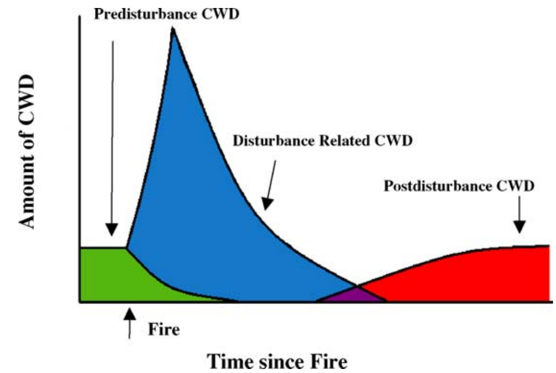

Fig. 2. Theoretical patterns of CWD types and amounts over time after a disturbance (fire), with a delay in post-disturbance accumulation. The area under the lines represents the amount of CWD. After Harmon et al. (1986) and Spies et al. (1988).

as ''CWD'' (Brown et al., 2003); here we follow Stephens and Moghaddas (2005) in using the term ''snag'' for standing dead trees and ''CWD'' for downed logs >7.62 cm diameter. A theoretical model of snag decomposition was developed by Thomas et al. (1979), using external characteristics such as the degree of retention of fine twigs and bark, and bole breakage, hypothesizing that snags would progress in a step-wise fashion from one class to the next over time (Fig. 1). Kimmey (1955) tracked Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.) and ponderosa pine, finding that decay reached the heartwood of most trees within 3 years of death. In fire-killed ponderosa pine in Washington, trees with decay increased from 35% in year 2 to 100% by year 5, with volume loss reaching 76% by the fifth year (Hadfield and Magelssen, 2000).

Ponderosa snags, especially smaller diameter trees, fall quickly. Chambers and Mast (2005) found that 41% of fire-killed trees fell by 7 years after fire in northern Arizona. Ten years after a fire in Oregon, Dahms (1949) found that slightly more than half of the dead *P. ponderosa* had fallen, rising to 78% by 12 years post-fire. Harrington (1996) reported that 75% of prescribed fire-killed ponderosa pine and Gambel oak fell within 10 years of the fire. Large snags stood longer in a study by Everett et al. (1999), where 58% of (≥23 cm) diameter at breast (dbh) ponderosa pine snags were still standing 66 years after a fire, but 50% of the snags <23 cm dbh fell or broke in the first 7–12 years.

Post-wildfire fuel loads (Fig. 2) have rarely been reported in the literature; we found only four peer-reviewed articles, none of which were in pure ponderosa pine forests. In a ponderosa–mixed conifer stand in northern Arizona 6 years after a mixed severity fire, Fulé et al. (2004) reported a mean of 55.7 Mg ha^{-1} of CWD. Five years after a severe wildfire in Madrean oak–pine forest, CWD averaged only 1.2 Mg ha^{-1} (Fulé et al., 2000). Tinker and Knight (2000) reported a mean of ∼160 Mg ha^{-1} of CWD in a lodgepole pine forest (*Pinus contorta* Dougl. ex Loud) after a severe wildfire in Wyoming. Pedlar et al. (2002) reported a mean of 342.6 m^3 ha^{-1} of CWD 1 year after a fire in a mixed conifer/deciduous forest in Ontario. We found only one estimate of post-wildfire fuel loads in ponderosa pine: CWD biomass was projected to reach 107–204 Mg ha^{-1} after the Rodeo-Chediski fire in Arizona, based on a calculation of the biomass of fire-killed standing trees that were expected to fall (USDA, 2004).

To assess the dynamics of snags and down woody debris after severe wildfire, we selected seven wildfires covering a chronosequence from 3 to 27 years post-fire in northern Arizona to ask the following questions: (1) What are the rates and characteristics of changes in snag populations following a severe wildfire? (2) How do the post-wildfire fuel complexes of standing and fallen woody debris change over time? (3) What are the implications for management?

2. Methods

2.1. Study sites

We selected sites within ponderosa pine forests that had experienced severe crownfires (Table 1). To facilitate compar-

Table 1
Fires selected for study in the Coconino National Forest, AZ

Fire	Years since fire	Size (ha)	Mean elevation (m)	Forest type
Leroux	3	450	2580	*P. ponderosa*–mixed conifer
Pumpkin	4	6375	2403	*P. ponderosa*
Pipe	4	268	2302	*P. ponderosa*
Hochderffer	8	6626	2457	*P. ponderosa*
Horseshoe	8	3495	2314	*P. ponderosa*
Bear Jaw	9	315	2538	*P. ponderosa*–mixed conifer
Radio	27	1858	2190	*P. ponderosa–Q. gambelii*

isons, fires were chosen in a relatively compact geographical area in the Coconino National Forest around Flagstaff, AZ. Seven high intensity wildfires, ranging from 3 to 27 years old, met the following criteria: (1) no salvage logging or second fire in the last 30 years; (2) burns were on non-cinder basaltic soils with slope <45%; (3) minimum area of severe burning (>50% overstory mortality) of 260 ha. There was a gap of 18 years with no fires because no areas that fit the criteria exist for that time period, due to widespread post-fire salvage management until 1996.

Elevations of the fire-burned areas ranged from approximately 2200 m to 2600 m. Roughly half of the mean annual precipitation of 54 cm is in the form of snow with the rest coming as rain during the summer monsoon season. For the months of February and July the mean daily minimum and maximum temperatures for 2004 were −10.5 °C and 6.1 °C and 5.8 °C and 28.9 °C, respectively (National Oceanic and Atmospheric Administration, Station 023160 [Flagstaff Pulliam Airport], http://cdoincdc.noaa.gov/ansum/ACS). The dominant tree species of the forest is ponderosa pine, but at the higher elevations limber pine (*Pinus flexilus* James) and Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco) occur sporadically. At the lower elevations, Rocky Mountain juniper (*Juniperus scopulorum* Sarg.), and Gambel oak (*Quercus gambellii* Nutt.) also occur sporadically.

2.2. Field sampling

We sampled 30 plots at each fire. Each plot was 90 m^2 (15 m × 6 m) in size, 0.2 km apart from each other, and 0.5 km from any roads. Six of the seven fires were sampled on a grid system with a random starting point. Plots on the Radio fire were also placed on a grid, but it was not continuous because the burn had been partially salvaged, stopping where boulders prevented the use of heavy equipment; we sampled the unsalvaged areas. We made all field measurements between May and September 2004.

We measured forest floor depth, fine woody debris (<7.62 cm diameter), and CWD with 15-m long planar transects (Brown, 1974). We inventoried fine woody debris in three size classes (0–0.64 cm, 0.65–2.54 cm, and 2.55–7.62 cm diameter) (Brown, 1974). These size classes correspond to 1-h, 10-h, and 100-h moisture timelag classes, while CWD (logs > 7.62 cm diameter) corresponds to the 1000-h timelag class (Fosberg, 1970). Sound and rotten rating determination followed Maser et al.'s (1979) five class system, where class 1 is a freshly fallen tree, class 3 is rotten on the outside but solid in the core, and class 5 is almost totally decomposed. We considered classes 1 and 2 to be "sound" wood and the remaining classes to be "rotten." We took the 1 h and 10 h fuel class measurements in the first 1.8 m of the fuel transect, the 100 h fuel class in the first 3.1 m, and the 1000 h fuel class along the entire 15 m length of the transect. We measured duff and litter depth every 5 m. To capture the complexity of the arrangement of intersecting or "jack-strawed" logs, we measured the distance from the intersection of the planar transect with each sampled log to the next log that crossed the sampled log. We also noted the diameter and length of every log that touched a tallied log. 'Touched' was defined as an intersection of the centerlines of each log.

We measured all living trees and snags on a 6 m × 15 m belt transect centered on the 15-m planar transect. We measured species and height (m) on all living trees and snags taller than breast height (1.37 m). Diameter at breast height (cm) was measured on trees taller than 3 m. Regeneration trees (<1.37 m tall) were tallied by species and height class. State of decay from Thomas et al. (1979) was also recorded for all snags: "recent" (retaining bark and fine twigs), "loose bark," "clean" (bark mostly fallen off), "broken" at some point above breast height, and "fallen" (broken below breast height).

We calculated fine and coarse woody debris biomass using equations from Brown (1974) with southwestern species-specific coefficients from Sackett (1980). Sample statistics were determined (mean and standard error) but we did not test for statistically significant differences among sample means because the seven fires were unreplicated cases of severe burning, rather than replicates from a randomly manipulated population. Instead of assessing a null hypothesis of no difference among snag densities and CWD loading, we compared data from the chronosequence graphically and as percentages to assess relative changes over time and compare them with theoretical patterns of snag and CWD dynamics.

3. Results

3.1. Snag dynamics

Recent snag density (Fig. 3) declined rapidly from the most recent fires (3–4 years post-fire) up to the middle fires (8–9 year post-fire), similar to the theoretical curve illustrated in Fig. 1. In the recent fires, the mean density of recent snags was 270.4 ha^{-1}. Recent snags made up 37% of the total snag population (Table 2). In the middle-aged fires, the mean was

Table 2
Percentage of snags in each decay class for seven wildfires, in seven wildfire study sites in the Coconino National Forest, AZ

Snag type	Leroux (3 years)	Pumpkin (4 years)	Pipe (4 years)	Hochderffer (8 years)	Horseshoe (8 years)	Bear Jaw (9 years)	Radio (27 years)
Recent	57.7	80.6	14.4	2.1	5.2	20.0	45.1
Loose bark	20.5	11.3	60.4	10.3	19.0	9.5	9.7
Clean	0	0	0	1.0	0	0	23.9
Broken	9.8	3.2	7.0	20.6	24.1	9.5	5.3
Fallen	12.1	4.8	18.2	66.0	51.7	61.0	15.9

The decay classes are from Thomas et al. (1979).

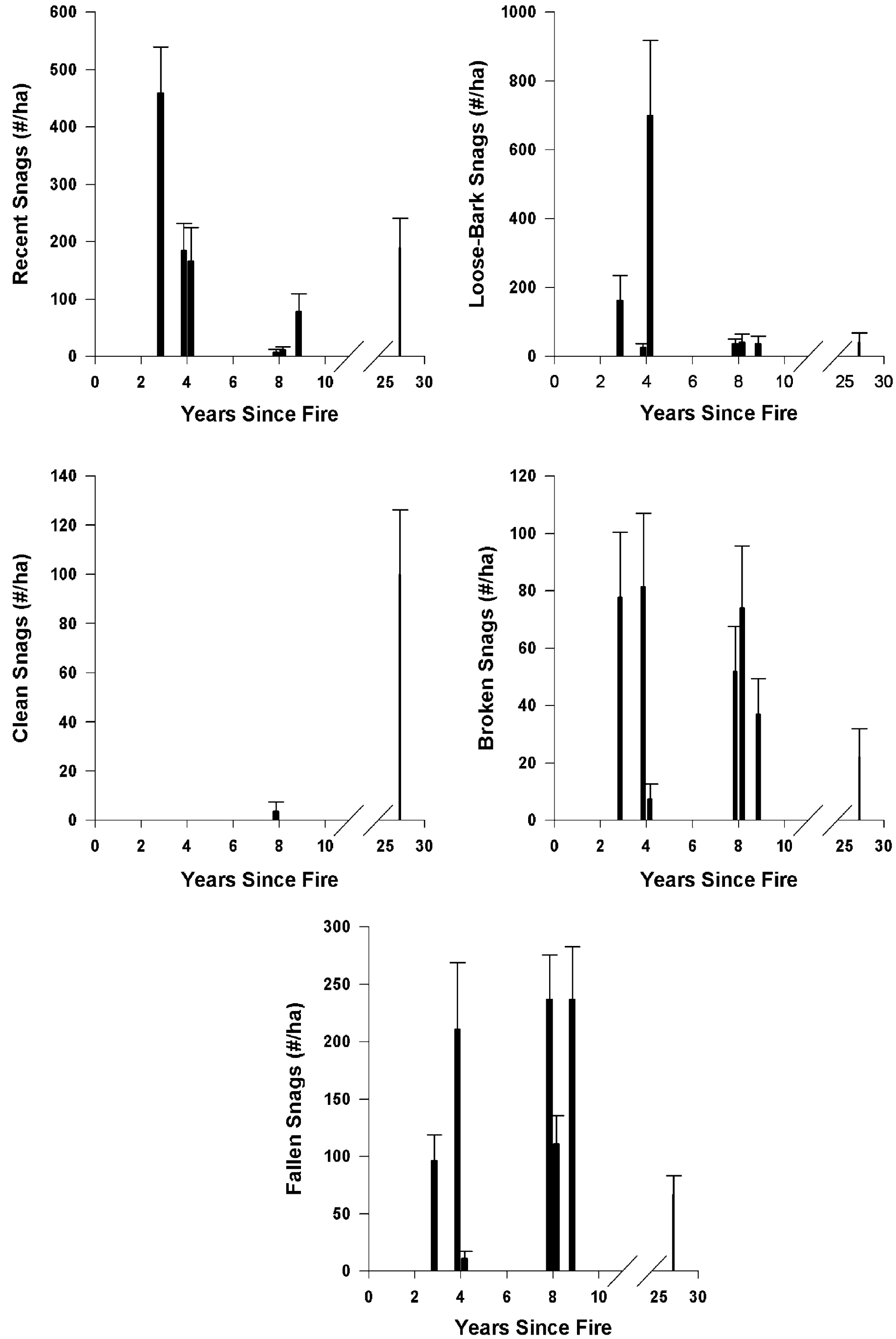

Fig. 3. Snag densities by decay class and time since fire in seven wildfire study sites in the Coconino National Forest, AZ. Error bars are standard errors. Fires are graphed in the order shown in Table 1.

32.1 snags ha^{-1} or 10% of the snag population, all ponderosa pine. The Bear Jaw fire also had Douglas-fir snags, averaging 25.9 ha^{-1}. The oldest fire, the 27-year-old Radio fire, appeared to violate the theoretical expectation by having numerous recent snags (Fig. 3), but these were small Gambel oak trees that had established post-fire and then died, averaging 188.9 ha^{-1}.

Loose bark snags were present in all fires, tending to decrease as time since fire increased (Fig. 3; Table 2). Plots in the most recent fires had a mean of 296.3 loose bark snags ha^{-1} (or 40.7% of the snag population) (all ponderosa pine), while the middle fires had a mean of 38.3 loose bark snags ha^{-1} (or 11.9% of the snag population). Considering only ponderosa pine in the middle fires, the mean dropped to 29.6 ha^{-1}. The 1995 Bear Jaw fire had also a mean of 3.7 limber pine and Douglas-fir snags ha^{-1} and a mean of 18.5 Gambel oak snags ha^{-1}. The 1977 Radio Fire had a mean of 3.7 Douglas-fir loose bark snags ha^{-1} and a mean of 37.0 Gambel oak loose bark snags ha^{-1}.

The category with the greatest contrast to the hypothetical pattern of snag decomposition was the ‘‘clean’’ category. There were no clean snags on the more recent fires and none on two of the three middle fires (Fig. 3). There was a mean of 3.7 clean aspen (*Populus tremuloides* Michx.) snags ha^{-1} on the 1996 Hochderffer fire. The 1977 Radio fire had a mean of 3.7 clean ponderosa pine snags ha^{-1} and a mean of 96.3 clean oak snags ha^{-1} (Fig. 3).

Broken snag densities decreased with the time since fire (Fig. 3), in contrast to the expected pattern, although the degree of decline was relatively less steep than for recent and loose bark snags through the 8–9-year-old fires. The more recent fires had a mean ha^{-1} of 55.6% or 7.6% of the snag population, and the middle fires had a mean ha^{-1} of 54.3% or 16.9% of the snag population and the 27-year-old fire had 22.2 broken snags ha^{-1} or 5.3% of the snag population. Almost all were ponderosa pine, but the 1995 Bear Jaw fire had 11.1 Douglas-fir broken snags ha^{-1} and the 1977 Radio fire had 7.4 Gambel oak broken snags ha^{-1}.

Fallen snags (broken below breast height) increased over time until the oldest fire, when the trend reversed (Fig. 3). The more recent fires had a mean of 106.2 fallen snags ha^{-1} (14.6% of the snag population) and the 8–9-year-old fires had a mean of 195.1 fallen snags ha^{-1} (60.8% of the snag population). Most fallen snags were ponderosa pine; only the 1995 Bear Jaw fire had another species (Douglas-fir). The 1977 Radio fire had a mean of 66.7 fallen snags ha^{-1} of ponderosa pine, Douglas-fir, and Rocky Mountain juniper (15.9% of the snag population). Despite our best effort to control for tree cutting, the density of stumps per hectare varied from a mean of 3.7 stumps ha^{-1} in the 1997 Radio and 2000 Pipe fires to 159.3 stumps ha^{-1} in the 1995 Bear Jaw fire.

3.2. Coarse woody debris

The mean total biomass of CWD per fire ranged from 3.3 Mg ha^{-1} in the 2000 Pumpkin fire to 41.3 Mg ha^{-1} in the 1977 Radio Fire. Sound CWD material was dominant until the oldest fire. The recent fires (Leroux, Pumpkin, and Pipe) averaged 7.7 Mg ha^{-1} of sound CWD (72% of the total CWD), and the middle fires averaged 34.3 Mg ha^{-1} sound CWD (96% of the total CWD), peaking 8 years after fire (Horseshoe/ Hochderffer) (Fig. 4). The rotten fuel biomass was highest in the oldest fire (35.8 Mg ha^{-1} = 86% of the total CWD), but the recent fires had on average almost two times the rotten biomass as the middle fires (3.1 Mg ha^{-1} versus 1.7 Mg ha^{-1}). Fine woody debris patterns differed by timelag class. The 100 h fuels were highest in the 8/9-year-old fires (mean = 5.8 Mg ha^{-1}) as were the 10 h fuels (mean = 1.8 Mg ha^{-1}) (Table 3). The mean and maximum diameters (cm) of logs encountered on the fuel transects showed no pattern with time since fire (Table 4).

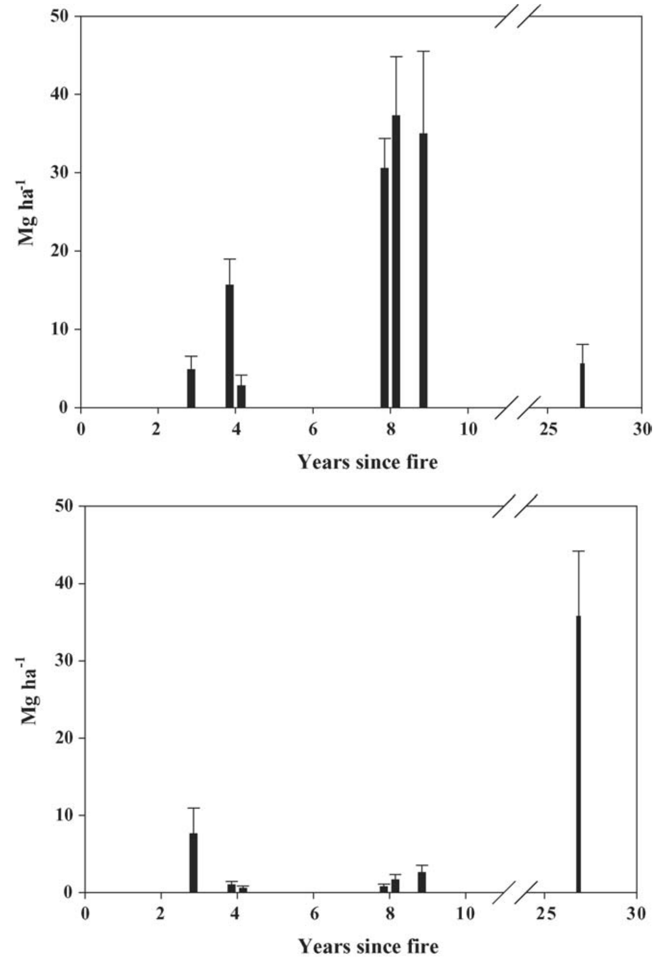

Fig. 4. Sound CWD biomass (top panel) and rotten CWD biomass (bottom panel) by time in seven wildfire study sites in the Coconino National Forest, AZ. Error bars are standard errors. Fires are graphed in the order shown in Table 1.

Characteristics of jackstrawed logs did not differ greatly among fires and showed little relationship to time since fire. The mean number of logs in a jackstraw was 2.6 in the recent fires, 2.3 in the middle fires, and 3.5 in the 27th year since fire.

Table 3
Biomass in Mg ha^{-1} of fine woody fuels (<7.62 cm diameter) on the seven wildfire study sites

	Years since fire	1 h fuels		10 h fuels		100 h fuels		Total	
		Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
Leroux	3	0.09	0.02	0.83	0.2	2.07	0.54	2.99	0.62
Pumpkin	4	0.2	0.04	1.2	0.28	1.26	0.51	2.66	0.64
Pipe	4	0.2	0.05	0.76	0.16	3.89	0.83	4.85	0.89
Hochderffer	8	0.15	0.03	2	0.49	5.1	1.59	7.25	2.06
Horseshoe	8	0.13	0.03	1.38	0.33	4.34	1.06	5.85	1.32
Bear Jaw	9	0.23	0.05	1.9	0.36	7.9	1.58	10.03	1.83
Radio	27	0.16	0.05	0.42	0.13	2.6	1.11	3.18	1.12

Fuels are listed by moisture timelag class (see text). S.E.: standard error.

Table 4
Diameters and total number of logs encountered on the fuel transects of each fire

	Years since fire	Log diameter (cm)		Logs per 15 m transect
		Mean (S.E.)	Maximum	Mean (S.E.)
Leroux	3	17.8 (1.9)	55.9	1.2 (0.3)
Pumpkin	4	13.9 (1.5)	33.0	0.5 (0.2)
Pipe	4	12.6 (0.9)	33.0	3.2 (0.6)
Hochderffer	8	14.5 (1.0)	44.5	4.6 (0.6)
Horseshoe	8	17.5 (1.3)	45.7	3.6 (0.5)
Bear Jaw	9	19.4 (1.9)	91.4	2.7 (0.5)
Radio	27	22.0 (1.7)	81.3	2.6 (0.4)

S.E.: standard error.

The mean percent of fuel transects with no logs was highest in the recent fires (47%), lowest in middle fires (12%), and 20% in the oldest fire. The mean distance to the first contact was 0.7 m in the recent fires, 1.1 m in the middle fires, and 1 m in the oldest fire.

Forest floor depth varied from site to site without a clear relationship to time since fire. There was a slight trend for deeper litter over time as the recent fires (Leroux, Pumpkin, and Pipe) averaged 0.7 cm, the middle fires averaged 1 cm, and the oldest fire (Radio) had a mean of 1.6 cm. Duff was deepest in the oldest fire (1.6 cm), but was higher in the recent fires (1.2 cm) than in 8-year-old fires (mean 0.8 cm), not necessarily showing a relationship with time (Fig. 5).

3.3. Living tree structure

Regeneration was variable, both in species and in number (Table 5). The 27-year-old Radio fire was dominated by oak regeneration and the 9-year-old Bear Jaw fire by aspen. The other five fires had only conifer regeneration or none at all (4-year-old Pipe fire).

Density of overstory trees (taller than 1.37 m) varied widely from fire to fire (Table 5), but ponderosa pine density generally declined with time since fire. The most recent fires had a mean of 355 ponderosa pine ha^{-1}, which declined to a mean of 79 ponderosa pine ha^{-1} in the 8- and 9-year-old fires, and finally reached a mean of 70.4 ponderosa pine ha^{-1} in the 27-year-old fire. Our site selection criterion of >50% mortality was based on a subjective assessment, since we did not have pre-fire density data, but only the Pumpkin fire with approximately 700 living trees ha^{-1} was an outlier, with a higher density of surviving trees than snags. The Radio fire (the oldest) had a mean of 1644 trees ha^{-1} (95% oak). The only other fire to have more than one species of tree was the Bear Jaw fire, where 64% of the established trees were ponderosa pine, 33% were aspen, and 3% were Douglas-fir. Average diameters of living conifers (Table 5) ranged from 20 cm to 25.5 cm on the six more recent fires, dropping as low as 12.8 cm on the Radio fire.

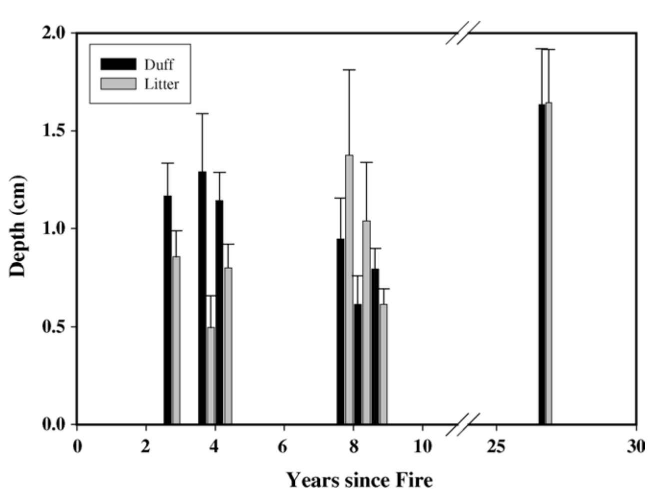

Fig. 5. Litter and duff depth after fires in seven wildfire study sites in the Coconino National Forest, AZ. Error bars are standard errors. Fires are graphed in the order shown in Table 1.

4. Discussion

The chronosequence of seven wildfires was characterized by relatively rapid snag fall and relatively light CWD loads, though the data are limited in several ways. We attempted to minimize variability in soils, slopes, and geographical region, but were constrained by the small number of unsalvaged severe wildfires available for study. The seven fires probably differed in site characteristics, pre-fire management history, and post-fire influences of factors such as pathogens, weather events, recreational use, and firewood cutting. Unfortunately no detailed pre-fire data were available. The possibly unique characteristics of any of the seven cases of severe wildfire studied here should be borne in mind when interpreting the results.

The recent fires were essentially dominated by two kinds of snags: recent or loose bark. By the eighth year after fire, most of those snags had fallen either partially or completely. The high rate of snag fall after ±5 years is consistent with what others have found (Keen, 1929; Harrington, 1996; Everett et al., 1999; Chambers and Mast, 2005). The rapid snag fall is illustrated by the increase from an average of 7.6% and 14.6% of the snag population as broken or fallen, respectively, in the recent fires to 16.9% and 60.8% of the snag population as broken or fallen in the middle fires (Table 2).

It appeared that the process of changing from vertical to horizontal fuel was not necessarily one in which most snags pass through every state possible from recent to broken to

Table 5
Mean regeneration (stems ha^{-1}) and mean overstory density (trees ha^{-1}) by group (conifers [Con.] vs. deciduous species [Dec.]) in seven wildfire study sites in the Coconino National Forest, AZ

	Years since fire													
	3 (Leroux)		4 (Pumpkin)		4 (Pipe)		8 (Hochderffer)		8 (Horseshoe)		9 (Bear Jaw)		27 (Radio)	
	Con.	Dec.	Con.	Dec.	Con.	Dec.	Con.	Dec.	Con.	Dec.	Con.	Dec.	Con.	Dec.
Regeneration (trees ha^{-1})														
Mean	14.8	0	170.4	0	0	0	3.7	0	48.1	0	1051.9	1663.0	25.9	1248.1
Standard error	11.6		61.8				3.7		44.5		282.3	484.2	12.7	405.7
Overstory (trees ha^{-1})														
Mean	370.4	0	655.6	0	40.7	0	14.8	0	40.7	0	188.9	96.3	70.4	1574.1
Standard error	86.3		79.3		27.4		14.8		31.3		58.6	85.5	33.5	316.9
dbh (cm)														
Mean	24.0		20.6		20.7		25.0		25.5		20.0	5.7	12.8	6.0
Standard error	1.0		1.0		1.4		4.3		3.4		2.0	2.5	0.7	0.2

Trees shorter than breast height (1.37 m) are regeneration. Average diameters at breast height (dbh) are shown for trees taller than 3 m.

fallen. Snags can break in any spot along the bole and at any state of bark retention or wood decay, as noted by Everett et al. (1999) and Hadfield and Magelssen (2000), rather than following the hypothetical step-wise process from one category of snag to another as presented in Fig. 1. We found no clean bark snags in early fires and no ponderosa pine clean bark snags in the middle and oldest fire. More clean snags might have been found if a number of large, old trees were burned in a fire, because old trees have substantial heartwood. The average dbh of surviving trees in the seven fires studied ranged from approximately 13 cm to 26 cm, indicating that these stands consisted of young, second-growth trees with little or no heartwood.

Coarse woody debris reached its highest level after the middle fires, when most of the snags had fallen. The 27-year-old fire had a similar total CWD loading. This did not match the model proposed by Harmon et al. (1986) because they hypothesized a post-disturbance spike in CWD that gradually declines over time (Fig. 2), while our data suggest a relatively persistent plateau in the amount of CWD, at least over 27 years. In the first 8 or 9 years since fire, CWD was sound wood with a modal score of 2 on the log decay scale (Maser et al., 1979). But in the 27 year post-fire site, the modal score had increased to 4. This change of state from sound to rotten represents an increase in fire hazard due to the greater ignitability and persistence of smoldering of rotten wood (Brown et al., 2003). No clear pattern emerged related to jackstrawing, but the oldest fire results indicate that many jackstraws still exist on the landscape even after more than one-quarter of a century.

Coniferous regeneration and the densities of trees taller than 1.37 m varied widely from fire to fire. Post-fire regeneration patterns reported at several locations across the southwest by Barton (2002) and Savage and Mast (2005) also showed differential patterns. Several decades after severe wildfires, landscapes in some cases reverted to ponderosa pine forest cover, but a substantial proportion of the fires studied by Savage and Mast (2005) remained in stable grass- or shrub-dominated ecosystems decades after burning. Barton (2002) argued that high oak regeneration and minimal pine regeneration following wildfire in Madrean pine–oak forests indicated a transformation to long-term oak dominance. There is no clear evidence from our limited sample of fires that the post-fire snag densities or CWD loadings are related to which successional pattern may be followed over time. The extent to which conifers survive as seed trees, occurrence of favorable moisture conditions for regeneration (Swetnam et al., 1999), and the presence of sprouting species (oak and aspen) may be the most important factors in guiding post-fire succession.

4.1. Management implications

The paradox of post-wildfire fuels data is that while we can measure the biomass, variability, and arrangement of snags and CWD with high precision, it is not a simple matter to draw concrete implications for future fire behavior and effects. Contemporary fire behavior models commonly used in the United States such as Behave (Andrews, 1986), Farsite (Finney, 1998), and Nexus (Scott and Reinhardt, 2001) are based on Rothermel's (1972) surface fire behavior model, which is primarily influenced by fine fuels (<7.6 cm diameter). Even though it would be possible to create custom fuel models with the data we collected, there are two problems: first, custom fuel models require extensive testing for stable results (Andrews, 1986); second, since the surface fire behavior model is primarily sensitive to fine fuels, even accurate custom fuel models would not reflect differences in CWD biomass. Fine fuels such as seeded grasses (e.g., Barclay et al., 2004) would have a bigger impact than CWD on predicted fire behavior.

Rather than modeling, we compared post-wildfire CWD data with the loadings reported in standard fire behavior models and other CWD resource guidelines. Post-fire fuels are conceptually most similar to slash fuel models, numbers 11–13 in Anderson (1982), because these fuel models are designed for situations in which fire spreads through light to heavy loads of downed woody debris. However, the total fine woody debris in the post-fire sites ranged from 2.7 Mg ha^{-1} to 10.0 Mg ha^{-1}, well below the range of 25.8–130.1 Mg ha^{-1} in the slash fuel models (Anderson, 1982). The complexity of fuel modeling means that there is not a monotonic relationship between fine fuel load and fire behavior variables such as flame length or heat/area. But the lower values at the seven wildfire sites imply that surface fire behavior at these sites would likely be substantially less intense than even that of a light logging slash fuel model.

Soil heating could serve as a criterion for evaluating future fire effects, using the heat of combustion for forest fuel (approximately 18,620 kJ/kg) (Pyne et al., 1996) and multiplying it by a range of values between 50% and 95% (as no wildland fuel burns completely). Using an average of 37.3 Mg ha^{-1} (approximate total sound + rotten CWD in 8–27-year-old fires), total heat release could range from 3.5×10^5 MJ ha^{-1} to 6.6×10^5 MJ ha^{-1}. Based on assessments of heat transfer under slash piles, approximately 10–15% (DeBano et al., 1998) of this energy would be directed downward into the soil. However, the ecological effects in terms of subsurface temperatures would vary depending on the capability of the soil to transmit heat, soil moisture content, location of heat-susceptible propagules, etc.

Fire suppression complexity is a management-oriented criterion, since heavy downed fuels could slow fireline construction. Depending upon their experience and physical condition, a 20-person crew can cut, backfire, and hold a fireline at rate of 181–301 m/h in light slash, model 11 (USDA, 1998). A bulldozer, however, can cut fireline from 140 m/h to 1037 m/h, depending upon the terrain and operator experience (USDA, 1998). In the post-fire sites we studied, the relatively low CWD loads as compared to fuel model 11 indicate relatively less resistance to control and mechanical equipment would likely be used in the event of a large, spreading fire in these sites because they are relatively flat and accessible.

Historical precedents where reburns have occurred could shed light on what might happen to the fires examined in this study. However, while there are anecdotal accounts of severe reburns in post-wildfire fuels (USDA, 2004), the only well documented occurrences of wildfire severity being affected by the fuels that remained after previous fire are from forests that are more productive than those of the southwest (Odion et al., 2004). Historical reburn effects in southwestern forests merit additional study.

Recommendations for optimal ranges of CWD in ponderosa pine forests have been developed by Graham et al. (1994) and Brown et al. (2003). To maintain long-term forest productivity, Graham et al. (1994) recommended that between 15 Mg ha^{-1} and 29.5 Mg ha^{-1} of CWD be maintained in ponderosa pine/Arizona fescue stands and that between 11 Mg ha^{-1} and 23 Mg ha^{-1} be maintained in ponderosa pine/Gambel oak stands in Arizona. Brown et al. (2003) assessed a variety of factors, including CWD benefits for wildlife habitat and productivity, balanced against potentially negative effects of soil heating and fire hazard. They set the optimal level for balancing these concerns between 11.2 Mg ha^{-1} and 44.8 Mg ha^{-1} of CWD in warm, dry forest (Brown et al., 2003). As illustrated in Fig. 6, we found a wide range of CWD loading, from 3.3 Mg ha^{-1} to 41.3 Mg ha^{-1}, in the seven fires we studied. The lowest values, in the youngest fires, are below both the minima of both of the recommendations. However, as material accumulated by the eighth or ninth year post-fire, CWD levels rose to within the optimal ranges but never exceeded the maximum value proposed by Brown et al. (2003) (Fig. 6).

Recognizing the limited sample size and constrained inference of this study, and the fact that management actions must be site- and goal-specific, we consider three options for management when dealing with post-wildfire landscapes forests similar to those we studied: salvage logging, prescribed burning, and passive management. Salvage logging has been suggested to be beneficial in three ways: removal of CWD could lessen wildfire effects (USDA, 2004), insect population growth (Simon et al., 1994), and soil erosion (Poff, 1989). But salvage logging may be detrimental to soil (Klock, 1975; Potts et al., 1985; Maser, 1996; Beschta et al., 2004), wildlife (Saab and Dudley, 1998), and vegetation (Roy, 1956; Sexton, 1998; Martinez-Sánchez et al., 1999). Since the fuel loads found in

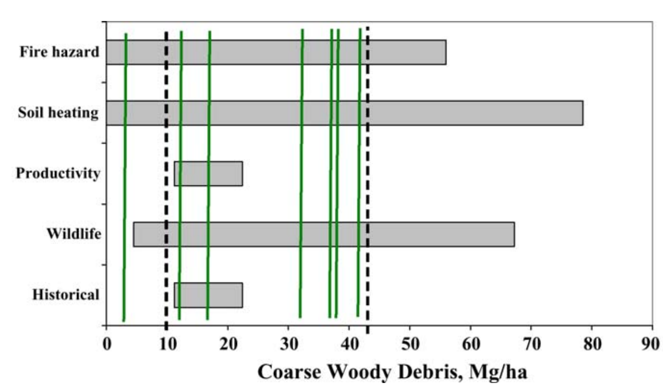

Fig. 6. CWD loadings from the seven wildfire study sites in the Coconino National Forest, AZ (solid vertical lines), compared with the "optimum ranges" of CWD for warm dry forest types from Brown et al. (2003). The optimal range that seemed suitable for multiple resource considerations (dashed vertical lines) was delineated by Brown et al. (2003). Values presented by Brown et al. (2003) were for downed CWD except in the wildlife category, which included snags.

our study fell within the ranges that are recommended as being both beneficial to the ecology of the site and not a wildfire threat, salvage logging based on future fire hazard does not seem appropriate for these sites.

In situations where post-fire fuels may be seen as excessive, prescribed burning is an alternative way to lower fuel load while retaining CWD benefits (Brown et al., 2003). By controlling burning conditions, managers can manipulate to some extent the heat produced by burning CWD (controlling damage to plants and soils) and how much of it is consumed (preserving the animal habitat and erosion control benefits), while lowering the near-term fire hazard by reducing the fine fuel. Negative impacts include the possibility of fires escaping control, smoke, fires producing more heat than anticipated leading to soil damage, and excessive consumption of CWD.

Passive management may be appropriate where managers believe that the fuel complex of snags and CWD does not exceed thresholds such as those suggested by Brown et al. (2003) and where salvage actions are not indicated for other reasons (public safety, insect infestation, and economic issues). While inferences based on the fire chronosequence studied here are limited, there is no evidence that continued passive management of these sites would have negative effects. Under any of these management options, it is important to recognize that regeneration of post-crownfire landscapes in the southwest is uncertain, with some post-fire landscapes transitioning to shrubfields or grasslands for at least several decades (Savage and Mast, 2005). Active revegetation with native plants may be productive.

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