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Ten years after wildfires: How does varying tree mortality impact fire hazard and forest resiliency?

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ABSTRACT

Severe wildfires across the western US have lead to concerns about heavy surface fuel loading and the potential for high-intensity reburning. Ponderosa pine (*Pinus ponderosa*) forests, often overly dense from a century of fire suppression, are increasingly susceptible to large and severe wildfires especially given warmer and drier climate projections for the future. However, the majority of research on fuel dynamics after wildfires has focused on high-severity burned areas in more productive forest types. We sampled fuel loadings in 2009 and 2010 across a range of tree mortality on two high-severity wildfires that occurred in 2000: the Pumpkin Fire in Arizona, and the Jasper Fire in South Dakota. We established 60 plots per fire, 10 in each of five post-fire mortality classes: 0–20%, 21–40%, 41–60%, 61–80%, and 81–100% mortality, based on percentage of trees killed, plus unburned control areas. We measured height, diameter, status (alive or dead) and crown base height of each tree, plus fuel loading by size class, litter and duff depth, and herbaceous biomass. Ten years after wildfire, low mortality (0–40%) plots resembled unburned plots in almost every fuels attribute. Basal area in low-mortality plots exceeded reconstructed historical ranges and fire hazard reduction targets by up to 130%. However, coarse woody debris (CWD; woody material >7.62 cm) loadings fell below a recommended "optimum" range and herbaceous fuels were sparse. Mid-mortality (41–80%) plots were characterized by more open stands and increased surface fuel loadings, basal area was close to target ranges and CWD loadings were within the recommended range. High mortality (81–100%) plots had few trees but CWD loadings exceeded recommended levels by up to 28%, and herbaceous fuels were adequate to carry a surface fire. These findings suggest that post-fire management should be targeted to the level of mortality. Low mortality and unburned areas should be targeted for reducing stand densities and promoting understory growth, to minimize crown fire hazard and increase site potential. Burned areas with >80% tree mortality have the lowest crown fire hazard, but may benefit from fuel reduction efforts such as low intensity prescribed burning to reduce both fine fuels and some of the CWD. Stand structures and surface fuel loads in mid-mortality plots most resembled historical targets and met numerous restoration objectives for ponderosa pine-dominated forests, especially given predicted climate changes. These areas can be maintained through low-intensity prescribed burning to prevent them from becoming overly dense and thus enhance their longer-term resiliency to future disturbances.

1. Introduction

Increasing size and severity of wildfires in recent decades (Westerling et al., 2006; Littell et al., 2009) compounded by predictions for warmer and drier climates (Diffenbaugh et al., 2008), has led to increased concerns about the impact of wildfires on future fire hazard, wildlife habitat and forest resiliency. Most previous research has focused on high-severity burned areas in productive forest types. In mixed conifer forests severe fires may lead to increased surface fuel loading and thus high-intensity reburning in the future, especially in managed forests (Thompson et al., 2007; Odion et al., 2004). However, the few studies following wildfires in drier forest types such as ponderosa pine (*Pinus ponderosa*) suggest that fuel loadings in severely burned ponderosa pine types may not present a severe fire hazard (Passovoy and Fulé, 2006; Keyser et al., 2009). Further, even though most wildfires result in a mosaic of burn severities (Van Wagner, 1983), few studies have documented changes in stand structure and fuel complexes through time across the range of fire severity. Low- and moderate-severity areas of a wildfire pose different concerns relative to forest resiliency and fire hazards, especially in overly dense, dry conifer types that evolved under a surface fire regime. Some

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post-wildfire changes in forest structures and fuels may be in line with management goals such as thinning overly dense forests and thus promoting understory fine fuels necessary for carrying surface fires.

Stand structure, including both live and dead trees, as well as surface fuels are critical components of fire hazard and forest resiliency. Resilient forests are those that not only accommodate gradual changes related to climate but also return toward a prior condition after disturbance either naturally or with management assistance (Millar et al., 2007). Overstory stand characteristics such as canopy fuel loading and continuity influence the potential for active crown fire spread, and canopy base height (CBH) impacts surface fires transitioning into tree canopies (Gaines et al., 1958; Agee and Skinner, 2005). Whereas high intensity crown fires greatly reduce canopy fuels, often low intensity surface fires do not reduce basal areas (BA) or tree density enough to mitigate fire crowning potential nor reduce stand densities to levels that more closely align with historical ranges (Fulé et al., 2002; Pollet and Omi, 2002; Lentile et al., 2006; Battaglia et al., 2008). Snags (standing dead trees) play important ecosystem functions by providing wildlife habitat and a future source of dead woody material important for stand development; thus maintaining some large snags even in post-wildfire environments is often a management consideration (Balda, 1975; Bull et al., 1997; Ganey, 1999). Productive ecosystems also require the maintenance of both coarse and fine surface fuel loadings. Whereas some coarse woody debris (CWD; woody material >7.62 cm) is needed to maintain wildlife habitat and soil productivity, high CWD can present excessive fire hazard (Brown et al., 2003) and these concerns often lead to post-fire logging (Donato et al., 2006; McIver and Ottmar, 2007; Thompson et al., 2007; Keyser et al., 2009). In addition to CWD, fine fuels are essential for forest productivity and to allow for the spread of surface fires (Zimmerman, 2003).

Few studies have quantified future fire hazards of mixed-severity wildfires, nor the degree to which wildfires influence forest resiliency to future disturbances, especially in ponderosa pine dominated forests. Keyser et al. (2008) quantified stand structure, woody debris, and tree regeneration in severely burned areas 5 years after the Jasper Fire in South Dakota and Passovoy and Fulé (2006) examined surface fuel loading in a chronosequence of severe fires in the Southwest. We expanded upon their results by stratifying fuel complexes, including stand characteristics, CWD, FWD, and herbaceous fuels, by varying levels of tree mortality 10 years after two 2000 wildfires in ponderosa-pine dominated forests in the Southwest and Black Hills. Our objectives were to quantify stand structure and surface fuels along a gradient of tree mortality from unburned to >80% mortality, to determine which areas exceeded various stand and surface fuel targets, and to infer implications for fire behavior, wildlife habitat, and site productivity of the varying fuel complexes. We hypothesized that: (1) as mortality increases, basal area and stand density will decrease; whereas canopy base height will increase until mortality approaches 100% where canopy base height will be non-existent; (2) as mortality increases, surface fuels will increase in every fuel class; (3) combined effects of varying fuel complexes among mortality levels will lead to different potential fire hazards, and thus different levels of resiliency to absorbing future disturbances.

2. Methods

2.1. Study Sites and plot design

In 2001, we established permanent transects that encompassed a range of burn severities, defined by tree mortality, on four 2000 wildfires in the Intermountain West (Sieg et al., 2006). We selected two of the four original sites for re-measurement: the Pumpkin Fire on the Kaibab National Forest in northern Arizona and Jasper Fire on the Black Hills National Forest in western South Dakota. The Pumpkin Fire burned 5972 ha between 25 May and 15 July 2000. Fire severity was heterogeneous, with the majority of the burned area characterized as low-severity based on canopy and soil damage (USDA Forest Service, 2000a). Elevation ranged from 2256 to 3048 m with an average precipitation of 57.6 cm, of which half falls in July and August, and half in winter (Fig. 1). The precipitation for the 2 years of sampling was 42.2 and 68.1 cm in 2009 and 2010, respectively. The Jasper Fire burned 33,795 ha between 24 August and 9 September 2000. Elevation of the Jasper Fire ranged from 1524 to 2134 m and the mean precipitation is 54.3 cm, the majority of which is received in April, May and June (Fig. 1). Of the entire fire area approximately equal portions of burned areas were classified as low-severity, moderate-severity and high-severity (USDA Forest service, 2000b). In 2009 the precipitation was 51.2 cm, compared with 56.3 cm in 2010. Before the fires, both sites were dominated by relatively homogenous stands of ponderosa pine and the wildfires exhibited similar mosaics of fire behavior, severity, and tree mortality (Sieg et al., 2006).

We sampled stand structure and woody surface fuels in 2009 and herbaceous fine fuels in 2009 and 2010 on the Pumpkin and Jasper Fires, treating each fire as an independent case study. Originally there were twelve 10- $\times$ 200-m transects within each fire, across varying levels of tree mortality, plus 12 transects in unburned areas nearby. Due to management practices such as thinning and prescribed burning there are only eight remaining untreated burned transects for use in this study on the Pumpkin Fire. We used the original eight untreated transects on the Pumpkin Fire and the 12 transects on the Jasper Fire, to capture a full range of tree mortality within each fire. We used these transects to determine mortality ranges for potential plot locations within each fire. In 2009, we sampled a total of 60 plots on each fire, across all transects. We randomly established 10, 0.04-ha plots in each of five mortality classes (0–20%, 21–40%,...,81–100%), plus 10 unburned plots within the unburned transects. Percent mortality was based on percent tree mortality as of 2009 and we excluded plots with high tree mortality due to factors other than fire. Each plot was permanently identified using washers and stakes at the center and each corner in order to re-measure herbaceous biomass over 2 years.

2.2. Field measurements: stand structure

In each plot, we tallied all trees as alive or dead, including broken off stumps resulting from fire mortality to determine percent mortality. All trees >5.1 cm diameter at breast height (DBH) were marked with numbered metal tags in 2001, thus we used this as the size minimum for our study. Pre-wildfire tree density was calculated by adding fire-killed stumps and snags together with live trees. We measured DBH, crown base height, and tree height on all living trees >5.1 cm DBH to quantify postfire stand structure. Crown base height was measured as the height of the lowest live branch for each tree, using a hypsometer. For standing dead trees >5.1 cm, we measured height and DBH. In calculations of canopy base height (CBH) we used the lowest 20th percentile of crown base height within each plot following the example of Fulé et al. (2001, 2002). Regeneration (trees <5.1 cm DBH) was tallied within the entire plot. Basal area was calculated by using the quadratic mean diameter equation.

2.3. Field measurements: surface fuels

In each 0.04-ha plot, we measured coarse woody debris (CWD), fine woody debris (FWD), litter/duff depth and total herbaceous bio-

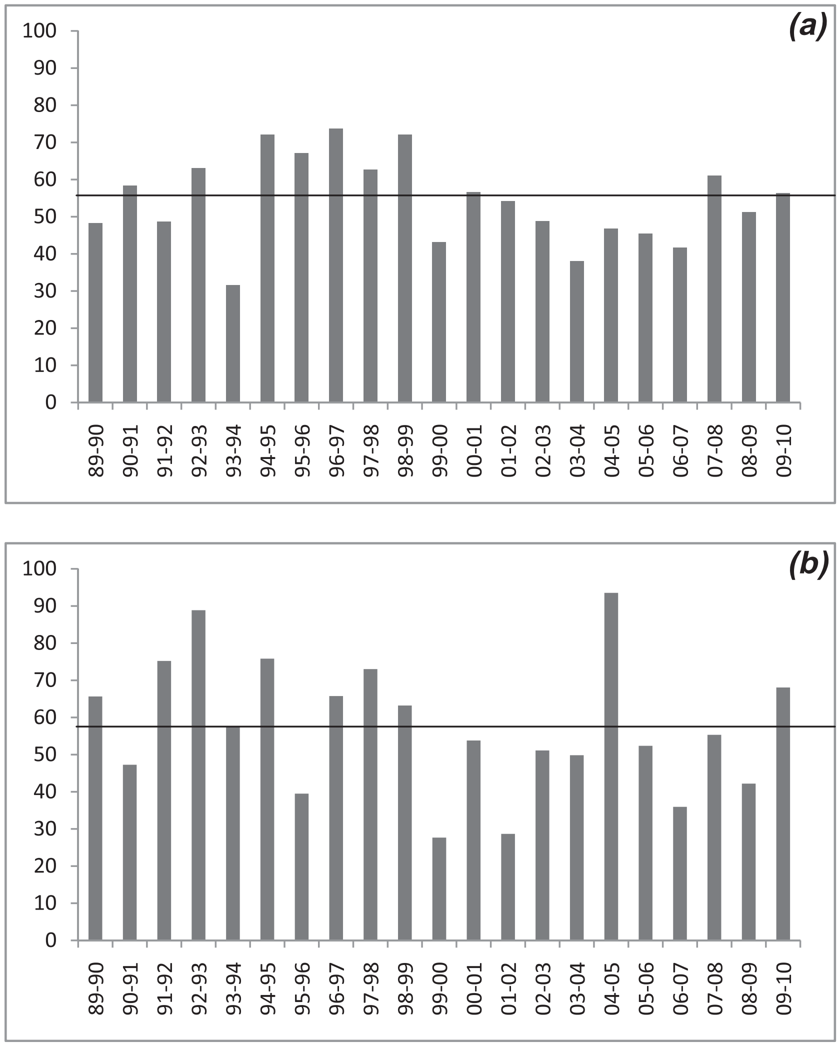

Fig. 1. Annual precipitation annual water year (October to September) precipitation between 1990–2010 and 20-year mean (black line), encompassing sampling years (2009 and 2010) for (a) the Jasper Fire based on the Hill City weather station (High Plains Regional Climate Center, 2006. Available online at www.hprcc.unl.edu (accessed 07.02.11.)) and (b) the Pumpkin Fire based on the Flagstaff 4SW weather station; for the two missing monthly values we used Flagstaff WSO AP weather station (Western Regional Climate Center, 2010. Available online at www.wrcc.dri.edu (accessed 07.02.11.)).

mass. Downed woody debris and litter/duff depths were measured along two 15-m planar transects within each plot, using methods established in Brown (1974). FWD was classified by size classes (0–0.64, 0.65–2.54, 2.55–7.62 cm), which correspond to 1-, 10- and 100-h moisture time-lag classes (Fosberg, 1970). CWD, or 1000-h fuels, were inventoried as logs $\geqslant$ 7.62 cm and categorized by sound and rotten classes (Fosberg, 1970). Fuel loadings for individual size classes were totaled to estimate total surface woody fuel loading, using equations from Brown (1974) with Southwestern species coefficients (Sackett, 1980).

We quantified herbaceous biomass in 2 years (2009 and 2010) in a subset of eight randomly-selected plots in each mortality class and in unburned plots, for a total of 48 plots per fire. We clipped non-woody plants, both dead and alive at ground level, in a total of eight, 0.25-m^2 quadrats randomly located along each of the two planar transects in each plot. We changed random locations of quadrats each year to avoid previously clipped plots. Biomass samples were dried for 48 h at 70 °C and weighed (Moore et al., 2006). We timed clipping to coincide with the historical season of high fire activity and the time in which the fires took place: June in Arizona (Brown et al., 2001; Fulé et al., 2003) and August in South Dakota (Brown and Sieg, 1996, 1999).

2.4. Data analysis

Data from each fire were analyzed as separate case studies. In all analyses plots were the experimental units and in every test the sample size was 60, except herbaceous biomass which was measured in 8 plots per mortality class, for a total of 48 plots. We first tested for differences in prefire tree density among mortality classes. We then tested for differences among mortality classes in stand structural attributes (CBH, basal area, tree density, snags, and regeneration) and surface fuels (CWD, FWD, herbaceous biomass, litter, and duff) for each fire. We used Shapiro–Wilks tests to test the assumption of normal distribution and Levene tests to test for equal variances among residuals. For residuals that at least visually met the assumptions, through q–q plots and histograms, we used analysis of variance (ANOVA) to examine response variables among mortality classes. Data for only one variable, number of seedlings (regeneration), were highly non-normal and transformations were not adequate to address non-normality, so we compared means with nonparametric Kruskal–Wallis tests. For these tests, we had six mortality classes for a resulting 5 degrees of freedom used in the model. We used repeated measures ANOVA to analyze herbaceous biomass data. Our model tested the main effects of mortality class and year, plus year-by-mortality class interaction. We used JMP (SAS Institute Inc., 2007) for these analyses, with $\alpha = 0.05$. To assess significant differences among mortality classes, when significance was observed in the overall test we used a Tukey's HSD test.

We used multivariate analyses to examine differences in the combined fuel complexes among mortality classes. We used Multi-response Permutation Procedures (MRPP; Mielke and Berry, 2001) in an Excel macro developed by the Rocky Mountain Research Station Statistician. We used 2-year herbaceous biomass averages, FWD, CWD, stand density, and basal area as the response variables and grouped by mortality class with $n = 8$ for each group. We had a total sample size of 48 to incorporate plots that had herbaceous sampling. CBH was omitted due to the lack of data in high mortality areas, which the model could not accept. Litter and duff were also omitted in the analysis due to the lack of variability among mortality classes. Before analysis, data were standardized by dividing each observation by the mean for that variable. When overall significant differences were detected we used Peritz Closure (Petrondas and Ruben, 1983) multiple pairwise comparisons to separate mortality class multivariate means.

We established thresholds for several variables based on previous studies and fuel models. Thresholds for maximum basal area and tree density were based on historical norms in both regions, as well as fuels reduction targets outlined by studies in these areas. Although highly variable, evidence suggests that historical tree densities ranged between 140 and 250 trees ha^{-1} and basal area ranged between 9.2 and 18.4 $m^2\ ha^{-1}$ for ponderosa pine forests in the Black Hills and the Southwest (Fulé et al., 1997, 2002; Brown and Cook, 2006). The threshold for CBH was estimated using NEXUS (Scott, 1999). We used the standard fire behavior fuel model 10 and 90th percentile fire weather extreme for both regions. We used a wind reduction factor of 0.2, and we held crown bulk density (CBD) constant as a mean observed in both sites determined using Cruz et al. (2003) equations, while varying CBH to determine the predicted transition from passive crown fire to surface fire. For snags, many studies and forest plans recommend retaining a minimum of three to five >40 cm DBH snags ha^{-1}, but snags this large are uncommon on our study sites (Kaibab National Forest Plan, 1986; Black Hills National Forest Plan of Land and Resource Management,1997; Ganey, 1999; Chambers and Mast, 2005). Some national forest management plans also protect snags >30 cm DBH in standards for wood cutters (Kaibab National Forest Plan, draft), thus we used this target DBH for our lower threshold. For CWD, we used Brown et al.'s (2003) "optimum" range that meets the needs for wildlife and understory productivity, without presenting excessive fire hazard and soil heating potential. This "optimum" range was defined as 11.2–44.8 Mg ha^{-1} for dry forests, including ponderosa pine. FWD and herbaceous biomass minimum loadings necessary to maintain an active surface fire were based on standard fire behavior fuel models 2, 9 and 10 (Rothermel, 1972).

3. Results

3.1. Stand structure

Our first hypothesis was supported by our findings for tree density and basal area, but not for CBH. Pre-wildfire, tree density on the Jasper Fire ranged from 415 to 598 trees ha^{-1}, and did not differ significantly among mortality classes or unburned plots ($P = 0.1474$, $F_{5,54} = 1.7129$). Pre-wildfire tree density on the Pumpkin Fire also did not differ among mortality classes ($P = 0.7331$, $F_{5,54} = 0.5559$), ranging from 489 to 697 trees ha^{-1}. In contrast, post-wildfire, both tree density (Fig. 2a and b) and basal area (Fig. 2c and d) varied significantly among mortality classes on both wildfires. Basal area and tree density were lower at higher mortality levels. On both fires unburned plots and burned plots with up to 40% mortality remained above stand structural targets, in terms of both basal area and tree density.

On the Jasper Fire, CBH on unburned plots averaged 2.35 m compared to the burned plots average of 6.27 m (Fig. 2e). CBH on the Jasper Fire was significantly lower on unburned plots than on all burned plots and higher on the highest mortality plots than other burned plots, but did not differ among midrange mortality classes with a general increase in CBH with increasing mortality. On the Pumpkin Fire, CBH was significantly lower on unburned plots than burned plots except those in the 0–20% mortality class (Fig. 2f). The majority of plots in high mortality areas did not have any live trees on either fire, thus the average CBH for high mortality plots was based on only a few plots, with only one or two live trees on each. We found that with all other variables remaining constant, the transition from a surface to passive crown fire on the Jasper Fire was predicted with a CBH of <2.5 m, compared to <3.5 m on the Pumpkin Fire. On both fires all mortality classes including unburned plots had average CBH below these thresholds.

Density of snags of all sizes varied significantly among mortality classes on the Jasper ($P < 0.0001$, $F_{5,54} = 7.81$) and the Pumpkin ($P = 0.0322$, $F_{5,54} = 2.66$) Fires. On the Jasper Fire there were the fewest snags on unburned and low mortality plots, with the highest mortality plots having the most snags. On the Pumpkin Fire only unburned and 40–60% mortality plots varied significantly, all other plots had relatively the same number of snags. Only two snags on all plots met or exceeded a minimum DBH of 40 cm, but the average number of snags >30 cm ranged from 0 to 20 snags ha^{-1} across all mortality classes on the Jasper Fire and 2.5–15 snags ha^{-1} on the Pumpkin Fire (Fig. 2g and h). Thus, target snag densities for snags >30 cm were met on plots with >40% mortality on the Jasper Fire and all but 0–20% on the Pumpkin Fire.

On the Jasper fire, only 13 out of 50 burned plots (26%) had any regeneration, 10 of which had <124 seedlings ha^{-1}. The mean per mortality class ranged from 0 to 173 seedlings ha^{-1}, with the most regeneration in the 20–40% mortality class ($\chi^2 = 15.01$, $P = 0.0104$). Unburned plots of the Jasper Fire had substantial regeneration, with a mean of 420 seedlings ha^{-1}. On the Pumpkin Fire regeneration was low on both burned and unburned plots. Ten out of the 60 burned and unburned plots had regeneration and only five plots exceeded 70 seedlings ha^{-1}. The mean per mortality class ranged from 0 to 80 seedlings ha^{-1}, with 40–60% mortality class having the most regeneration ($\chi^2 = 4.21$, $P = 0.5193$). Significance observed among mortality classes was skewed by the high number of zero values, and little variance.

3.2. Surface fuels

Our hypothesis that surface fuels would increase with increasing mortality levels was supported for some fuel classes, including CWD, FWD and herbaceous biomass, but not for litter or duff depth. Litter depth did not differ significantly on either the Jasper Fire ($F_{5,54} = 0.31$, $P = 0.9033$) or the Pumpkin Fire ($F_{5,54} = 0.39$, $P = 0.8515$) among mortality levels including unburned plots. Duff depth was significantly lower on burned sites compared to unburned sites on both the Jasper Fire ($F_{5,54} = 6.31$, $P < 0.0001$) and the Pumpkin Fire ($F_{5,54} = 6.65$, $P < 0.0001$), but did not vary among mortality classes. Average duff depth on the Jasper Fire varied from 1.42 cm on unburned plots to 0.6 cm on burned plots; whereas on the Pumpkin Fire it varied from 1.15 to 0.24 cm.

Woody fuel loadings, encompassing both CWD and FWD, generally increased with mortality. For subsequent analyses and discussion we used total CWD (both rotten and sound), due to small amounts of rotten CWD, which did not differ across mortality levels on either fire ($F_{5,54} = 1.997$, $P = 0.0938$; $F_{5,54} = 1.799$, $P = 0.1286$).

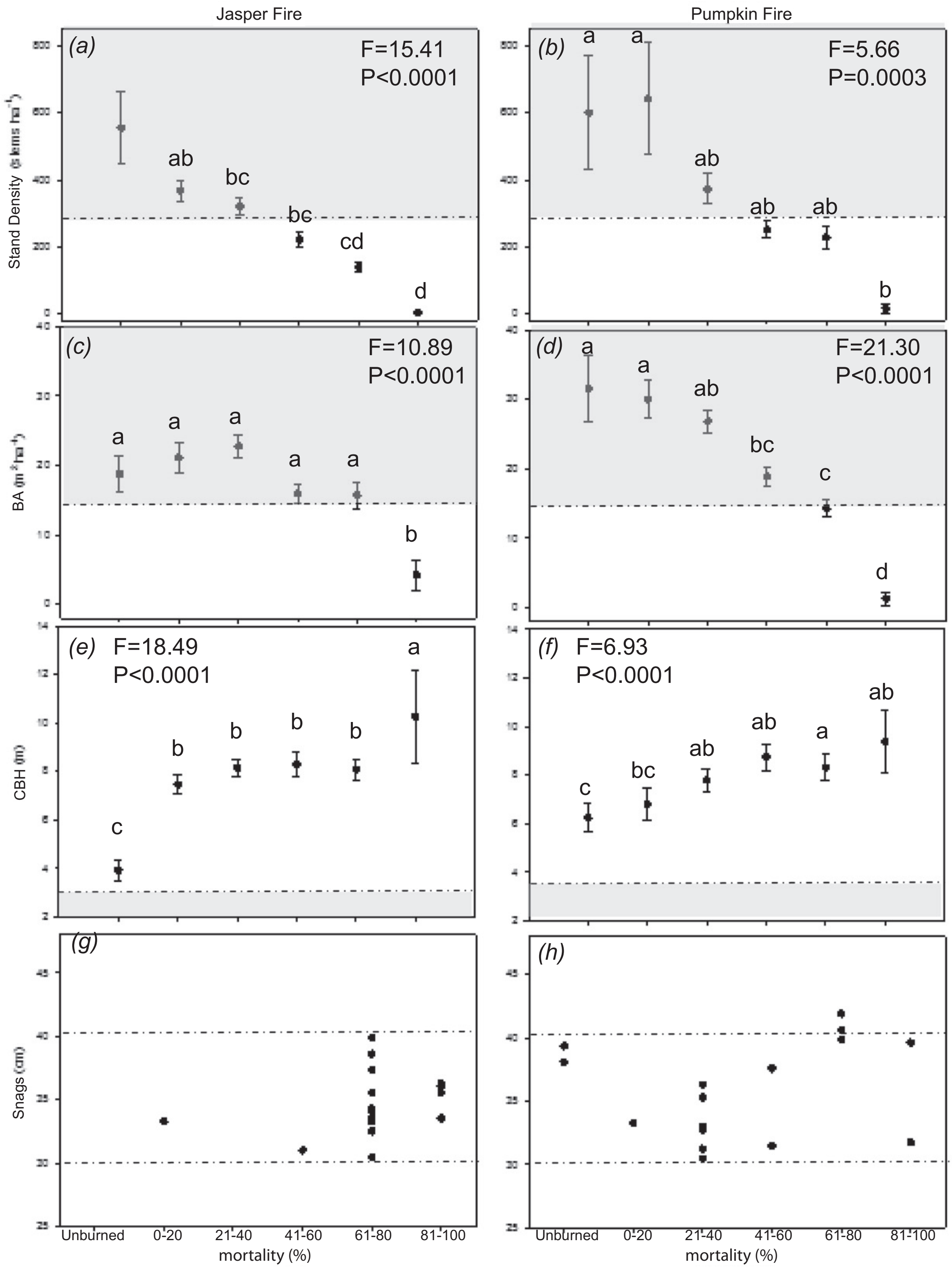

Fig. 2. Stand structure characteristics (mean ± SE) by mortality class with a total sample size of $n = 60$ plots on the Jasper Fire (left panel) and Pumpkin Fire (right panel). (a) and (b) average stand density and (c) and (d) average basal area. The gray box and dotted line indicate maximum targets and historical ranges based on Fiedler (1996), Fulé et al. (2002), and Brown and Cook (2006). (e) and (f) CBH. The gray box indicates the predicted point at which a surface fire becomes a crown fire (Scott, 1999). (g) and (h) points indicate individual tree DBHs not plot level means, of snags >30 cm DBH (minimum DBH based on draft Kaibab National Forest Plan for woodcutters), and >40 cm DBH (minimum DBH based on Chambers and Mast, 2005). Letters above means (except in g and h) indicate groupings from Tukey's HSD tests.

Rotten CWD on the Jasper Fire ranged from 0.10 to 6.83 Mg ha^{-1} and on the Pumpkin Fire quantities ranged from 0 to 3.81 Mg ha^{-1} and constituted between 0% and 7% of the total CWD on both fires. There was the least amount of total CWD on unburned and low mortality plots, with the highest fuel loads at the high mortality levels. The differences were significant for both fires (Fig. 3a and b). There was no significant difference between unburned and mortality classes up to 40%; above this level CWD levels were higher than those on unburned plots. Between 40% and 80% mortality, CWD loadings were within Brown et al.'s (2003) "optimum" range. However, plots with >80% mortality had CWD loadings above this threshold, and unburned plots and most burned plots with <40% mortality had CWD loadings below the "optimum" range (Fig. 3a and b).

FWD trends varied between fires. On the Jasper fire, FWD generally increased with increasing mortality, similar to the trend in CWD (Fig. 3c). The lowest quantities occurred in unburned areas and FWD loadings on mortality classes up to 40% were similar to those on unburned areas. The highest FWD loadings occurred in the 60–80% and 80–100% mortality classes. Plots with mortality levels >20% on the Jasper Fire had adequate quantities of FWD (6.50–6.72 Mg ha^{-1}) to maintain a surface fire (Rothermel, 1972). On the Pumpkin Fire, FWD was highly variable among mortality classes, and did not show a clear pattern of increasing with increasing mortality level (Fig. 3d). Additionally, FWD loadings on the Pumpkin Fire only exceeded the minimum threshold at >40% mortality.

Herbaceous biomass also followed the same general trend of increasing with increasing mortality. The full model was significant on both the Jasper (F = 35.06, P < 0.0001) and Pumpkin (F = 44.15, P < 0.0001) Fires. Plots with higher mortality had more herbaceous biomass in both years of sampling. Biomass significantly differed among mortality classes on both fires (P < 0.0001). Herbaceous biomass also differed significantly between years on the Jasper Fire (P = 0.0403) and on the Pumpkin Fire (P < 0.0001), but the same general trend was observed in both years. However, the year-by-mortality class interactions were significant on both fires (Jasper P = 0.0129 and Pumpkin P < 0.0001). In 2010 there was less herbaceous biomass on both fires compared to 2009. On both fires, the

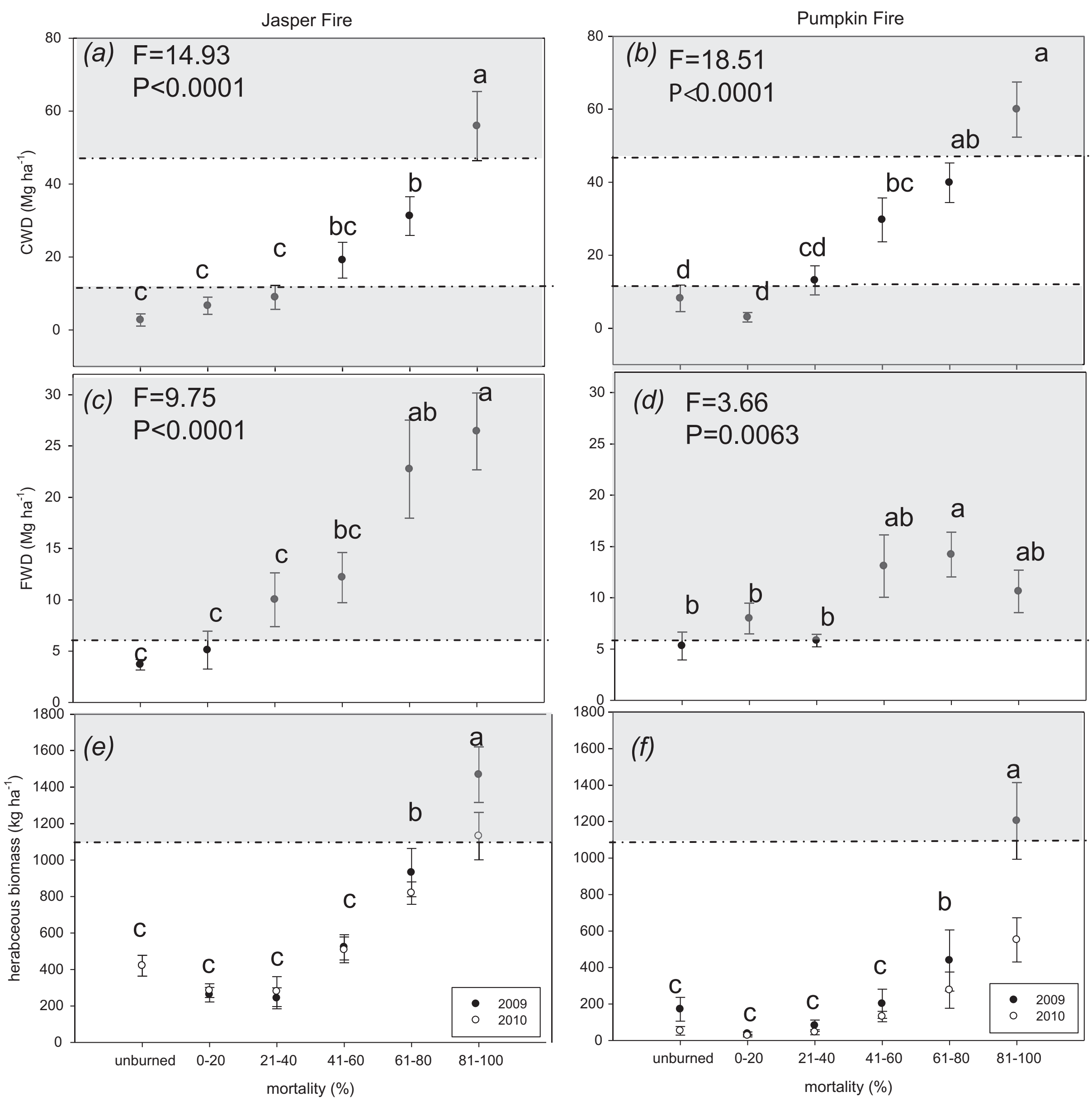

Fig. 3. Surface fuels (mean ± SE) by mortality class on the Jasper Fire (left panels) and Pumpkin Fire (right panels) (a) and (b) CWD with a total sample size of n = 60, with the "optimum" range of CWD for dry coniferous forests (Brown et al., 2003) between the dotted lines; (c) and (d) average FWD with a total sample size of n = 60, and (e) and (f) 2009 (black dots) and 2010 (clear dots) herbaceous biomass with a total sample size of n = 48, with the thresholds for both FWD and herbaceous biomass based on the minimum level needed to sustain surface fire spread (Rothermel, 1972). Letters above means indicate groupings from Tukey's HSD tests.

least amount of biomass was recorded in the 0–20% mortality class, while the highest was recorded in the 80–100% mortality class. There was not a significant increase above unburned biomass levels until mortality levels surpassed 60% in 2010 and 80% in 2009 for both fires (Fig. 3e and f). In 2009, both fires had adequate quantities of herbaceous biomass (800–1100 kg ha^{-1}) to maintain an active surface fire (Rothermel, 1972) in plots with mortality levels >80%, as did plots in the 60–80% mortality class on the Jasper Fire. However, in 2010 only plots with >80% mortality on the Jasper Fire had sufficient quantities of herbaceous biomass to maintain a surface fire.

3.3. Stand structure and surface fuel interactions

MRPP analyses indicated significant differences in fuel complexes among mortality classes ($P < 0.0001$) on both fires. Based on pairwise comparisons, fuel complexes on unburned areas differed significantly from those on plots with $\geqslant$60% mortality. Fuel complexes did not differ between unburned plots and plots with mortality levels $\leqslant$40%. On the Jasper Fire, groupings based on overall fuel complexes were slightly different, although generally followed the same pattern. There was no difference between 0 and 20% mortality and unburned plots, as well as no difference between 0–20% and 20–40%. However, there was a significant difference between unburned plots and 20–40% plots, but overall fuel complexes did not differ between the 20–40% and 40–60% mortality classes. Higher mortality classes (60–80% and 80–100%) differed from both low mortality classes as well as from each other. On the Pumpkin Fire the groupings of the mortality classes were precise, with no overlap. Unburned plots and burned plots with up to 40% mortality all fell in the same grouping. Midranges of mortality, from 40% to 80%, were not significantly different from one another, and the highest mortality range was dissimilar from all others.

4. Discussion

4.1. Stand structure

On both fires total basal area and tree density decreased with increasing mortality, as we hypothesized. However, basal area and tree density did not differ significantly from unburned levels up to 40% mortality and generally did not achieve target threshold values until over 40% of the trees were killed in the fires. Several studies and fire model predictions suggest that canopy fuels should be reduced in ponderosa pine forests to achieve a maximum basal area of 9.2–18.4 m^2 ha^{-1}, to both reduce crown fire spread and provide forest structures more in line with historical ranges (Fiedler, 1996; Fulé et al., 1997; Brown and Cook, 2006). On both fires tree densities were >250 trees ha^{-1}, the upper limit of tree density targets, until mortality exceeded 40% (Fiedler, 1996; Fulé et al., 2002; Brown and Cook, 2006; Fig. 2a and b). Thus, assuming the spatial pattern is similar to unburned areas, low-mortality plots likely have a high active crown fire potential similar to unburned stands.

Our findings supported our stand structure hypotheses in terms of basal area and stand density, but not our hypothesis that CBH would increase with increasing levels of mortality. CBH differed between burned and unburned plots similar to other studies of low severity, surface fires (Gaines et al., 1958; Agee and Skinner, 2005). However, contrary to our hypothesis CBH did not differ significantly among varying mortality classes on burned plots. This lack of difference was likely due to the relatively homogenous stands pre-fire, with similar DBH and crown base height (Sieg et al., 2006). A more significant difference would be likely in uneven-aged stands where smaller trees were removed with burning, leaving larger trees intact.

Crown fire potential likely varies greatly among mortality classes due to changes in these stand characteristics. Both low-mortality and unburned areas had fuel attributes of dense ponderosa pine forests that are more prone to crown fires compared to historical stand structures in the region (Covington and Moore, 1994). A difference between unburned plots and low mortality plots is the potential for a surface fire to transition to a crown fire, due to lower CBH and slightly higher surface fuel loading in unburned areas (Van Wagner, 1977; Vaillant et al., 2009). In contrast, as mortality increased the lower canopy bulk densities and higher CBH reduced the crown fire hazard. Midrange mortality areas will likely experience isolated tree torching, whereas high mortality areas (>80%) are unlikely to have crown fires due to the lack of an overstory. Contrary to our expectations, all mortality classes including unburned plots had a mean CBH above the threshold for a transition from a surface fire to a passive crown fire. However, these results must be interpreted with caution as the model used for this threshold is known to under-predict the potential for crown fire (Cruz and Alexander, 2010).

Snags provide important ecosystem functions, and thus managers often have target snag densities. Large snags, both in terms of diameter and height are important for cavity nesting birds and other wildlife (Bull et al., 1997). On our sites, only two trees met the >40 cm DBH requirement, but snags >30 cm DBH were more prevalent on both study areas. On the Jasper Fire, plots with >40% mortality met minimum snags density standards (5 snags ha^{-1}) recommended for ponderosa pine forests of >30 cm DBH trees (USDA, 2007). On the Pumpkin Fire, all plots with >20% mortality satisfied this minimum snag requirement. The large percentage (87%) of small diameter snags we observed is similar to results from other studies in these areas (Chambers and Mast, 2005; Spiering and Knight, 2005).

Regeneration on both wildfires was highly variable. Regeneration on the Jasper Fire was lower compared to other studies in the Black Hills that found an abundance of seedlings though they did observe patchy regeneration (Bonnet et al., 2005; Keyser et al., 2008; Battaglia et al., 2008). However regeneration on unburned plots was similar to these studies. These observed differences may be due to unfavorable regeneration conditions between these studies (before 2005) and our study in 2009. The precipitation for 2004 through 2007 was below normal, which could have resulted in mortality of some of the regeneration, especially heat-stressed seedlings in open, high-mortality areas. The regeneration we observed would be sufficient to provide overstory recruitment in some areas, without presenting excessive ladder fuels (<500 trees ha^{-1}; Battaglia et al., 2008). On the Pumpkin Fire, the highly variable, but generally low regeneration was similar to other studies in the Southwest where regeneration tends to be sporadic and episodic (Larson and Schubert, 1970; White, 1985; Bailey and Covington, 2002; Savage and Mast, 2005); thus pine seedlings may become established in the future under more ideal conditions for regeneration (Haire and McGarigal, 2010).

4.2. Surface fuels

On both fires, CWD loadings on unburned and low mortality plots fell below the "optimum" range, whereas levels on the highest mortality plots (80–100%) were above. In midrange mortality plots, between 40% and 80% mortality, CWD fell within the acceptable range (11.2–44.8 Mg ha^{-1}). This suggests that other than high severity areas of a fire, quantities of CWD do not present excessive fire hazard, but only mid-mortality plots have sufficient CWD to provide other ecosystem benefits, as defined by Brown et al. (2003). Even on high severity plots, the concern about high levels of coarse woody fuels may be relatively short-lived. Roccaforte et al. (submitted for publication) found that CWD peaked between

6 and 12 years post-fire in a Southwestern chronosequence study and only remained above the "optimum" range on wildfires $\leqslant$10 years old.

In contrast, low mortality and unburned plots had CWD loadings below the "optimum" range, suggesting that within these forests moderate intensity burning would be preferable to promote moderate accumulations of CWD. Other management actions, such as thinning, that leaves some activity fuels, would also be effective in increasing CWD levels (Brown et al., 2003). Similar to other studies of low-severity fires, low intensity burning was not effective in promoting either moderate accumulations of CWD or reducing basal area and other management actions would be required to achieve these goals (Pollet and Omi, 2002).

Due to high CWD loadings in high-mortality areas, we also assessed the percentage of fire-killed trees that were currently on the ground to determine if these fuel loadings would continue to increase, especially in light of the recent debate on salvage logging (Donato et al., 2006). On the Jasper Fire, 57% of all fire-created snags have fallen and are now contributing to CWD loadings and 78% of the snags have fallen on the Pumpkin Fire, which is consistent to other studies (Chambers and Mast, 2005). However, at the highest mortality class, fall rates were similar to other mortality classes on the Jasper Fire with 58% of snags down, whereas on the Pumpkin 94% were on the ground. The remaining snags, roughly 43% and 22%, respectively, of the original fire-created snags across all mortality classes on the Jasper and Pumpkin Fires, will continue to contribute to increased CWD loading as they fall, and thus loadings may exceed the "optimum threshold" for lower mortality levels as well. However, the majority of this woody material is on the ground; therefore the increase in loadings will be less substantial than loadings 10 years after the fire. The majority of the CWD in both fires is sound, which has lower flammability than rotten CWD. This suggests that flammability, while currently low, may increase in the next 10–20 years as CWD degrades.

In terms of mitigating future detrimental effects of high CWD loadings, logging plans immediately post-fire that utilize careful planning and implementation could ultimately be beneficial in creating sustainable forest conditions (Hutto, 2006; Peterson et al., 2009). Targeted removal of smaller diameter trees in high severity areas may be a more effective method to reduce CWD in the long-run, but remains unstudied. Alternatively, as snags begin to fall and fine fuels accumulate, prescribed burning when 1000-h fuel moistures are high would consume some CWD while diminishing detrimental soil heating. However, prescribed fire in these high mortality areas could kill pine seedlings, so excluding patches with regeneration in burning plans might be needed to promote pine establishment. Overall, there is limited regeneration found on both wildfires, thus a prominent overstory in high mortality areas is unlikely in the next several decades. As a result, future fuel loading and fire hazard may be different than observed in other studies in both ponderosa pine (Battaglia et al., 2008) and other forest types (Odion et al., 2004; McIver and Ottmar, 2007).

Herbaceous biomass followed the same trend on both fires and in both years of sampling, but actual quantities differed. Lower biomass on the Pumpkin Fire in both years than on the Jasper Fire can be explained in part by different sampling times for each respective fire, which corresponded with the season of high fire activity and not the peak biomass time. These differences may also relate to the more even distribution of precipitation throughout the growing season in the Black Hills (Shepperd and Battaglia, 2002). At least two other factors may have contributed to lower biomass in 2010 on both fires compared to 2009. First, on both fires, the month before sampling, precipitation was lower in 2010 than 2009. On the Jasper fire precipitation in July of 2009 was 8.99 cm, whereas it was only 5.66 cm in 2010 (High Plains Regional Climate Center, 2006. Available online at www.hprcc.unl.edu (accessed 07.02.11.)). On the Pumpkin Fire the difference was even larger with 1.14 cm versus 0.03 cm in June of 2009 and 2010, respectively (Western Regional Climate Center, 2010. Available online at www.wrcc.dri.edu (accessed 07.02.11.)). The second factor that might influence herbaceous biomass was precipitation the year before sampling. On both fires, precipitation in 2009 was lower than the 20-year average; whereas 2008 precipitation on both sites was average or above. Several studies have shown that precipitation of the previous year may impact herbaceous production (Milchunas, 2006; Laughlin and Moore, 2009).

Minimum loading of herbaceous biomass to maintain an active surface fire was only achieved on the highest mortality plots. Thus, limited herbaceous biomass would curtail surface fire spread in all lower mortality classes, whereas mid- to high mortality areas may support surface fires in some, but not all years. From the 2 years of sampling, only at the highest mortality class (80–100%) on the Jasper Fire had a high potential for surface fire spread consistently across all years, based on herbaceous biomass alone. Several studies have shown that basal area needs to be 14 m^2 ha^{-1} or less to significantly increase herbaceous productivity (Uresk and Severson, 1998; Wienk et al., 2004; Sabo et al., 2008). As discussed previously, basal area on our study sites was above this level until tree mortality exceeded 40% or 60%, indicating that it is unlikely to see large quantities of herbaceous biomass below this level of mortality. However, areas with >60% mortality, the combination of FWD loadings and herbaceous biomass should be sufficient to maintain an active surface fire. Creating conditions that allow for surface fire spread is an important step in the process of managing for more resilient ponderosa pine landscapes (Fulé et al., 2001). Due to the stand and surface fuel characteristics we found that mid-mortality classes (40–80%) were the most resilient to future disturbances, especially given future climate change predictions (Allen et al., 2002; Fulé, 2008).

5. Conclusion

Our study addresses the need for post-wildfire management on large, mixed-severity wildfires in ponderosa pine-dominated forests. We propose that management actions be tailored to burn severity and the focus of these actions should be on enhancing forest resilience given projected climate change (Hunter et al., in press). Management actions in low-severity portions of the fire would likely be similar to those actions recommended for unburned areas, as these areas are characterized by stand densities up to 130% higher than recommended levels for meeting both fire hazard and forest restoration targets. Thinning or other overstory fuels reduction measures should be taken to decrease crown fire potential.

High mortality areas have high CWD loadings and herbaceous biomass, but there is little possibility of a crown fire due to the limited number of live trees. Our data illustrate that stands with 80–100% mortality have excessive CWD loadings on both fires and may warrant fuels-reduction measures to lessen the potential for a high-intensity surface fires. Thus, treatments such as logging or prescribed fire to remove a portion of dead trees may be warranted. But, our understanding of the effects and effectiveness of such treatments is limited, as most studies have been conducted in highly productive forest types. Ten plus years post fire prescribed burning in high mortality areas may be warranted to decrease potential wildfire intensity, as these areas also have adequate fine fuels to sustain surface fires needed to ignite woody fuels as they rot. However, avoiding patches of pine seedlings in burning plans may be needed if regeneration of the pine forest is a goal.

Moderate levels of mortality resulted in stands that met many restoration goals of basal area reduction, increased snag recruitment and herbaceous production, and moderate levels of CWD.

Areas with 40–80% tree mortality would benefit from the reintroduction of frequent surface fires to maintain current stand structures. This would ensure that these stands would continue to resemble historical stand structures and fuels complexes, as well as meet fuel reduction targets. Currently these areas are the most sustainable and are likely more resilient given projected climate changes (Fulé, 2008), but require management to prevent them from becoming overly dense in the future. Given the challenges associated with managing large landscapes (Peterson, 2011), mid-mortality stands resulting from wildfires and resource use fires (Holden et al., 2007) provide more resilient structures that are likely easier to maintain compared to thinning overly-dense ponderosa pine stands.

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