Project Title: Ten years after the Biscuit Fire: Evaluating vegetation succession and post-fire management effects

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Abstract

Increases in the area of high-severity wildfire in the western U.S. have prompted widespread management concerns about post-fire forest succession and fuels. Key questions include the degree to which, and over what time frame: a) forests will regenerate back toward mature forest cover, and b) fire hazard increases due to the falling and decay of fire-killed trees, with and without post-fire (or 'salvage') logging. While a number of recent studies have begun to address these questions using chronosequences and model projections, we had the unique opportunity to track regeneration and fuel dynamics over a decade of post-fire succession by re-visiting our network of sample plots distributed in the 2002 Biscuit Fire in southwest Oregon. The ten-year benchmark for this iconic 'mega-fire' presented excellent learning opportunities, spanning gradients of forest type, fire combinations (single and reburn fires), and post-fire management intensity. We addressed questions on the following topics:

1) *Rates and patterns of natural succession in a large mixed-severity landscape fire*. Re-sprouted hardwoods and shrubs currently dominate the early-seral landscape, but conifer establishment has continued during the first post-fire decade, with increasing densities in most sample sites (71%) and patch-scale medians now averaging 2444 trees ha^{-1}. Survival of early conifer cohorts to the 10-year point has been ~69%. The mixed-severity mosaic is still a key determinant of successional trajectories: within 400 m of live-tree edges, tree densities exceed 1000 trees ha^{-1} and occupancy is 80-90%; whereas beyond that distance, densities and occupancy decline rapidly. The interiors of large patches have less representation of Douglas-fir (otherwise the most abundant conifer in the area) and greater dominance by knobcone pine (which has *in situ* seed sources via serotinous cones) and resprouted hardwoods/shrubs. Changes in conifer density and height (i.e., growth) had little relation to hardwood/shrub abundance or stature at 10 years post-fire, suggesting that competitive interactions have yet to exert a major influence.

2) *Decadal dynamics of live and dead fuels with and without post-fire logging.* Regarding surface woody fuels, which started from large treatment differences immediately following logging (stepwise increases with harvest intensity), we found converging trends among treatments at 10 years, with convergence nearly complete for fine fuels but not for coarse fuels. Fire-killed snag biomass of Douglas-fir decayed while standing at a statistically significant rate (single-exponential k=0.011), similar to or only slightly slower than down wood, suggesting that not all snag biomass will reach the forest floor. Live vegetation in this productive system (with abundant re-sprouting sclerophyllous vegetation) is beginning to dominate surface fuel mass and continuity (>100% cover) and likely moderates differences associated with woody fuels. Post-fire logging had little effect on live fuels or their change over time, suggesting high potential for stand-replacing early-seral fire regardless of post-fire harvest treatments.

3) *Effects of a reburn with and without prior post-fire logging*. Dead wood mass following an early-seral reburn (15-year interval) was 169 Mg ha^{-1}, approximately half that after a single long-interval fire (309 Mg ha^{-1}). The difference was due to greater time for decay and combustion in a second fire. Charring (black C creation) was also higher in the reburn by a factor of 2 for logs and 8 for snags. Notwithstanding future disturbances, projections suggest the near-halving of dead wood in reburn stands will persist for ~50 years, and then attenuate

by 100-150 years, illustrating the importance of stochastic variations in disturbance interval for long-term dead wood dynamics. Unmanaged reburn sites had very similar conifer regeneration densities and hardwood/shrub biomass to single-burn sites, whereas reburn sites that had been logged after the first fire were characterized by a shift toward much greater dominance by hardwoods and shrubs after a repeat fire – particularly *Ceanothus* species.

Background and purpose

This final report describes results from the project, “Ten years after the Biscuit Fire: Evaluating vegetation succession and post-fire management effects” (Project number 11-1-1-4), which was funded under Task Statement 1 of FA-RFA-11-1, Re-measurement Opportunities. The task statement solicited proposals to “re-measure existing field studies to assess the effects of high-severity fire on vegetation succession, and/or evaluate the effects of post-wildfire management.”

The management of forests following stand-replacing wildfire lies at the nexus of many pressing environmental and societal concerns including the maintenance of biodiversity, local economic viability, ecosystem services, and the response of forests to climate change. Large-scale severe fires are expected to become increasingly common throughout much of North America, necessitating scientific data to inform post-fire management options and outcomes. To date, studies of post-fire vegetation and management have been largely limited to the few years immediately following fire or, if longer-term in scope, based on retrospective, prospective, or chronosequence inference. These approaches have been valuable, but necessarily either lack foundational short-term data, confound space and time, or project assumptions that are not fully validated. Thus there is a need for studies that track post-fire succession at the same locales through time.

The 2002 Biscuit Fire has been called the most educational fire this side of Yellowstone (see, e.g., Turner et al. 2003). As the largest forest fire in Oregon’s recorded history, the Biscuit became a national focal point for key issues relating to post-fire management—uncertain forest regeneration, fuel succession, and a relatively large post-fire (‘salvage’) logging plan (USDA 2004). Many concerns were based on notorious reforestation difficulties in the region. The Biscuit Fire proved to be an unparalleled learning platform on these issues, in part because portions of it burned over the 1987 Silver Fire, itself partly salvaged in 1988. This sequence of disturbances and management actions provided an excellent factorial combination of stands burned once, burned twice, and both of these with and without post-fire management (Figs. 1-2).

Early post-fire data from the Biscuit Fire yielded several surprises, influencing the way large severe fires are perceived and managed. These surprises included unexpectedly robust forest regeneration (Donato et al. 2006a, 2009a; Halofsky and Hibbs 2009), counterintuitive (increasing) effects of post-fire logging and planting on re-burn potentials (Thompson et al. 2007, 2010; Donato et al. 2006a), and abundant and diverse wildlife and vegetation communities following a re-burn (Fontaine et al. 2009, Donato et al. 2009b). These important results prompted a synthesis paper of early Biscuit findings by a large interdisciplinary team of federal and university scientists (Halofsky et al. 2011). While it was important, informative, and largely unprecedented to capture the initial effects of this combination of disturbances, many have wondered whether these documented effects were ephemeral, or instead represent the beginning of unique successional trajectories affecting forest processes for decades or centuries. Only through continued re-measurement campaigns can we shed light on this question.

From 2004-2006, we conducted field studies of vegetation and fuels in high-severity (>90% aboveground mortality) portions of the Biscuit Fire (Campbell et al. 2007; Donato et al. 2006a,b; Donato 2008; Donato et al. 2009a,b,c; Donato et al. 2013; Fontaine 2007; Fontaine et al. 2009; Fontaine et al. 2010; Law et al. 2004). These studies account for the majority of peer-reviewed scientific papers from the Biscuit Fire to date. One of our earliest publications reported some key surprises and generated especially high interest and scientific discussion (Donato et al. 2006a,b). These conclusions were later supported by multiple studies from other researchers (Shatford et al. 2007; Thompson et al. 2007; Halofsky and Hibbs 2009) as well as our own further publications (Donato et al. 2009a,b; 2013). Re-measurement of our sample allowed an evaluation of how those earlier findings have played out over the first post-fire decade.

Our objectives were to:

1) *Quantify rates and patterns of natural post-fire succession by environmental setting*. We previously reported surprisingly robust conifer regeneration in this landscape-scale mixed-severity fire (Donato et al. 2009a). In a region where mesic northern forest communities intergrade with xeric southern chaparral-type communities (Whitaker 1960), we were especially interested in these key questions: a) At 10 years post-fire, what are apparent rates of continued conifer establishment, as well as survival and growth of early cohorts? b) How are these processes influenced by associated broadleaf/shrub abundance and growth, and what are the implications for successional trajectories? c) Within the mixed-severity mosaic, how does proximity to live-tree edge influence decade-scale tree establishment processes?
2) *Quantify decade-scale effects of post-fire logging intensity on live and dead fuel profiles.* Our previous findings (Donato et al. 2006a,b; Donato et al. 2013) were consistent with a large number of studies (McIver and Ottmar, 2007; Monsanto and Agee, 2008; Keyser et al., 2009; McGinnis et al. 2010; Ritchie et al., 2013; Dunn and Bailey 2015; Peterson et al., 2015) showing a spike in surface fuels immediately following logging, a spike which scales with harvest intensity. Most studies also agree that, later, large woody surface fuels will accumulate to a greater degree (or at least converge) in unlogged stands. However, the time scale over which a convergence or switch occurs, and how it varies by logging intensity, is poorly quantified for most systems. We addressed these questions: a) How do stands logged at various intensity (including unlogged) compare in terms of surface woody fuel accumulation at 10 years post-fire? b) How do stands logged at various intensity (including unlogged) compare in terms of live fuel accumulation at 10 years post-fire? c) For fire-killed wood, how do the relative rates of fragmentation, standing decay, and surface decay vary by species, size class, environmental setting, and logging treatment, and how do these influence surface accumulations? With these data in hand, we could readily address an additional question: d) What are the implications of these decay rates of fire-killed wood for long-term emissions of carbon to the atmosphere?

3) *Quantify the longer-term influence of reburning on live and dead succession, and the effect of prior post-fire logging on that influence.* Part of the Biscuit Fire burned over the partially-salvaged 1987 Silver Fire (15-year interval). Earlier studies indicated unique vegetation communities establishing in the re-burn area (Donato et al. 2009b), as well as slightly higher fire severity in stands that were previously salvaged (Thompson et al. 2007). However, the influence of reburning on dead wood has scarcely been studied, as has the degree to which reburn impacts on the ecosystem are influenced by prior salvage logging. We asked these questions: a) How does reburning influence dead wood abundance, character, and long-term dynamics compared to that following a single (long-interval) stand-replacing fire? b) How does prior post-fire logging (after the first fire) influence stand and community structure in the event that a reburn occurs?

Study description and location

Original design

The study was conducted in the Klamath Mountains of southwest Oregon (Fig. 1), on the Rogue-Siskiyou National Forest. The area is in the *Abies concolor* and mixed-evergreen forest zones, comprising white fir, Douglas-fir, and tanoak plant associations (USDA 2004). All study sites were mature/old forest prior to burning with >90% aboveground mortality in 1987 and/or 2002. We previously demonstrated that all treatments captured similar ranges in pre-fire forest density, basal area, and composition to the extent possible (Fontaine et al. 2009; Donato et al. 2009a,b; Donato 2008).

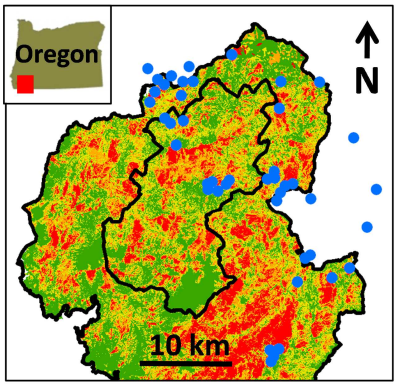

Figure 1. *The 2002 Biscuit Fire, SW Oregon. Red= high-severity burn areas. Inner perimeter line is 1987 Silver Fire. Blue dots represent sites in which multiple sample plots are located, from our previous studies.*

We used a combination of 'intensive' and 'extensive' plots, both of which are based on widely accepted regional protocols (see, e.g., Donato et al. 2009a). Plots were permanently marked with high-precision GPS and re-bar. The 1-hectare intensive plots are based on regional Forest Inventory protocols (USDA 2003) and are distributed among the various post-disturbance conditions (Table 1, Fig. 5). The 0.008-hectare extensive plots are based on agency regeneration stocking surveys (USDI 2003) and spaced at 250-meter intervals along elevational contours within high-severity burn patches (Table 1, Fig. 5).

In intensive plots, we measured all vegetation and woody biomass components. Forbs were quantified on a subplot basis (species, cover, height), woody shrubs on an individual basis (crown/stem metrics, live+dead components), conifer seedlings individually in both circular and long rectangular subplots (species, age, height), and ground cover recorded. Snags were recorded for species, dbh, height, decay class, bole scorch, branch loss rating, and charring (USDA 2003; Donato et al. 2009a; Campbell et al. 2007; Donato 2008). Down wood was measured by size class along 300 m of planar intercept transect in each plot (Brown 1974).

Permanent photo points were taken at each subplot. Measurements in extensive plots, which were designed to be more rapid and sample across a larger area, included all conifer seedlings individually (as in intensive plots), cover of all vegetation by species/height class, and ground cover (Fontaine et al. 2009; Donato et al. 2009a).

Re-measurement

Despite very challenging conditions of dense vegetation and deadfall, we successfully located virtually all the original plot centers that were part of the re-measurement campaign. High-precision GPS, a metal detector, and permanent photo points were all essential tools. A small proportion of plots could not be re-measured due to either intervening treatments (e.g., slash burning) or unsafe access conditions; however our overall sample size and continuity between measurements were not significantly influenced by these exclusions. Moreover, as outlined in our proposal (and below), we were able to add several additional plots that expanded our scope to areas very distant from unburned edges and also resulted in a similar overall sample size. Our basic sampling approach was to repeat the measurements from the original study in each plot, with some additional data taken, as detailed below for each research objective. We organize our research objectives here by 1) natural vegetation succession, 2) post-fire management effects on live and dead fuels, and 3) reburn impacts.

Objective 1

Our sample design for quantifying vegetation succession in the unmanaged majority of the Biscuit Fire is described in detail by Donato et al. (2009a) and Fontaine et al. (2009). Briefly, we originally sampled 11 high-severity burn patches (size range 15 ha – 13,000 ha) using a combination of 1-ha intensive plots and 0.008-ha extensive plots distributed at regular 250-m intervals. Burn patches were saturated with plots to the extent practicable, notwithstanding constraints of safe access, etc. We augmented our sample with a dozen new extensive plots placed at sites identified to be the most distant from seed sources (surviving trees) based on agency spatial data. For the 10-year re-measure, we collected data on the whole vegetation community as described above for each plot type. For our objectives of quantifying natural tree establishment processes, we excluded seedlings that showed evidence of having been planted (these were present in only a handful of sites; evidence included protective netting, soil scalping, etc.). Although not permanent plots in the classical sense with tagged individual trees, careful attention to aging each regenerating tree (along with data on height, species, etc.) allowed estimates of survival, mortality, and additional recruitment. We evaluated interactions between broadleaf and coniferous components (and inferred successional trajectories) by relating their respective abundance and height profiles to each other.

Objective 2

Changes in live and dead fuels with and without post-fire logging were quantified by re-measuring our sample of intensive plots in logged and unlogged sites (Before-After-Control-Intervention [BACI] design; see Donato et al. 2013). To more accurately estimate biomass values and tissue allocation for live vegetation, we conducted a new set of allometries specific to the decadal growth form of post-fire resprouted hardwoods and shrubs, via destructive harvest and lab processing. To more accurately characterize the dynamics of snag decay and surface deposition, we added detailed estimations for each tree regarding apparent volume loss (i.e., reduction in branch/bole material due to breakage) as well as apparent density loss (i.e., the standard decay class metric of most studies).

Additionally, we collaborated with the Forest Service to obtain destructive samples of 115 boles and 250 branches still standing 10 years after the fire, as well as 60 logs and 86 branches deduced to have been killed in the Biscuit Fire (by presence of surface charring) and that fell within the next year (saw cuts datable to known salvage operations). In the case of standing dead boles, tree-average density was calculated as the average density of three transverse samples (cookies) collected from the lower, middle, and upper third of each tree, weighted by a factor of 0.60, 0.36, and 0.04, respectively to account for volume proportion by height (derived from the taper equations of Arney, 2009). Single transverse subsamples were used to determine wood density for branches and downed logs. Density was determined after oven-drying at 95^{o}C to constant mass; an 8% downward correction was then applied to account for oven shrinkage and afford direct comparison with published green-tree densities. Rate constants describing the decomposition of fire-killed trees were calculated as $k= -ln(D_t / D_o)/t$, where D_0 = live wood density, D_t = wood density measured 10 years after death in fire, and t= 10 years between 2002 and 2012. D_0 was assumed to be 0.39 and 0.45 g cm^{-3} for *Pinus* species and Douglas-fir, according to Maeglin (1972) and USFS (1965), respectively. All standing and downed dead wood samples were stratified by species group (Douglas-fir, *Pinus* species [*P. lambertiana* and *P. ponderosa*]), diameter class (range 7 to 146 cm DBH for boles, 0.2 to 7.6 cm diameter for branches), and across the study area wherein other fuel mass was evaluated. Rate constants describing fire-killed whole-tree fall were calculated as $k= -ln(C_t / C_o)/t$, where C_0 = plot-wide count of standing dead stems measured in 2004, C_t = plot-wide count of standing dead stems measured in 2012, and t= 8 years. Rate constants describing fragmentation of standing fire-killed trees were calculated as $k= -ln(M_t / M_o)/t$, where M_0 = bole, branch, and bark mass, allometrically-modeled from 2012 surveys with the artificial assumption that each tree was live and entire, and M_t= M_0 adjusted to account for any volume loss associated with breakage and fragmentation observed in 2012. In this way fragmentation as a flux rate is separate and independent from simultaneous volume loss via whole-tree fall or density loss via decomposition.

We compared major fuel components at each time point and their changes since treatment using linear mixed effects (LME) models that included fixed effects of treatment, time, and their interaction; as well as a random effect of plot to account for repeated measures (Zuur et al., 2009).

Objective 3

Longer-term effects of a reburn were evaluated by re-measuring our set of intensive plots from reburn areas (Donato et al. 2009b). Our analysis focused on the degree to which two sequential stand-replacing fires reduce dead wood biomass compared to a single fire, and how any difference may project forward into succession. We also quantified the degree to which dead wood particles were deeply charred by the reburn compared to a single fire (generation of black carbon, decay resistance, case-hardening, etc.). In these reburned sites, we also measured snag fragmentation and fall rate (methods as in Objective 2) to determine whether rates differed from single-burn areas. Finally, we compared live and dead vegetation components between reburn sites that had been salvaged after the first fire versus reburn sites that were unmanaged.

Key findings

Objective 1 – Natural succession

Regeneration of conifers continued throughout the first post-fire decade (Fig. 2). As a result, 71% of plots showed increased density from year 4 to year 10, while 16% showed a decrease (12% remained unchanged). Based on whorl counts, apparent survival of early cohorts (the first four years of establishment; i.e., those measured in the prior survey) was 69%. Increases were mainly in the higher-elevation and wetter plant associations of the fire area, while drier associations remained at similar densities. Patch-scale median conifer density at year 10 averaged 2444 trees ha^{-1} across our sample.

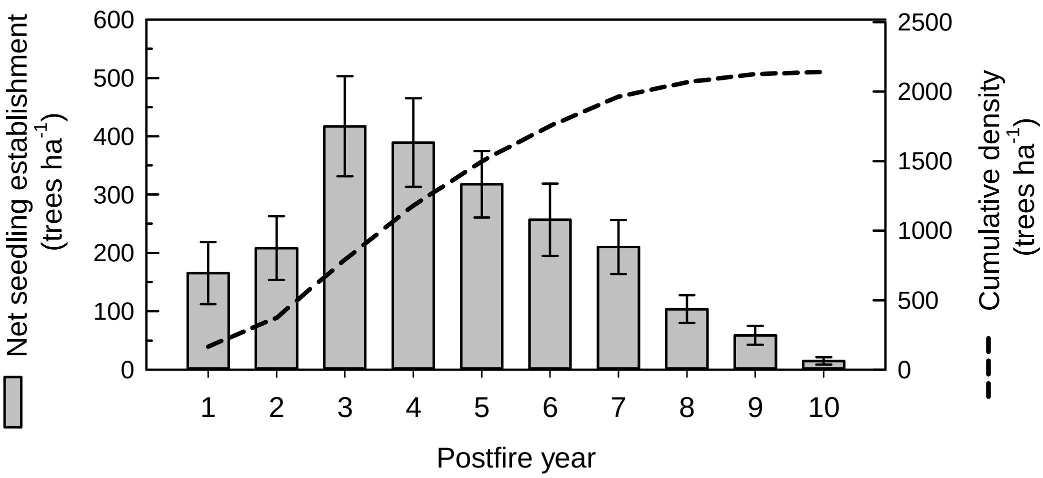

Figure 2. *Temporal patterns of conifer establishment during the first decade after the Biscuit Fire.*

Increases in density over time occurred at all distances from live-tree edges, but were strongest at distances less than 300-400 m (Fig. 3a). Competitive interactions, either among conifers (Fig. 3b) or between conifers and hardwood/shrubs (Fig. 3c) were apparently unimportant in driving changes in density over time.

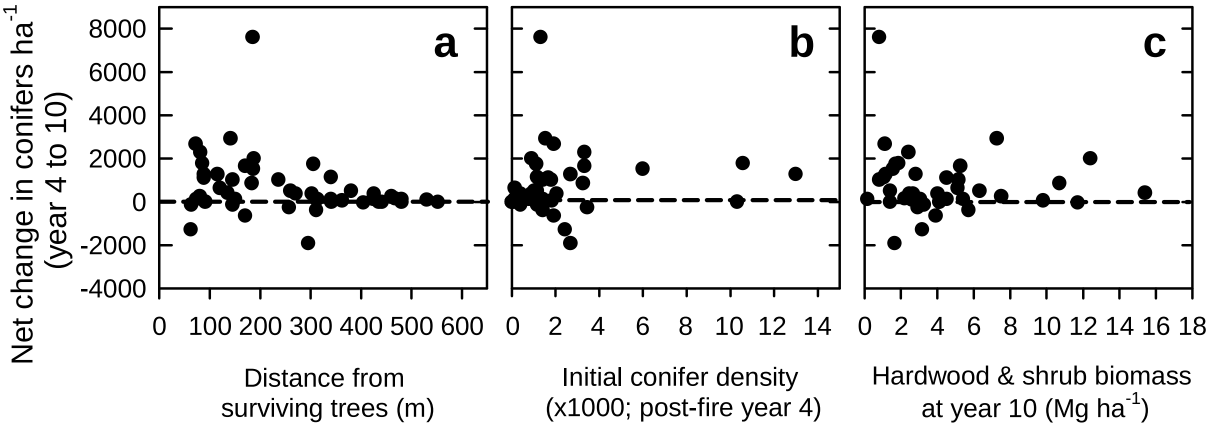

Figure 3. *Relationship between change in conifer density over time and a) proximity to seed source, b) conifer density (potential tree-tree competition), and c) hardwood/shrub abundance (potential competition between life forms).*

The net result of continued conifer establishment at varying distances was increased median density at virtually all distances; however the key threshold of ~400 m (identified in the previous study) was still apparent at a decade post-fire (Fig. 4). Below this distance, median densities exceeded 1000 trees ha^{-1} and most plots were occupied (stocked). Beyond 400 m, median densities were well below 500 trees ha^{-1} and occupancy was more variable.

Our extended sampling in the areas most distant from seed sources (1500-2700 m based on spatial analyses of USFS data from our earlier studies) revealed two interesting findings. First, the spatial data clearly underestimate the presence of seed sources, as many of the sites to which we traveled had surviving trees within a shorter distance than the GIS data indicated. As a result, the actual distance from live-edge of the new plots ranged from 525-1568 m, with a mean of 1007 m. This finding may partly explain the second one, which is that conifers were still present in about half of these most distant plots, and median conifer densities still registered at 127 trees ha^{-1}.

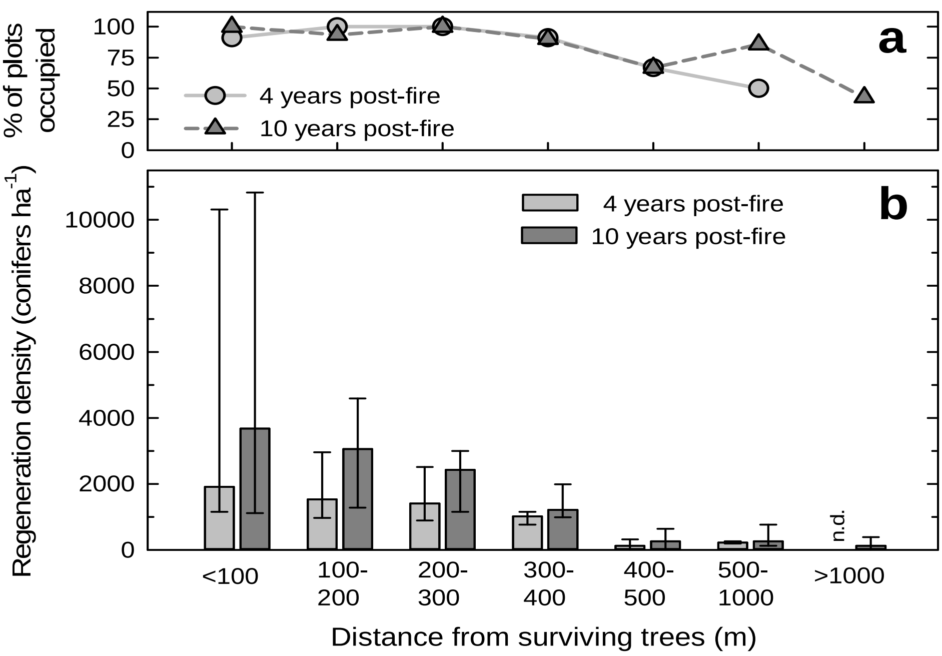

Figure 4. *Conifer occupancy and density as a function of increasing distance from live-tree edges (seed source).*

Conifer species composition varied not only by plant association, but by distance to edge. In particular, the relative balance of Douglas-fir (*Pseudotsuga menziesii*) and knobcone pine (*Pinus attenuata*), which together accounted for ~90% of all regenerating trees, showed a clear trend with a threshold, again at 400 m. Douglas-fir, which depends on seed dispersal from live trees, dominated among conifers within 400 m of an edge; while knobcone pine, which has serotinous cones and therefor an *in situ* seed source, attained co-dominance beyond 400 m (Fig. 5).

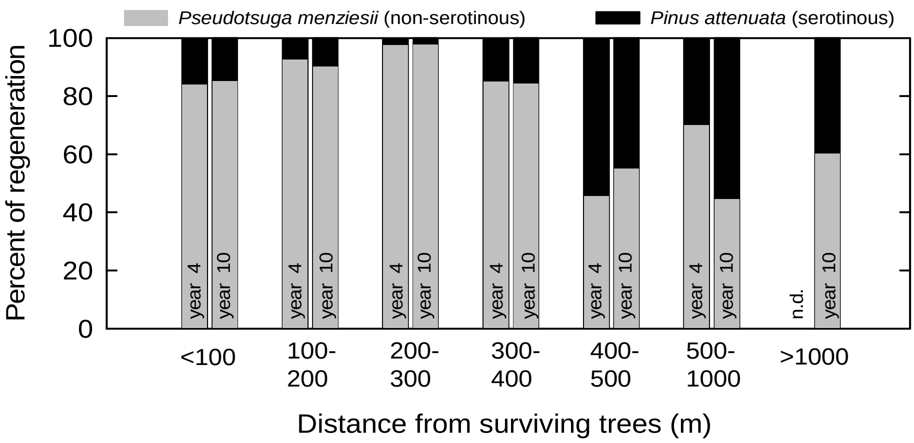

Figure 5. *Relative dominance of the two most abundant conifer species, Douglas-fir and knobcone pine (~90% of all regeneration), as a function of distance from live-tree edge.*

Despite extremely robust re-sprouting, growth, and general site dominance of hardwoods and shrubs in the Biscuit Fire (>100% cover), there were no strong competitive interactions apparent between conifers and hardwood/shrubs at 10 years post-fire. Neither the density nor height of conifers showed a significant relationship with hardwood biomass or height (Fig. 6).

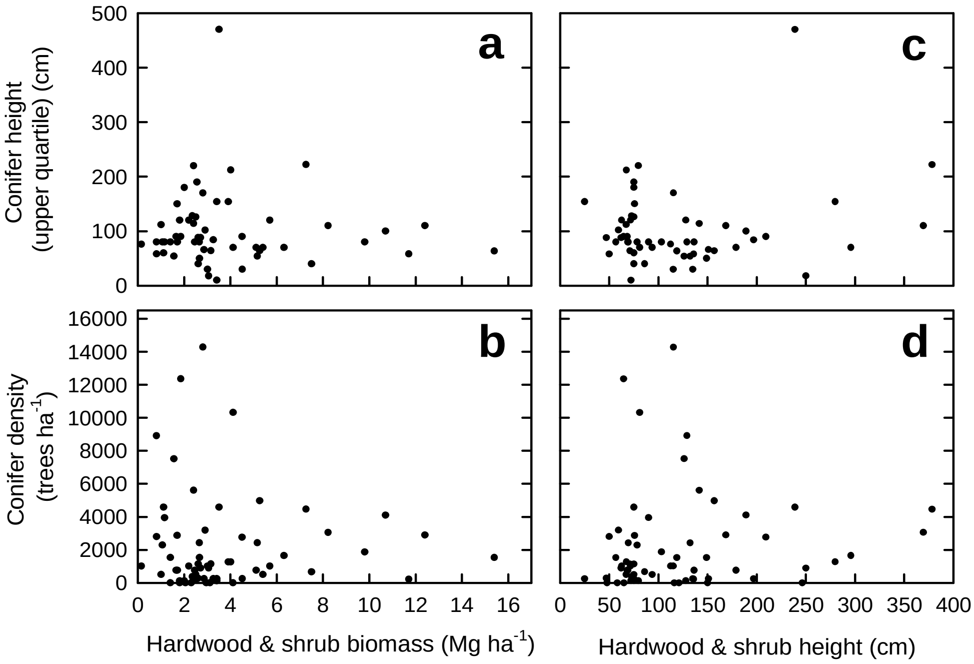

Figure 6. *Relationships between conifers and hardwood/shrubs in terms of abundance and stature.*

Objective 2 – Post-fire logging effects on fuels

Surface dead woody fuels, which differed strongly immediately following post-fire logging (stepwise increase in surface fuels with harvest intensity), showed converging trends among treatments at 10 years post-fire. For coarse fuels, high-intensity logged sites (>75% basal area removed, with cable yarding) showed little change over time, but moderate-intensity logged sites (25-75% basal area removed, with helicopter yarding) increased over time to effectively catch up to high-intensity sites (Fig. 7). Coarse fuels also increased in unlogged sites, but had not reached the level of either logged treatment at this time point (Fig. 7). For fine fuels, changes over time were qualitatively similar to coarse fuels, but were reaching parity among treatments at 10 years (Fig. 7). Forest floor mass was unaffected by post-fire logging and simply increased with successional time in all sites (Fig. 7).

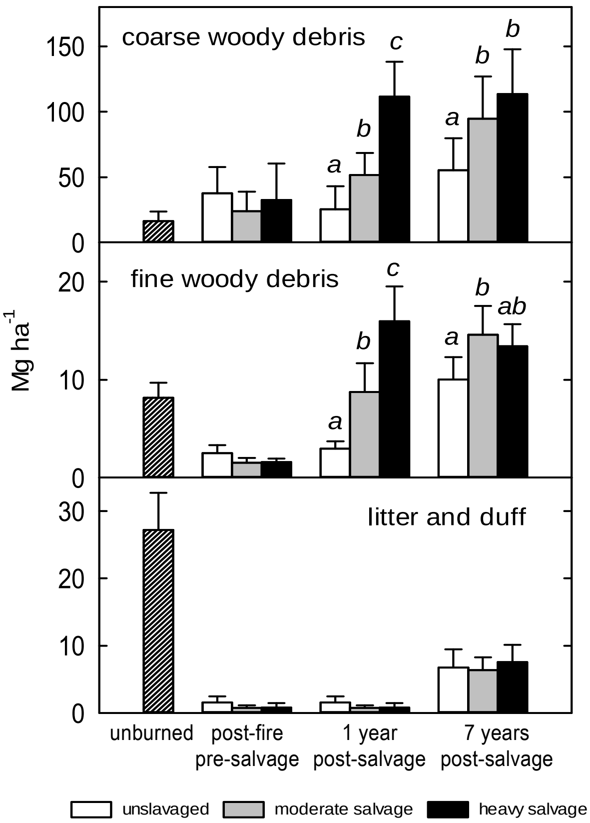

Figure 7. *Surface dead fuel loads at specified time points since stand-replacement fire and post-fire logging of varying intensity. Coarse woody debris are dead wood fragments >7.6 cm diameter. Fine woody debris are dead wood fragments <7.6 cm diameter. Litter and duff is comprised of dead leaf litter and the soil O-horizon. Error bars represent standard errors of the mean among 8, 13, 13, and 10 plots for unburned reference, unlogged, moderate-intensity logged, and high-intensity logged plots, respectively. Superscript letters appear when there is a significant difference among treatments within a given time point, based on the LME regression model.*

Standing woody fuels also showed some convergence among treatments, but statistical comparisons were largely unchanged at 10 years post-fire (Fig. 8). Most notably, it is clear that most of the fine fuels accumulating on the surface have been from small trees (“low suspended fine woody debris”) rather from large trees (Fig. 8). Inputs from that pool are, however, nearly complete, as there is little such material left to fall regardless of treatment.

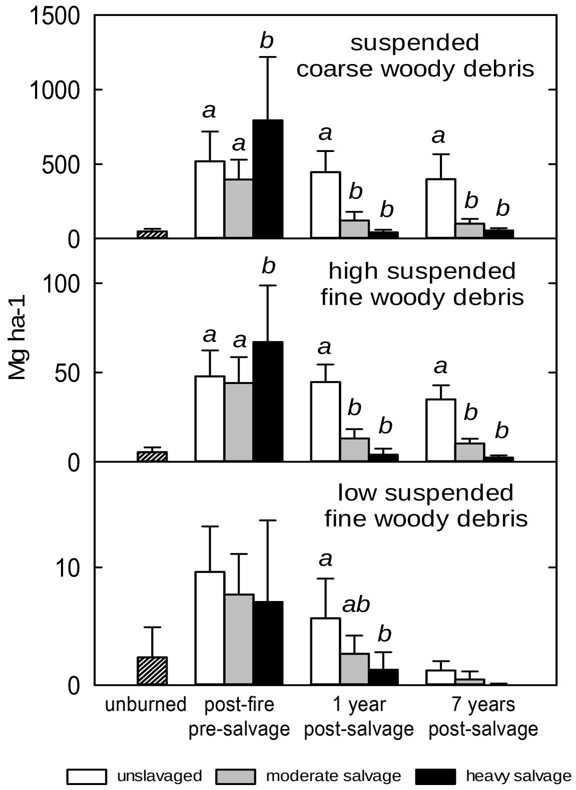

Figure 8. *Standing dead fuel loads following stand-replacing wildfire, post-fire logging, and in unburned reference stands. Wildfire occurred in 2002, salvage occurred in 2005. Suspended coarse woody debris includes all standing dead wood >7.6 cm diameter. High suspended fine woody debris are the branches and tops <7.6 cm diameter attached to dead trees >10 cm DBH (largely suspended at heights >4 m). Low suspended fine woody are the branches and tops <7.6 cm diameter attached to dead trees <10 cm DBH (largely suspended at heights <4 m). Error bars represent standard errors of the mean among 8, 13, 13, and 10 plots for unburned reference, unlogged, moderate-intensity logged, and high-intensity logged plots, respectively. Superscript letters appear when there is a significant difference among treatments within a given time point, based on the LME regression model.*

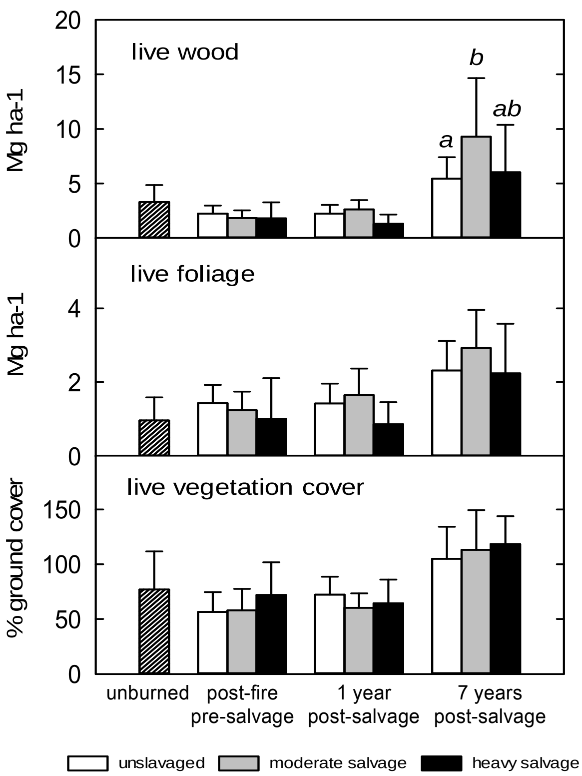

Figure 9. *Live fuel loads following stand-replacing wildfire, salvage logging, and in unburned reference stands. Wildfire occurred in 2002, salvage occurred in 2005. Live wood and foliage are almost entirely that of regenerating vegetation, largely resprouting shrubs. Live vegetation cover is the sum of projected shrub cover and the cover of ground surface vegetation. Error bars represent standard errors of the mean among 8, 13, 13, and 10 plots for unburned reference, unlogged, moderate-intensity logged, and high-intensity logged plots, respectively. Superscript letters appear when there is a significant difference among treatments within a given time point, based on the LME regression model.*

Live fuels in this productive system (characterized by robust re-sprouting hardwoods and shrubs) were largely unaffected by post-fire logging (Fig. 9). By post-fire year 10, mass of this fuel component is approaching that of fine dead woody fuels, and its cover is averaging over 100%, indicating high spatial continuity (Fig. 9).

Changes in dead fuel loads over time show that moderate-intensity logged plots changed in similar ways to unlogged plots, as there was still remaining material standing to fall and deposit on the surface, while high-intensity logged plots showed little to no change (Fig. 10). Importantly, changes in live fuels did not differ significantly among any of the treatments (Fig. 10).

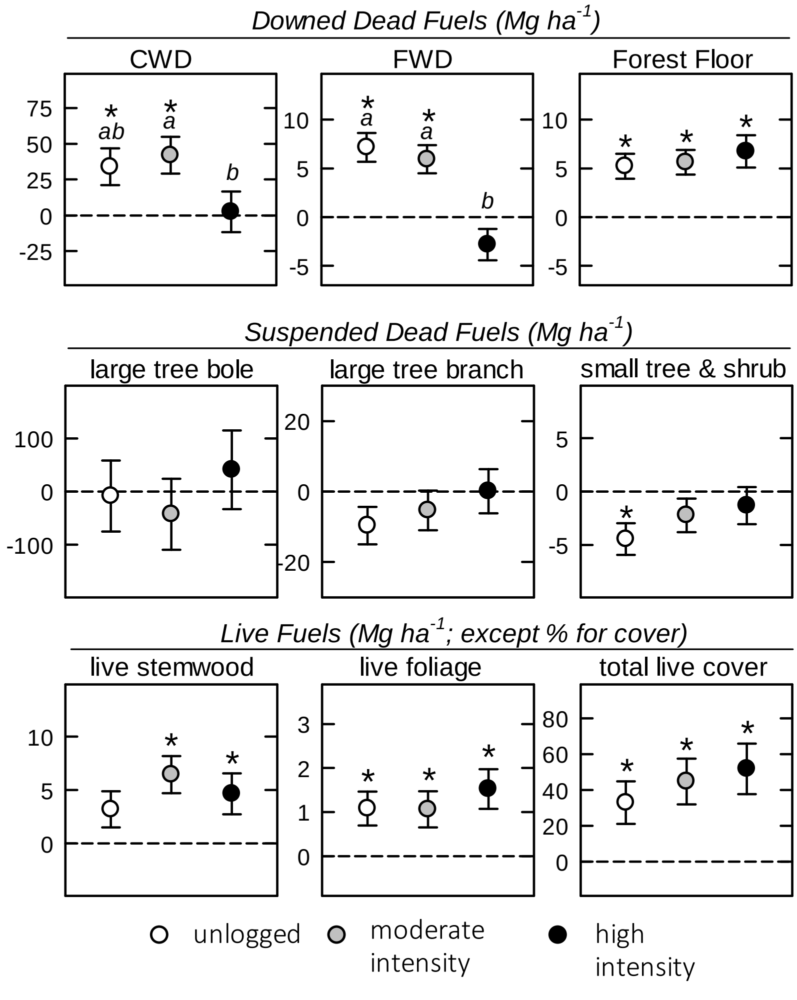

Figure 10. *Accumulation or depletion of various fuel classes during the 7 years following post-fire logging. Fuel classes are defined in previous figures. Error bars represent standard errors of the mean among 8, 13, 13, and 10 plots for unburned reference, unlogged, moderate-intensity logged, and high-intensity logged plots, respectively. Superscript letters appear when there is a significant difference among treatments; asterisks denote changes statistically different than zero (p<0.05; linear mixed effects model).*

Assessing the fall of standing dead fuels confirms that both whole-tree fall and fragmentation rates are higher for small-diameter than for large-diameter dead trees (Figure 11). Fall and fragmentation rates were generally not different between logged and unlogged stands, although large trees retained in logged stands did fall and fragment at faster rates than did similar-sized trees in unlogged stands. Average fire-killed bole decay rates were higher for Douglas-fir than for *Pinus* species (Figure 12). Observed differences between decay rates of standing and downed boles were minimal and not statistically significant for Douglas-fir (>85% of the trees killed in the Biscuit Fire). Notably, the 95% CI for the decay rate of standing dead bolewood excluded zero, suggesting that significant density loss occurred in standing snags. Moreover, the observed decay rates of fine woody fuels (<7.6 cm diameter; largely branchwood) suspended in the canopy were not discernably different than fallen fine woody fuels (Figure 12).

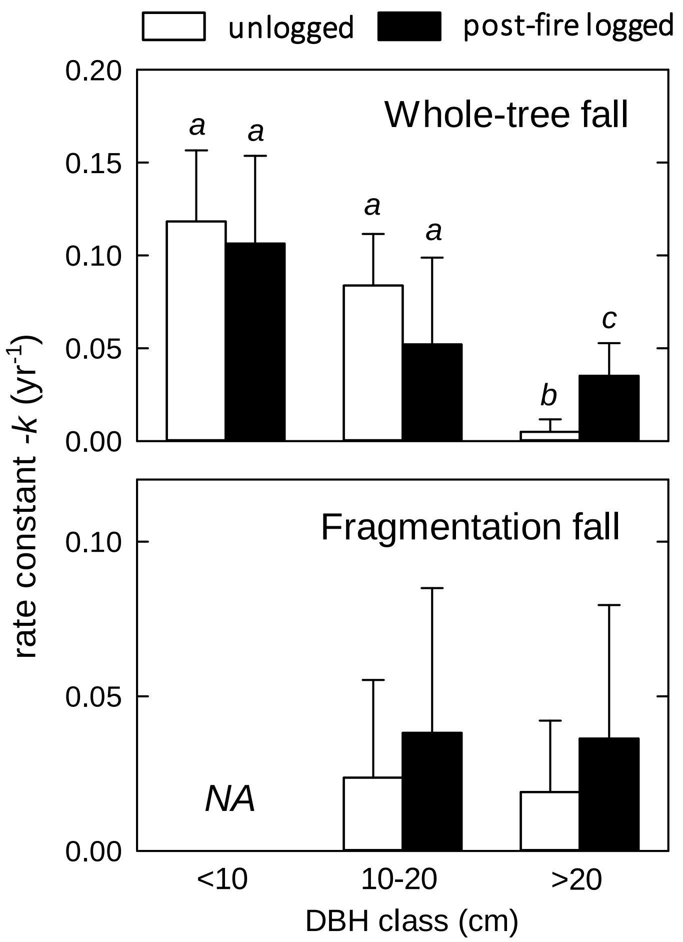

Figure 11. *Fall rates of fire-killed trees. Rates expressed as first-order exponential decay constants, where k = [ln(volume 10 years post-fire/volume 2 years post-fire)]/8 years. Error bars represent standard errors of the mean among 13 and 23 plots for unlogged and logged plots, respectively. Superscript letters appear when there is a significant difference among treatments (p<0.05, ANOVA applied to linear fractional loss rather than k constant). Fragmentation was not assessed for standing dead trees <10 cm DBH which typically experience whole tree fall.*

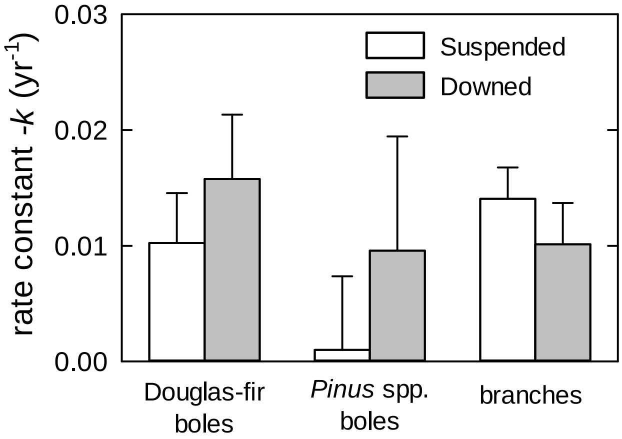

Figure 12. *Decay rates of fire-killed trees. Rates expressed as first-order exponential decay constants, where -k = [ln(live wood density/density 10 years post-fire)]/10 years. Error bars represent standard errors of the mean.*

As for the implications of decomposition of fire-killed trees for carbon emissions to the atmosphere, decomposition was predictably highest for fire-killed leaves and fine roots and lowest for large diameter wood (Fig. 13). Decomposition rates varied somewhat among tree species and was only 35% lower for trees still standing than for trees fallen at the time of the fire. We estimate a total of 4.7 Tg C was killed but not combusted in the Biscuit Fire, 85% of which remains 10 years after (Fig. 13). Biogenic carbon emissions from fire-killed necromass was estimated to be 1.0, 0.6, and 0.4 Mg C ha^{-1} yr^{-1}, 1, 10, and 50 years after the fire, respectively; compared to the one-time pyrogenic emission of nearly 17 Mg C ha^{-1}.

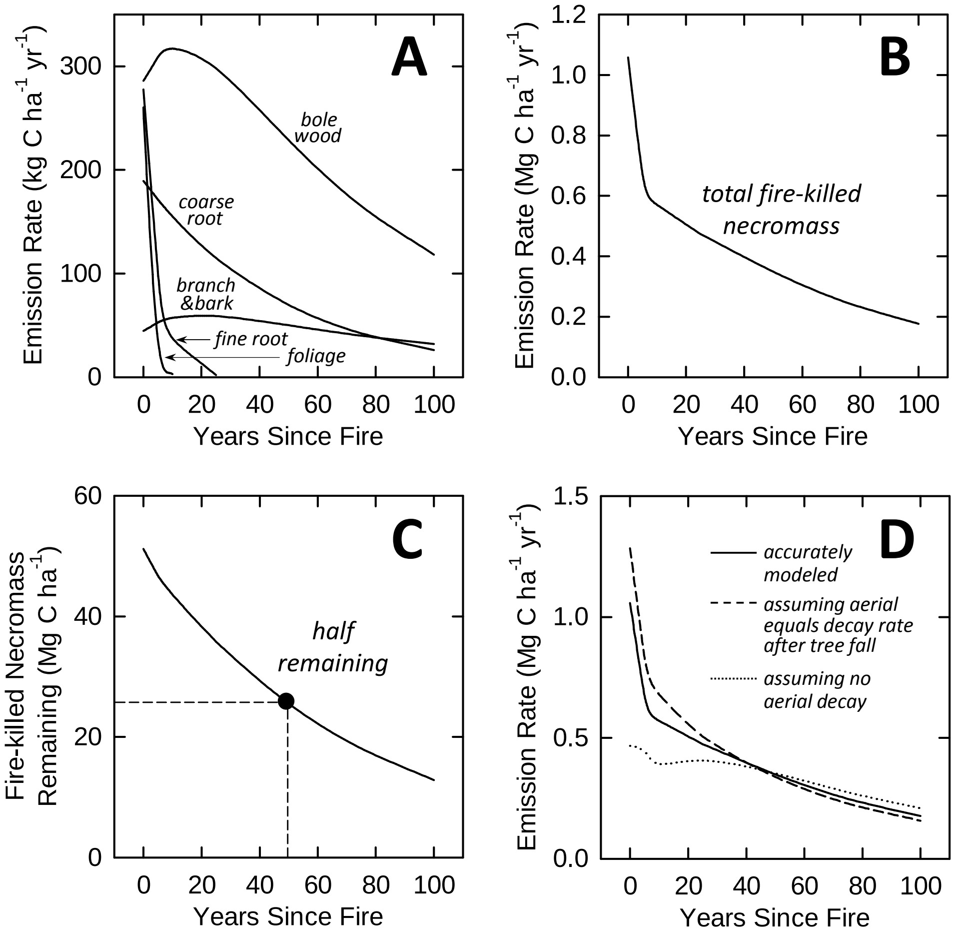

Figure 13. *Implications of decomposition of fire-killed trees for carbon emissions to the atmosphere over time.*

Objective 3 – Reburn impacts

Total dead wood abundance (standing+down) following the reburn (169 ±83 Mg ha^{-1} [95%CI]) was 45% lower than after a single fire (309 ±87 Mg ha^{-1}) (Fig. 14). Whereas a single fire in mature forest both consumed and created dead wood (by killing large live trees), an early-seral reburn only consumed dead wood (few large live trees to kill). The lower wood quantities in the reburn were due to both greater time for decay (15-year head start) and combustion in a second fire. Charred biomass (black carbon generation) was higher in reburned stands by a factor of 2 for logs and 8 for snags (Fig. 15). Re-measurement of reburn stands at 10 years indicated that, despite potential 'case-hardening' associated with higher charring levels, rates of snag fragmentation (k=0.024) and falling (k=0.003) were not different than in other high-severity areas of the Biscuit Fire.

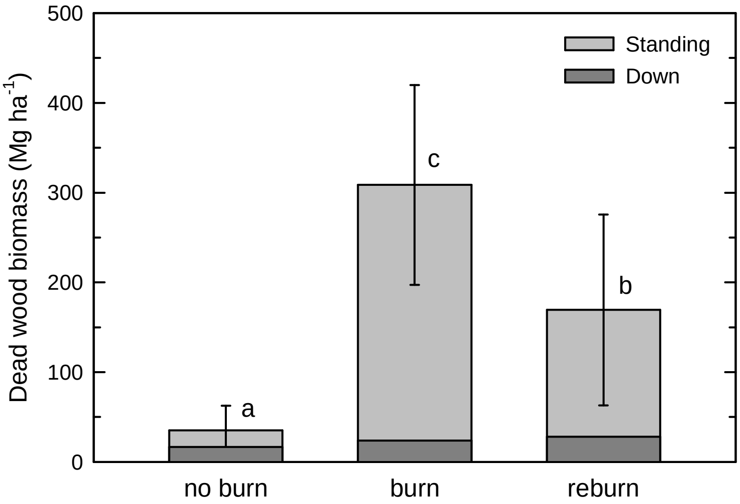

Figure 14. *Total dead wood biomass (mean, 95% CI) in stands of different recent disturbance histories. Confidence intervals are for the total dead wood pool. Lower-case letters above bars indicate whether confidence intervals mutually exclude other groups' means.*

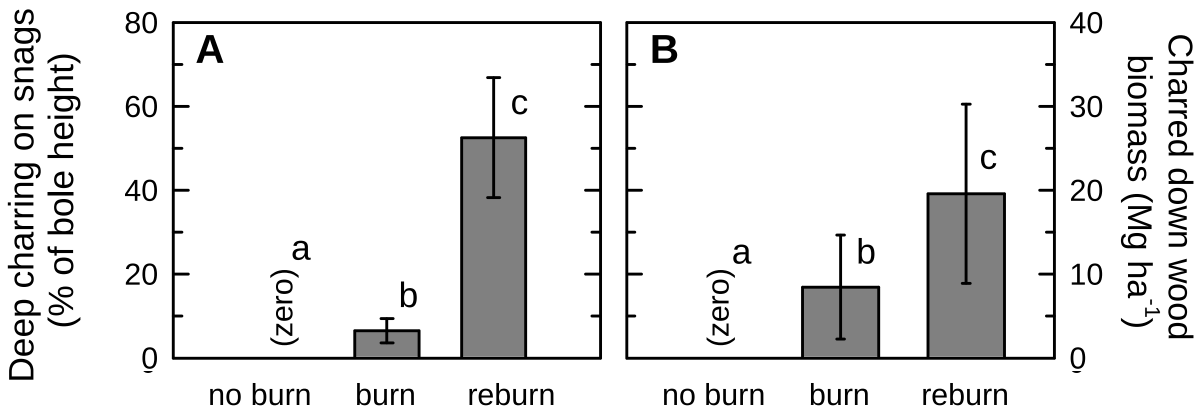

Figure 15. *Degree of charring on dead wood in stands of different recent disturbance histories (mean, 95% CI). Lower-case letters above bars indicate whether confidence intervals mutually exclude other groups' means.*

Parameterized into traditional dead-wood successional models (future disturbances aside), projections suggest: a) the near-halving of dead-wood mass in reburn stands will persist for ~50 years until the recruitment of new dead wood begins, with the difference attenuating by 100-150 years, and b) reburning shifts the temporal lowpoint of dead wood abundance to 50 years earlier in stand development (Fig. 16).

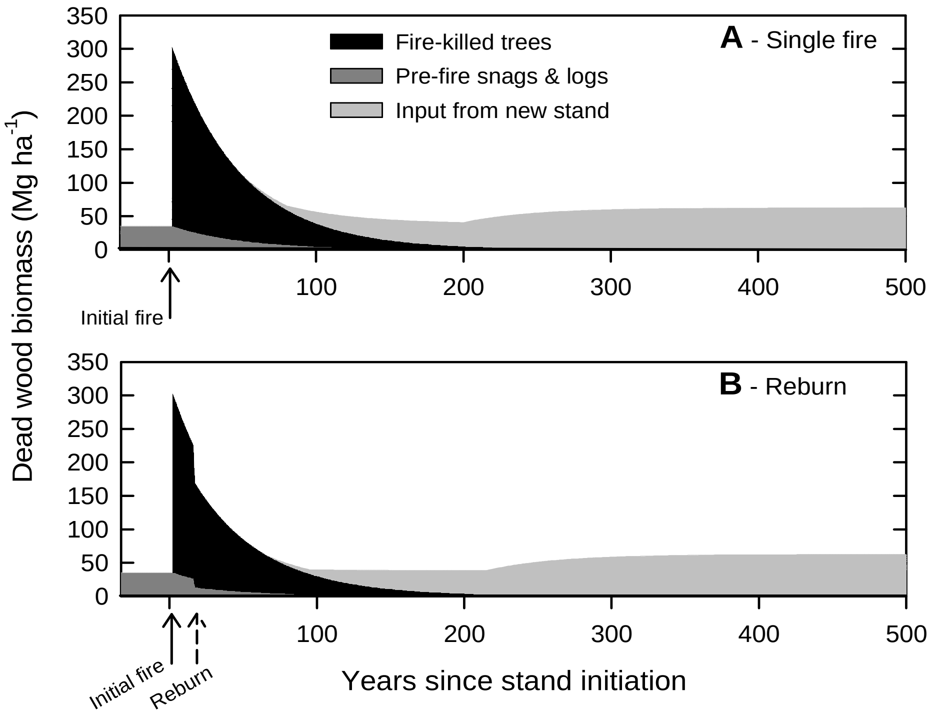

Figure 16. *Projections of dead wood biomass through succession, parameterized with data from this study, region-specific data on biomass mortality rates from our prior studies (Hudiburg et al. 2009), and relevant literature values for wood decomposition rates.*

Prior post-fire logging was associated with altered responses to a subsequent reburn. The most intuitive finding is that more dead wood mass persists through the reburn in the absence of logging (Fig. 17a). Perhaps less intuitive is that the difference is only in snag biomass, and not at all the down wood, which is highly similar regardless of logging history (Fig. 17a). As for live components, unmanaged reburn sites had very similar conifer regeneration densities and hardwood/shrub biomass to single-burn sites (Fig. 17 b-c; see Objective 1 above). However, reburn sites with prior post-fire logging were characterized by a shift toward much greater hardwood and shrub biomass and slightly lower conifer regeneration densities (Fig. 17 b-c). Shrubs in previously logged reburn sites were heavily dominated by *Ceanothus velutinus and Ceanothus integerrimus*.

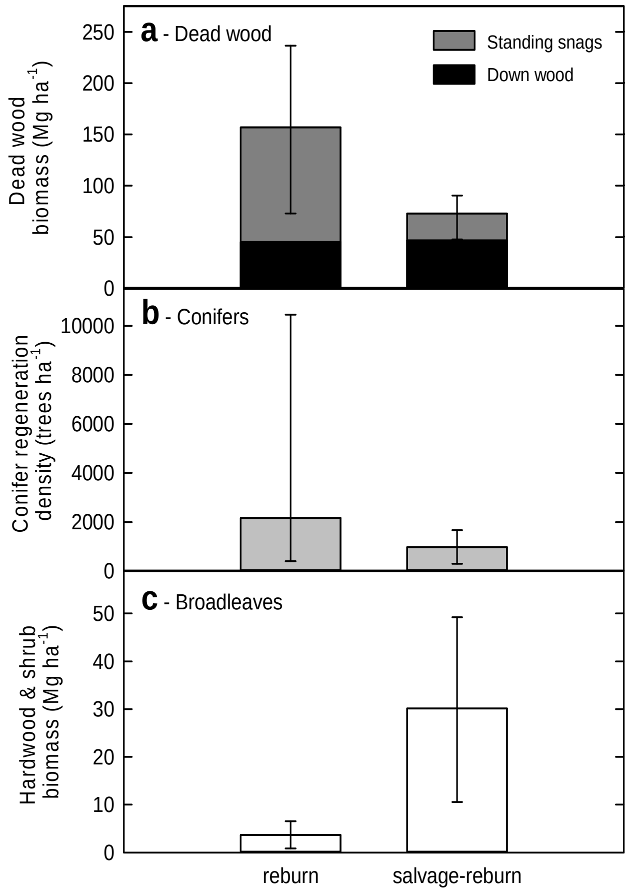

Figure 17. *Key components of stand and community structure following a reburn, as influenced by prior post-fire logging. Data are mean ± 95% CI, except conifer densities which are medians and 25th-75th percentile.*

Management implications

Objective 1

Re-establishment of tree cover is often a prime objective for land management agencies following large, severe disturbances like landscape fires (e.g., USDA 2004). However, funding for tree planting and vegetation management over such broad areas is limited, and many areas within these fires will develop through natural establishment and succession processes. The trends we observed at 10 years post-fire suggest that, in the unmanaged majority of the Biscuit Fire, conifer establishment, survival, and growth are all proceeding rather robustly and meeting or exceeding typical objectives for initial density and occupancy.

As is typical of early-seral 'pre-forest' conifer ecosystems following disturbances, most of the high-severity portions of the Biscuit landscape currently are dominated by shrubs and hardwoods, some of which will also develop into tall-stature trees (e.g., tanoak, madrone, chinquapin, canyon live oak). These vegetation components provide key structures for early-seral associate species (e.g., Fontaine et al. 2009; Swanson et al. 2014)). Surprisingly, at a decade post-fire, interactions with broadleaf vegetation do not appear to be exerting a major influence on conifer establishment and growth. Combined with generally high conifer survival rates overall, this suggests little benefit from intensive vegetation management at this time point. The strongest pattern evident in the 10-year data is the effect of patch size (distance from live trees): beyond ~400 m from live-tree edges, the regenerating community is increasingly dominated by broadleaves and the fire-dependent knobcone pine. Where these are inconsistent with management objectives, these areas could be a priority for tree planting and other vegetation management activities.

Objective 2

Mitigation of early-seral (post-disturbance) fire hazard has long been a key management concern across much of the western U.S. (Brown et al. 2003). Our objective in this re-measurement was to assess whether the initial surface-fuel pulse in harvested stands had converged with accumulating dead-fall in untreated stands by a decade post-fire. For the most part, treated stands still contained higher loadings of woody surface fuels, most strongly for coarse wood particles and less so for fine wood particles which were approaching parity among treatments. Converging trends were apparent at 10 years, but based on field measurements and measured decay rates, we estimate that full convergence (or crossing) of treatments for coarse fuels may not occur for another 1-3 decades, depending on harvest intensity. Notably, moderate-intensity logged stands in some ways behaved more like untreated stands than high-intensity logged stands during the first decade, in that enough wood was left on site to continue accumulating on the surface (although presumably less than unlogged stands at later time periods). This finding suggests moderate-intensity harvest may strike a balance between removal of some fuel mass while providing the habitat role and future inputs of retained wood.

These data suggest a high importance of live fuel mass in determining fire hazard in young post-fire forests. As in much of the Klamath ecoregion, various hardwood trees co-exist with conifers as both canopy sub-dominants and understory shrubs. Coppice re-sprouting after high-severity fire is rapid for most of these species. In the first year following logging, the majority of fine fuels within 2 m of the ground (i.e. surface layer) were composed of dead legacy wood, allowing the discrepancies between treatments attributed to slash generation to be most pronounced. Seven years later, the majority of surface-layer fine fuels were composed of regenerating vegetation and newly produced leaf litter, and spatial cover of live material exceeded 100% (high continuity). Since this shrub-dominated regeneration is progressing independently of logging treatments, future discrepancies between logged and unlogged stands (whether it be continued influence of logging slash in treated stands or developing influence of branch-fall in untreated stands) will likely have a diminishing influence on total surface-layer fine fuels. The preeminence of live vegetation in dictating surface fine fuel loads and continuity following wildfire is likely especially pronounced in forest types with high productivity and abundant re-sprouting. It may also explain why previous studies have found little difference in reburn severity between logged and unlogged sites (Thompson et al., 2007). We do not discount the role of dead woody fuels which certainly can contribute to fire behavior and effects (including firefighting difficulty), but these data suggest that the role of live fuels – especially in mesic productive forests – has been underemphasized in post-fire fuel assessment and management.

Objective 3

Much of the attention to post-fire fuel management is ultimately concerned with potential impacts of a reburn (Brown et al. 2003). Our previous studies indicated unique (but not depauperate) vegetation communities regenerating after a reburn event. Our current studies show that dead wood dynamics are also significantly affected, with stands originating from a reburn sequence carrying approximately half the dead wood as those originating from a single stand-replacement fire. For the management of dead wood, this finding points to the challenge of managing for consistent target levels within a highly variable mixed-severity fire regime. Stochastic variations in disturbance interval and severity, of which the reburn we measured is but one, result in wide (and mostly unpredictable) variations in dead wood quantities over time and space. Management targets will be most successful if they are flexible and can incorporate such wide variations that likely have always characterized these forest types.

Vegetation response to a reburn appears to be influenced by whether salvage occurred after the first fire. The strongest effect was a major shift toward much greater abundance and dominance of shrubs, particularly of the genus *Ceanothus* but also *Arbutus* and *Lithocarpus*. The mechanisms behind this shift are not fully resolved, but it's plausible that the additional soil disturbance of prior salvage activities may have affected the soil seedbank for the relevant species. It is possible that the N-fixing capacity of this *Ceanothus*-dominated community may act to compensate for some of the nitrogen losses from repeated fires (see Bormann et al. 2008). Overall these findings suggest that salvage-reburn sites may be candidates for more intensive vegetation management, including planting to augment natural conifer regeneration and/or management of extremely dense shrub fields if these components are inconsistent with objectives.

Relationship to other recent findings

Our finding of continual conifer establishment for at least a decade after wildfire is consistent with those of Shatford et al. (2007) for the Klamath region, and those of Freund et al. (2014) for other Pacific Northwest Douglas-fir forests. Those studies reported establishment periods of ~20-60 years, suggesting we may have so far quantified the first portion of a long process. At first glance, the generally robust establishment densities, survival, and growth of conifers we observed seem at odds with a large body of literature on notorious reforestation difficulties in southwest Oregon (e.g., Hobbs et al. 1992). However, those studies, in addition to being conducted following timber harvests rather than wildfires, emphasized the most difficult sites to regenerate (drier lower-elevation slopes), while the expansive Biscuit Fire comprised a much wider range of sites from high-elevation wet to low-elevation dry environments. The lack of relationship between broadleaf and coniferous vegetation abundance or stature was surprising given the well-known 'tug-of-war' between these components in this region (Whitaker 1960; Hobbs et al. 1992), but is consistent with recent studies (Shatford et al. 2007; Irvine et al. 2009), suggesting that facilitative interactions such as mycorrhizal associations, shade provision, and N-fixation may be as important as competitive effects. Our finding of greater proportional representation of a serotinous species in the interior of the largest burn patches is consistent with recent findings from the northern Rocky Mountains (Harvey et al., in review).

The results of this longitudinal study support a consistent narrative for dead woody fuels: an initial logging-induced pulse of surface material, followed by later convergence and crossing of logged and unlogged stands, with differences at both early and later time points scaling to logging intensity (Donato et al., 2006a, 2006b, 2013; McIver and Ottmar, 2007; Monsanto and Agee, 2008; Keyser et al., 2009; Ritchie et al., 2013; Dunn and Bailey 2015; Peterson et al., 2015). Ten years after the Biscuit Fire, convergence among treatments is apparent but not yet complete. Other empirical studies on surface fuel accumulation following postfire logging suggest that downed woody fuels in unlogged stands eventually surpass levels in logged stands after ~5 to 20 years of inputs via fall and fragmentation of fire-killed trees (McIver and Ottmar, 2007; Monsanto and Agee, 2008; Keyser et al., 2009; Peterson et al., 2015). With respect to fine wood fuels (1-100 hr), it appears to have taken ten years of tree fall and fragmentation for unlogged stands to achieve statistical parity with high-intensity logged stands on the Biscuit Fire, but will take a longer time to achieve parity with moderate-intensity logged stands that are still experiencing some surface accumulation. Coarse woody fuels (>100hr) remain higher in logged stands seven years after treatment (10 years post-fire) but are on a trajectory similar to that of fine woody fuels. Thus, the timing of fine woody fuel dynamics appears consistent with other studies, while the timing of coarse woody fuel convergence appears to be longer than reported in other studies. For example, the proportion of snag biomass reaching the surface by 10 years post-fire (~10-15%) is much lower in this Douglas-fir-dominated system than the fast accumulation rates reported for drier, pine-dominated forests (e.g., ~80% in 8 years, Ritchie et al. 2013).

The fire hazard posed by live fuels appears to grow to critical levels of mass and continuity by 10 years, regardless of postfire logging. Once regenerating shrubs approach 100% surface cover, fire burning under moderate to extreme weather conditions will likely result in near complete aboveground mortality of any vegetation in this canopy stratum, which for at least decades includes all regenerating conifers. Both McIver and Ottmar (2007) and McGinnis et al. (2010) similarly concluded that differences in fuel loading between logged and unlogged stands matter little to the risk of stand-replacement since both would experience complete mortality in an early-seral reburn. Similar live fuel structure may also explain why previous studies have found little difference in reburn severity between salvage-logged and unlogged sites (Thompson et al. 2007).

The decomposition rates of fire-killed wood observed in this study fall comfortably within that reported by other studies in the Pacific Northwest (Sollins 1982; Harmon et al. 1986; Janisch et al. 2005; Dunn and Bailey 2012); though this has as more to do with the range of reported values than it does with the accuracy of any one study, including this one. More important than the absolute rates (which depend on assumptions regarding pre-fire green wood density) is our observation that branches and boles decayed at similar rates whether standing or downed, and boles decayed only somewhat slower when standing than when downed. Assumptions that standing dead wood decays no differently than fallen wood can lead to an overestimation of ecosystem-wide wood decay and underestimation of legacy residence times; but by the same token, assumptions that aerial decay of wood is negligible necessarily lead to reciprocal biases and over-prediction of accumulating surface woody fuels (Harmon et al. 2011a, 2011b; Dunn and Bailey 2012, 2015).

As for the reburn impacts on dead wood we observed, there are few other studies with which to compare. Reductions in dead wood abundance with repeated non-lethal surface fires, such as prescribed burning, have been well documented (e.g., Aponte et al. 2014). For stand-replacing fires, one study reported that a reburn reduced dead wood to negligible levels (near 100% reduction relative to a single stand-replacing fire) in spruce forests of boreal North America (Brown and Johnstone 2011). The lower levels of snag mass we observed in the reburn are consistent with observations of Holden et al. (2006) from reburns in southwestern *Pinus ponderosa* forests. The dead-wood consumption levels we estimate for the reburn (~25%) are similar to rates reported for dead-wood consumption in single stand-replacing fires (Tinker and Knight 2000; Campbell et al. 2007; Donato et al. 2013), suggesting that, in this respect, reburns are effectively a repetition of single-burn effects rather than fundamentally different or multiplicative. Our findings did not support the expectation that charring of wood results in case-hardening and lower fall/decay rates (Holden et al. 2006; Hyde et al. 2011), as the highly charred snags in the reburn fragmented and fell at similar rates to snags outside of the reburn. Our findings on the influence of salvage logging on subsequent response to a reburn are somewhat similar to one of the few other relevant studies (D'Amato et al. 2011). They reported that blowdown sites that were salvaged then burned in a wildfire were lower in structural legacies (e.g., dead wood) and community heterogeneity than blowdown-fire sites with no intervening salvage. However, our findings differ from those of D'Amato et al. (2011) in that salvage effects on vegetation were apparently amplified, not diminished, when salvage sites were disturbed again.

Future work needed

These findings suggest several important lines of future inquiry. One is continued surveys of conifer establishment rates over time. Protracted regeneration periods differ from the traditional models of forest development in the Pacific Northwest, and, by extending the time before canopy closure occurs (Donato et al. 2012), sustain an early-seral period that is increasingly recognized as important for landscape biodiversity (Swanson et al. 2011, 2014; Campbell et al. 2014). Understanding the timeframes and factors associated with eventual tree dominance will provide a critical foundation for the emerging focus on provision of early-seral habitat (Franklin and Johnson 2012).

Because surface fuel loads have not completely converged at 10 years, especially for coarse fuels for which treatment-comparisons have barely changed since year 3, there is still uncertainty as to when convergence (or potential crossing) may occur. Snag dynamics in the Klamath region appear to differ from those reported in other regions (e.g., McIver and Ottmar, 2007; Monsanto and Agee, 2008; Keyser et al., 2009; Ritchie et al., 2013; Dunn and Bailey 2015; Peterson et al., 2015), suggesting a need to better parameterize snag-fall and decay rates for individual regions. Our data on snag decay rates represent a start to arriving on constraints for those parameters, but more comprehensive work is needed.

Most research and management of post-fire fuels has heavily focused on the dead wood component. The fuel profiles we observed 10 years after the Biscuit Fire strongly suggest a need to better understand the role of live regenerating fuels to overall fire potentials. The degree to which the live component can be managed to reduce fire potential (e.g., beta diversity in structure among stands, shaded fuel breaks, etc.) is rather poorly described for most early-seral forests, but may be at least as important as managing dead fuels.

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Deliverables

Table 1. Deliverables crosswalk table.

Deliverable Type	Description	Delivery Status
Peer-reviewed publication	Post-fire management effects on 10-year vegetation and fuel succession patterns	Article in review at journal: Campbell et al., "Dynamics of live and dead fuels with and without post-fire logging ten years after the Biscuit Fire (Oregon, USA)," submitted to *International Journal of Wildland Fire*.
Peer-reviewed publication	10-year vegetation succession patterns following a re-burn with and without post-fire management	Article in review at journal: Donato et al., "Burning the legacy? Influence of wildfire reburn on dead wood dynamics in a temperate forest," submitted to *Ecosphere*. Second article in prep: Fontaine et al. "Prior post-fire logging influences vegetation response to a reburn in a mixed-evergreen forest," for submission to *Canadian Journal of Forest Research*.
Peer-reviewed publication	Role of fire-killed trees in post-fire carbon emissions to the atmosphere, elucidated by a decade of decomposition	Article in review at journal: Campbell et al., "Biogenic carbon emissions from fire-killed trees following a large wildfire in Oregon, USA," submitted to *Journal of Geophysical Research - Atmospheres*
Peer-reviewed publication	Vegetation succession pathways as influenced by environmental gradients within a large wildfire	Analysis completed, article in preparation for submission to journal: Donato et al., "Tree regeneration dynamics a decade after a large landscape fire in a mixed-evergreen forest," for submission to *Forest Ecology & Management*.
Peer-reviewed Synthesis	Synthesis articles on early successional pathways and post-fire management effects	Two articles published: Donato et al. 2012. "Multiple successional pathways and precocity in forest development: Can some forests be born complex?" *Journal of Vegetation Science* 23: 576-584. Campbell and Donato 2014. "Trait-based approaches to linking vegetation and food webs in early-seral forests of the Pacific Northwest." *Forest Ecology and Management* 324: 172-178.

		Another article is in discussion, a collaboration with another JFSP re-measurement grant (Monica Turner et al.) to compare long-term lessons from mega-fires between the Rocky Mountains and Klamath region.
Management outing(s) / workshop(s)	Collaborating, demonstrating, and discussing our findings with agency land managers	We have met with and been in the field with RSNF silviculturist and fire personnel on multiple occasions (e.g., coordinated regarding field sites, collaborated on snag-fall study). Our scheduled post-study field outing has been postponed due to exceptional fire season. Sharing of findings is ongoing and will continue. We also co-organized an international symposium on post-disturbance ecology and management in Germany (June 2014). Participants attended from the western and eastern USA, northern Europe, central Europe, and Australia. The symposium included presentations and outings with ecologists and natural resource managers to discuss knowledge frontiers and uncertainties after major stand-replacing disturbances.
Website	Study approaches, results, and publications will be made available on a searchable website	Website constructed and live during the project period (https://sites.google.com/site/ecologyofdisturbance/). Site is currently offline for maintenance and will be re-posted when findings are published in journals.
Conference presentations	Present at national/international conferences	We delivered seven invited or contributed oral presentations: Campbell et al. 2012. Occurrence of wildfire reburns in forests of the Pacific Northwest. Northwest Scientific Annual Meeting, Boise, Idaho. Campbell et al. 2012. Naturally regenerating forests in the Pacific Northwest. Ecological Society of America Meeting, Portland, Oregon. Donato. 2014. Managing after large, severe disturbances in western North America. Czech University of Life Sciences, Prague, Czech Republic. Campbell et al. 2014. Moving beyond structure to function in post-disturbance forest communities. International Symposium on Post-Disturbance Ecology & Management, Bavarian NP, Germany.

		Fontaine et al. 2014. Managing post-disturbance ecosystem structure for resilience to future disturbances. International Symposium on Post-Disturbance Ecology & Management, Bavarian NP, Germany. Donato et al. 2014. Understanding structural complexity in post-disturbance forest communities: What's next? International Symposium on Post-Disturbance Ecology & Management, Bavarian NP, Germany. Donato et al. 2015. Ten years after the Biscuit Fire's 15 minutes: fuels, regeneration, and post-fire management. USFS Symposium on Disturbance and Salvage Logging, Powdermill, Pennsylvania.

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