Burning the legacy? Influence of wildfire reburn on dead wood dynamics in a temperate conifer forest

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Abstract. Dynamics of dead wood, a key component of forest structure, are not well described for mixed-severity fire regimes with widely varying fire intervals. A prominent form of such variation is when two stand-replacing fires occur in rapid succession, commonly termed an early-seral "reburn." These events are thought to strongly influence dead wood abundance in a regenerating forest, but this hypothesis has scarcely been tested. We measured dead wood following two overlapping wildfires in conifer-dominated forests of the Klamath Mountains, Oregon (USA), to assess whether reburning (15-yr interval, with >90% vegetation mortality) resulted in lower dead wood abundance and altered character relative to once-burned stands, and how any differences may project through succession. Total dead wood mass (standing + down) following the reburn (169 ± 83 Mg/ha [95%CI]) was 45% lower than after a single fire (309 ± 87 Mg/ha). Lower levels in reburn stands were due to, in roughly equal parts, additional combustion and greater time for decay. Although a single fire in mature forest both consumed and created dead wood (by killing large live trees), a reburn only consumed dead wood (few large live trees to kill). Charred biomass (black carbon generation) was higher in reburned stands by a factor of 2 for logs and 8 for snags. Projecting these stands forward (notwithstanding future disturbances) suggests: (1) the near-halving of dead-wood mass in reburn stands will persist for ~50 yr until the recruitment of new material begins, and (2) the reburn signature on dead wood abundance will remain apparent for over a century. These findings demonstrate how a single stochastic variation in disturbance interval can impart lasting influence on dead-wood succession, reinforcing the notion that many temperate forests exist in a state of dead-wood disequilibrium governed by site-specific disturbance history. Accounting for such variation in disturbance impacts is crucial to better understanding forests with complex mixed-severity disturbance regimes and with increasing stochasticity under climatic change.

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Introduction

Among the most universally applied conceptual models in forest ecology are those describing the succession of dead wood over a forest sere (e.g., Harmon et al. 1986, Spies et al. 1988, Bormann and Likens 1994, Sturtevant et al. 1997, Pregitzer and Euskirchen 2004, Hall et al. 2006, Bradford et al. 2009, Kashian et al. 2013). Changes in dead wood over time influence

several key ecosystem functions including nutrient cycling, wildlife habitat, fuel for wildfires, and carbon storage (Harmon et al. 1986). Dead wood models often describe a more or less "U-shaped" dynamic over successional time, with a pulse of dead wood from a stand-initiating disturbance, decay of this pool over time, and recruitment of new dead wood via mortality in the next stand. This parsimonious framework has proven relevant across a diversity of ecosystems characterized by infrequent stand-replacing disturbances. A key uncertainty, however, is the degree to which stochastic variations in disturbance frequency may cause deviations from expected trends in dead wood abundance.

Understanding the influence of disturbance variability is crucial not only in light of changing climate and disturbances (Turner 2010), but because our knowledge of forest disturbance regimes has evolved since the foundational dead-wood models were described. For example, the systems from which those models (Harmon et al. 1986, Spies et al. 1988) were derived, the temperate coniferous forests of the Pacific Northwest (PNW), were traditionally thought to develop along a linear trajectory in between infrequent, singular stand-replacing disturbances (e.g., Munger 1930, Franklin et al. 2002). However, it is becoming clear that portions of these forests, like many across the temperate zone (e.g., Collins et al. 2009, Marcoux et al. 2013), are influenced by more complex disturbance regimes that include intermediate, mixed-severity, or temporally clustered disturbances (Donato et al. 2009b, Halofsky et al. 2011, Perry et al. 2011, Tepley et al. 2013). A prominent example is when a young forest regenerating after a stand-replacing fire is burned by a second stand-replacing fire early in succession (i.e., early-seral "reburn;" *cf.* Thompson et al. 2007, Collins et al. 2009, Donato et al. 2009b, Holden et al. 2010, Miller et al. 2012, Teske et al. 2012). Fire risk, spread potential, and mortality susceptibility are all high early in succession in many forest types (Martin et al. 1974, McIver and Ottmar 2007, Odion et al. 2010, Thompson et al. 2011, Stephens et al. 2012), making reburns a potentially important disturbance factor. Indeed, one of the earliest dead-wood succession papers (Spies et al. 1988) hypothesized that reburns could be a key driver significantly reducing dead wood abundance far into the next successional cycle. To date, this hypothesis remains largely unexplored.

Reburns may consume a high proportion of ecosystem wood mass compared to single stand-replacing fires. Wood mass in mature forests is dominated by live boles which experience little combustion (Campbell et al. 2007), whereas wood in early-seral stands is dominated by disturbance-killed trees in various states of decay and surface accumulation and thus more available to combustion (Passovoy and Fulé 2006, Hyde et al. 2011). Modeling exercises have suggested reburns as a possible explanation for lower-than-expected dead wood levels in forests of a given age across a large landscape (Nonaka et al. 2007). Two successive fires were reported to diminish dead wood abundance to negligible levels in boreal forests (Brown and Johnstone 2011). Repeated low-severity surface fires diminish dead wood stocks within mature forests (Aponte et al. 2014), but how this translates to stand-replacing early-seral fires remains unclear. Reburns may also char a greater amount of dead wood, which may affect key ecosystem attributes such as long-term black (stable) carbon storage and wildlife habitat (Bradbury 2006, Singh et al. 2012). These and other potential impacts have made reburns an increasingly prominent management concern (McIver and Starr 2001, Brown et al. 2003, USDA 2004, Peterson et al. 2009, Thompson and Spies 2010, Donato et al. 2013). However, few data exist to inform the ecological context of reburn as it affects dead wood.

The Klamath Mountains of the U.S. west coast provide a key opportunity to explore these questions. The area constitutes the southern portion of the PNW Douglas-fir region, which has profoundly shaped concepts in forest ecology globally, including foundational studies of dead wood succession (e.g., Spies et al. 1988). The Klamath Mountains best exemplify the more complex disturbance regimes that characterize parts of the Douglas-fir province west of the Cascade Range crest (and other similar temperate forests). Forests of the Klamath region are high in productivity and biomass, yet fire-prone and subject to highly variable mixed-severity fire regimes (Agee 1993, Halofsky et al. 2011, Perry et al. 2011). This combination suggests reburns to be an important disturbance regime component as much as anywhere in the temperate zone.

In this study we use field data from a natural experiment afforded by two overlapping wildfires with a 15-yr reburn interval to evaluate the effects of early-seral reburn on short- and long-term dead dynamics. Rather than comparing dead wood patterns among different disturbance regimes such as surface fires vs. stand-replacing fires—as in previous work (e.g., Agee 2002, Wright et al. 2002)—our objective was to quantify the effects of an important stochastic variation within a given fire regime. We addressed the following questions: (1) How much dead wood remains following a reburn relative to after a single fire, and does reburn result in different rates of wood charring and snag fall/fragmentation? (2) How is the classic U- or reverse J-shaped model altered by severe reburn? (i.e., does a reburn leave stands with a substantially altered dead wood legacy into the future); and (3) How do reburn effects compare to other processes influencing dead wood abundance (e.g., variations in mortality and decay rates associated with local climatic variation)?

Methods

Study area

The study was conducted in the Klamath Mountains, a coastal range in northwestern USA (latitude/longitude 42°26′ N, 123°49′ W). Vegetation is representative of the cool/moist *Abies concolor* zone (Franklin and Dyrness 1973), growing on steep slopes; shallow soils derived from metamorphic and igneous parent materials; and influenced by warm, dry summers and cool, moist winters (annual precipitation ranges from 1500 to 3000 mm). Like most coastal PNW forests, the sites were high in aboveground biomass, typically ranging from ~400 to 700 Mg/ha (Donato et al. 2013). Fire regimes are typical of the mid- to higher elevations of the Klamath Mountains as well as the central-southern Cascade Range—mixed-severity regimes with fire intervals ranging from ~20 to >150 yr, and stand-replacing patches occurring as part of a burn mosaic (Agee 1993, Wills and Stuart 1994, Taylor and Skinner 1998, Stuart and Salazar 2000, USDA 2004, Perry et al. 2011, Tepley et al. 2013).

We studied areas burned by the 1987 Silver Fire and the partly overlapping 2002 Biscuit Fire (USDA 2004, Donato et al. 2009*b*, Thompson and Spies 2010). Each fire was mixed-severity, but given our study objectives, we sampled high-severity patches only (defined here as patches greater than one hectare incurring ≥90% overstory mortality). Sampled disturbance histories (Fig. 1) were: reburn (twice-burned with stand-replacement within a 15-yr interval, 1987–2002), once burned (single stand-replacing fire in 2002 after a > 100-yr interval), and unburned (no recent fire, sampled immediately adjacent to study fires). The 2002 burn in our study sites occurred during a 9-d period of relatively extreme weather conditions when 50% (102,273 ha) of the Biscuit Fire area burned: daytime temperatures of 26.6 °C, 32.4% relative humidity, and 12.5 km/h winds (Thompson and Spies 2009). Our study stands are described in detail and mapped by Donato et al. (2009*b*). Study stands were verified to the extent possible to be of similar abiotic and biotic character prior to recent disturbances (Donato 2008, Donato et al. 2009*b*). Specifically, all three conditions averaged between 19 and 25 m^2/ha of basal area, between 226 and 256 trees/ha (of stems >20 cm dbh), and were (formerly) mature to old-growth forest >100 yr old dominated by Douglas-fir (*Pseudotsuga menziesii*) with components of white fir (*Abies concolor*), sugar pine (*Pinus lambertiana*), and several evergreen hardwoods (*Lithocarpus densiflorus, Arbutus menziesii, Chrysolepis chrysophylla*). All stands were on 20–40° slopes on westerly, northerly, and easterly aspects; mid- to upper slope positions; elevations of 700–1150 m; and nonserpentine soils (Donato et al. 2009*b*). No stands had experienced prior timber harvest based on stump absence and agency spatial data.

This study took advantage of an existing arrangement of two superimposed wildfires. Like most natural experiments (see Fulé et al. 2004, Johnstone 2006), detailed prefire data were unavailable, preexisting differences were minimized but likely not absent, and random assignment of treatments was not possible. Statistical inference in the strict sense thus applies to our study area. We interpret our results in this context.

Data collection and processing

We sampled dead wood biomass 2 yr after the Biscuit Fire (17 yr after the initial Silver

Fig. 1. Examples of mature forest stands that have experienced no recent fire (a), one stand-replacement fire (b), and two stand-replacement fires within 15 yr—that is, early-seral reburn (c). Photos by J.B. Fontaine and D.C. Donato.

Fire). Later re-sampling was precluded by postfire (salvage) logging in most of our single-burn stands after our first sampling occurred (USDA

2004). A one-hectare plot (based on USDA Forest Inventory and Analysis protocols (USDA 2003*b*)) was randomly placed (azimuth and distance from arbitrary point within stand determined from random compass spin and timepiece second hand, respectively) in each of 18 stands: six re-burned, six once-burned, and six unburned stands immediately outside the Biscuit Fire perimeter. Standing snags were measured in four regularly spaced circular subplots at varying spatial scales to account for variable stem densities (objective criterion of ~60 trees sampled per plot), with a default radius of 10 m and range of 6–16 m (*per* Law et al. 2001, Campbell et al. 2004, Irvine et al. 2007, Meigs et al. 2009, Donato et al. 2013). We recorded tree species, diameter at breast height (dbh; 1.37 m above ground), total height, decay class (1–5, see Sollins 1982), broken-topped (y/n), bole fragmentation, and percent of stem surface with deep wood char. We sampled coarse woody detritus (CWD; >7.62 cm diameter) along four 75-meter planar intercept transects (Harmon and Sexton 1996) radiating from plot center in subcardinal (NW, NE, SW, SE) directions. Each intersecting piece was recorded for diameter, species, decay class, and deep wood charring (Donato et al. 2009*a*). The total sample included 1268 snags and 752 logs, the latter sampled along 5400 m of planar intercept transect.

We computed stem mass for all trees using species- and region-specific allometric volume equations and wood densities (Means et al. 1994, Van Tuyl et al. 2005). Snag biomass estimates were adjusted for individual-specific metrics of wood charring, top breakage, and decay class (Harmon and Sexton 1996, Donato et al. 2013), as well as plot-specific ocular estimates of branch consumption (*sensu* USDA 2003*a*). We computed mass of down woody detritus from planar-intercept data using geometric scaling and species- and decay-specific wood density values (Harmon and Sexton 1996). Down wood was corrected on an individual piece basis for mass loss due to charring (Donato et al. 2009*a*). For both standing and down wood, we assigned decay classes 1–3 as "sound" and classes 4–5 as "decayed."

Data analysis

We statistically evaluated dead wood characteristics among disturbance histories by comparing 95% confidence intervals (CI), whereby

intervals mutually excluding other groups' means indicate significant differences (Ramsey and Schafer 2002). Comparisons were made for total dead wood pools as well as standing and down components separately, sound and decayed pools, and the percentage of dead wood encased in deep char.

The role of reburn in long-term dead wood dynamics could be explored by additional comparisons between once- and twice-burned stands at a second time point, or through modeling exercises. With the former option precluded by intervening management treatments, we took two complementary approaches to assessing longer-term dynamics. The first was to revisit the reburn stands in 2012 (10 yr post reburn) to repeat the tree survey and compute snag-fall and fragmentation rates. A rate constant describing whole-snag fall (for trees >20 cm dbh, due to low initial numbers of small stems) was calculated as $k = -\ln(C_t/C_o)/t$, where C_o = plot-wide count of standing dead stems measured at time 1, C_t = plot-wide count of standing dead stems measured at time 2, and t = 8 yr. A rate constant describing fragmentation of snags (>20 cm dbh) was calculated as $k = -\ln(M_t/M_o)/t$, where M_o = proportion of boles still standing but fragmented at time 1 (accounting for breakage levels at that time point) and M_t = proportion of boles still standing but fragmented at time 2 (accounting for breakage levels at that time point), and t = 8 yr. We then compared these rates of snag fall and fragmentation with those from a related study of snag dynamics in high-severity portions of the Biscuit Fire outside the reburn area (Campbell et al. *in press*).

The second approach was to combine our data with established values of wood decay rates and fire consumption from our previous studies in the Klamath region and other relevant studies (Sollins 1982, Harmon et al. 1986, Spies et al. 1988, Campbell et al. 2007, Hudiburg et al. 2009, Donato et al. 2013). For this exercise, we explored the relative contribution of decay and combustion in explaining any differences in dead wood abundance in reburned stands—a necessary evaluation because the reburn had experienced not only two fires, but also had a 15-yr head start on wood decay. To do so, we compared the observed quantity after reburn to the predicted quantity had no second fire occurred. The predicted quantity

was derived by subtracting 15 years' worth of decomposition loss starting from the dead wood pool after a single fire. The difference between the observed and predicted amounts (i.e., the loss which could not be explained by decomposition alone) provides an estimate of combustion effects. We accounted for uncertainty using a Monte Carlo framework, with 1000 iterations of randomly selected values from the 95% CI for the post reburn dead wood pool and a published range of plausible decay rates (single exponential decay constant [k] of 0.013 to 0.030; Sollins 1982, Harmon et al. 1986, Spies et al. 1988, Donato et al. 2013). The resulting estimate of reburn consumption was then compared to published estimates of consumption by single fires in our study region (Campbell et al. 2007, Donato et al. 2013).

Finally, we incorporated our observed reburn effects into a classic model of long-term dead wood dynamics (*sensu* Spies et al. 1988), by projecting our postfire data through successional time and comparing single and reburn stands. The model was structured with a starting point of the dead wood quantities we measured postfire, which subsequently decays over time, plus recruitment (and decay) of new dead wood as the stand ages. In addition to our measured data, therefore, two additional parameters were necessary: recruitment rate of new dead wood through forest succession, and decay rate of all wood. For the former, we used stand-age-specific estimates of biomass accumulation and dead wood recruitment (biomass mortality rate) for the Klamath region, from our previous studies (Hudiburg et al. 2009). Those values were 0.6 Mg·ha^{-1}·yr^{-1} for stands 50–80 yr old, 1.6 Mg·ha^{-1} yr^{-1} for stand 80–200 yr old, and 2.6 Mg·ha^{-1}·yr^{-1} for stands >200 yr old. For the latter, we applied a central estimate of $k = 0.021$ from the range of literature values described above (Sollins 1982, Harmon et al. 1986, Spies et al. 1988, Donato et al. 2013). We accounted for uncertainty in both of these inputs by exploring how the curves vary with adjustments of ±25% to each parameter (*sensu* Nonaka et al. 2007). This approach, in addition to yielding a sensitivity analysis regarding input assumptions, allowed comparison of the magnitude of reburn effects relative to that associated with other factors that drive dead wood recruitment and decay (e.g., local climatic variation) (Kennedy et al. 2008).

RESULTS

Dead wood abundance and characteristics

Mean total dead wood biomass in reburned stands (169 Mg/ha) was approximately half that in once-burned stands (309 Mg/ha) (Fig. 2).
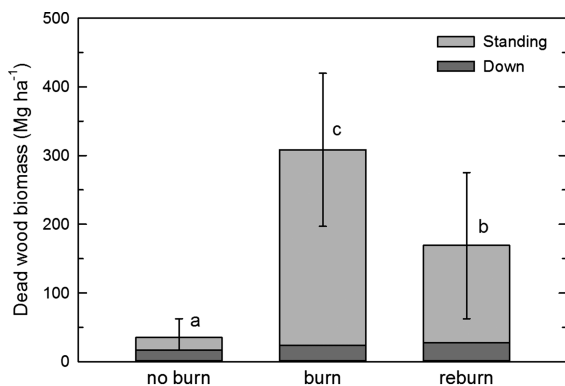

Fig. 2. Total dead wood biomass (mean, 95% CI) in stands of different recent disturbance histories. Confidence intervals are for the total dead wood pool. Lower-case letters above bars indicate whether confidence intervals mutually exclude other groups' means. Reburned stands contained significantly lower dead wood biomass than once-burned stands.

While dead wood biomass increased from unburned to once-burned stands (a net generating event for dead wood), it decreased from once-burned to reburned stands (dead wood-consuming event) (Fig. 2). Most of the difference between reburned and once-burned stands was in the snag component, while down wood pools were similar (Figs. 2 and 3). In terms of physical character, both standing and down wood pools were charred to a much greater degree in reburn stands (Fig. 3). The reburn also contained greater quantities of decayed wood, in both the standing and down pools.

Lower dead wood abundance in reburned stands resulted from approximately equal contributions of: (1) having a longer time for decay since the first fire; and (2) combustion in the second fire. Applying known decomposition rates to the 15 yr of extra decay time yields a predicted quantity of 200–254 Mg/ha in postfire stands had they not reburned, or 18–35% (mean 26%) below the level immediately following a single fire (309 Mg/ha). Measured mean dead wood quantity in reburn stands (169 Mg/ha) was below this predicted range. The estimated reburn combustion, from the Monte Carlo analysis combining variability in both the predicted and measured quantities, yields an estimated reburn
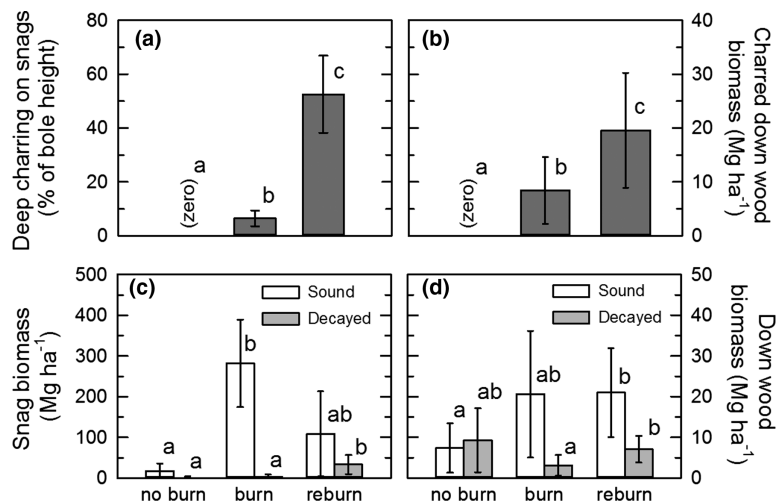

Fig. 3. Characteristics of dead wood biomass (mean, 95% CI) in stands of different recent disturbance histories. Lower-case letters above bars indicate whether confidence intervals mutually exclude other groups' means (assessed separately by pool). Reburned stands were characterized by much higher rates of wood charring (panels a and b), greater decay levels (panels c and d), and a lower proportion of dead wood biomass held in snags (panels c and d) relative to once-burned stands.

consumption of an additional 59 Mg/ha (uncertainty range 18–99 Mg/ha), accounting for another 19% (6–32%) reduction from the 309 Mg/ha present in single-burn stands.

Rates of whole-snag fall in reburn stands averaged k = 0.003 (95% CI, 0.000–0.006), not different than those in high-severity portions of the Biscuit Fire outside the reburn (k = 0.005, 95% CI, 0.000–0.012; Campbell et al. *in press*). Rates of snag fragmentation averaged k = 0.024 (95% CI, 0.017–0.032), also similar to rates outside the reburn (k = 0.019, 95% CI, 0.015–0.023; Campbell et al. *in press*).

Reburns and long-term stand development

During the first century of stand development, the magnitude of the dead wood difference in reburned stands was similar to that associated with an increase of 25% in background rates of wood decomposition (Fig. 4). In contrast, the reburn difference exceeded the range associated with a 25% variation in biomass mortality rate, but only until the new stand begins recruiting significant new dead wood at ~80–100 yr (Fig. 4).

Re-parameterizing the dead wood succession model with our empirical reburn data (Fig. 5) projects that reburned stands will contain lower dead wood biomass relative to once-burned stands for several decades. The initial 45% difference is maintained approximately through the first 50 yr of stand development, until the new stand begins recruiting new dead wood through mortality processes (Table 1). The difference would attenuate after 100–150 yr (Table 1), under the assumption of no intermediate disturbances in the new stand (i.e., isolating the effect of a single reburn).

Discussion

Dead wood abundance and characteristics

These results suggest that stands originating after a reburn sequence will carry approximately half the dead wood legacy than those originating after a single fire. While intuitive qualitatively, actual quantifications of this effect have scarcely been reported. Reductions in dead wood abundance with repeated nonlethal surface fires, such as prescribed burning, have been well documented (e.g., Aponte et al. 2014). As for stand-replacing fires, other studies with which to compare our findings are limited, but one study reported that a short-interval fire 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 mechanism behind the lower wood abundance in the reburn was clearly two-fold—greater decay time and combustion in a second fire. The apparent combustion influence—that is, the difference that could not be explained by decomposition alone—had a central estimate of 59 Mg/ha, or 26% of the biomass estimated to be present just before the stand reburned (central estimate of 226 Mg/ha). This low combustion of wood that was dead at the time of fire agrees well with the
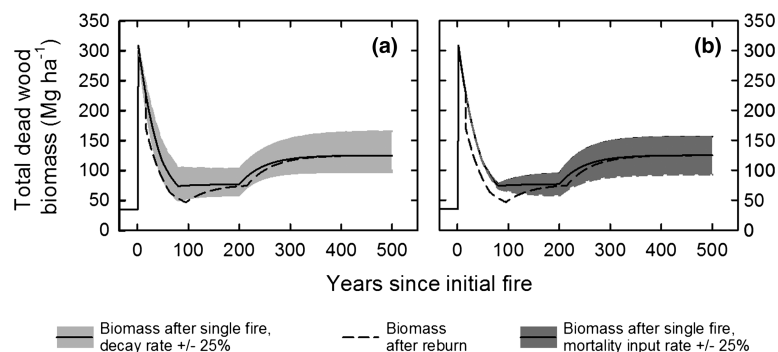

Fig. 4. Evaluation of reburn impacts on long-term dead wood succession in the context of variation in other key drivers. Reburn effects were of similar magnitude to a 25% increase in the wood decay rate (a). Reburn effects exceeded those associated with a 25% change in biomass mortality rates at first (i.e., in young stands when new dead-wood recruitment is negligible), until the stand begins recruiting new dead wood at ~80–100 yr (b).
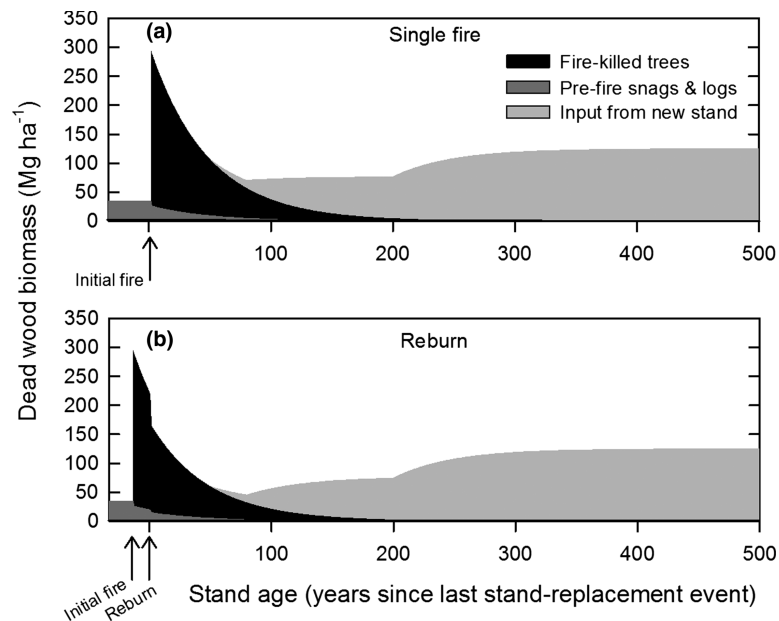
Fig. 5. Projections of dead wood biomass through succession (notwithstanding future disturbances), parameterized with data from this study, region-specific data on biomass mortality rates from our prior studies (Hudiburg et al. 2009), and relevant literature values for wood decomposition rates (Sollins 1982, Harmon et al. 1986, Spies et al. 1988, Donato et al. 2013). Stands originating after a reburn sequence are projected to carry less dead wood for >100 yr of stand development.

Table 1. Estimated dead wood biomass (mean ± 95% CI) at major time points of stand age, defined as years since most recent stand-replacing fire.

Stand age†	Single burn (Mg/ha)	Reburn (Mg/ha)	% difference in reburn vs. single burn
25	190.4 (±68.7)	104.4 (±65.6)	−45
50	113.2 (±40.7)	62.3 (±38.8)	−45
100	74.9 (±14.2)	57.1 (±13.6)	−24
150	76.3 (±5.0)	70.0 (±4.8)	−8
200	108.2 (±1.7)	74.6 (±1.7)	−3
250	119.2 (±0.6)	107.4 (±0.6)	−1

† Years since most recent fire.

<30% values reported in studies of dead-wood consumption by single wildfires (Tinker and Knight 2000, Campbell et al. 2007, Donato et al. 2013). As such, the combustion influence of reburns on dead wood may be characterized as a simple repetition of single-fire effects rather than uniquely high combustion in a reburn.

Short-interval reburns differ fundamentally from long-interval stand-replacing fires, in that they can only be a consuming event for large dead wood, whereas a single severe fire is most often a net generating event for dead wood. Single fires burn through mature forests dominated by live tree boles which, due to their high moisture level, rarely experience significant combustion even when all trees are fire-killed (Campbell et al. 2007). In contrast, reburns occur in young sapling forests (McIver and Ottmar 2007, McGinnis et al. 2010), meaning there are few large live trees to kill to generate new dead wood. Although our study sites are from one reburn event under its individual circumstances, we posit that this general mechanism would operate similarly across various forest types and burning conditions.

A greater proportion of legacy wood was in snags after a single fire, while the proportion shifted more toward down logs after the reburn. These findings agree well with those of Holden et al. (2006), who reported significantly lower snag densities, and greater snag charring, in multiple-burned areas relative to once-burned areas in drier interior forests dominated by *Pinus ponderosa*. Similarly, D'Amato et al. (2011) reported lower legacy structure after multiple interacting disturbances in sub-boreal pine forests. The lower snag biomass we observed was likely not driven by direct consumption of standing wood material, but rather to snags having fallen, decayed, and combusted on the surface by the second fire.

In addition to having greater levels of decay, the most notable reburn effect on dead wood character was the much greater charring of snags in twice-burned stands (Fig. 3). Char matter has distinctive properties including a chemical composition low in labile cellulose and hemicellulose (affecting decay rate and fuel availability to future fires), more porous surface textures (affecting water absorption capacity), and substrate properties influencing colonization by fungi and bryophytes (Pyne et al. 1996, Czimczik et al. 2002, Bradbury 2006). Charred wood retains macro- and micronutrients at substantially greater levels than uncharred wood or soil (Marañón-Jiménez et al. 2013), potentially sustaining site productivity despite multiple high-severity fires. Pyrogenic black carbon is also relatively stable and resides in soil for centuries to millennia, acting as a long-term carbon sink (Singh et al. 2012). Thus, reburns may result in a relatively high proportion of woody biomass diverted into this long-term carbon and nutrient pool relative to long-interval stand-replacement fires.

A common expectation of highly charred snags is that they are more "case-hardened" (e.g., Holden et al. 2006, Hyde et al. 2011) and thus have greater resistance to decay, fragmentation, and falling. With the much higher snag charring rates in the reburn (Fig. 3), we expected lower rates of snag fall and fragmentation than in high-severity fire areas outside the reburn. However, the rates were not different. Thus, greater case-hardening does not appear to be an important outcome of reburning in this system, at least not in a way that significantly alters subsequent snag dynamics.

Reburns, fire regime variability, and long-term stand development

These data show that substantial variations may occur within the traditional U- or reverse J-shaped model of dead wood succession, depending on the disturbance processes that initiate the stand. The pulse of dead wood we observed after a single fire fits the classic dead wood successional models (e.g., Harmon et al. 1986, Spies et al. 1988, Sturtevant et al. 1997, Hall et al. 2006, Bradford et al. 2009, Kashian et al. 2013), whereas the reburn appears poised to quantitatively (but not qualitatively) alter the long-term pattern (Fig. 5). Isolating the effect of one short-interval reburn (i.e., briefly assuming no other fires occur during stand development), this one event would reduce legacy wood by nearly half for up to ~50 yr, until the new stand begins recruiting new dead wood (Hudiburg et al. 2009), and would still have a measurable effect beyond 100 yr. This long-term imprint is consistent with previous hypotheses and modeling studies, suggesting that past reburns may explain anomalously low dead wood levels observed in some mature stands in otherwise dead wood-rich regions (Spies et al. 1988, Nonaka et al. 2007). The reburn influence was comparable to the effects of commonly observed variations in key factors such as overall decomposition and mortality rates that govern variations in necromass across a landscape (Fig. 4) (Harmon et al. 1986, Nonaka et al. 2007, Kennedy et al. 2008, Hudiburg et al. 2009). The analysis by Spies et al. (1988) suggests it would take several centuries for dead wood to reach a site-level steady-state in long-lived Douglas-fir forests, and as such, most forests in the region exist in a state of dead-wood disequilibrium driven by local disturbance history. Our data support this notion and show quantitatively how the nature of the event(s) that initiate a stand are a key factor.

As in many temperate conifer forests (e.g., Collins et al. 2009, Marcoux et al. 2013), fire regimes in portions of the maritime PNW Douglas-fir region are increasingly recognized as variable in frequency and severity—not just consisting of long-interval stand-replacing fires (Halofsky et al. 2011, Perry et al. 2011, Tepley et al. 2013). Within this larger context, our study focused

on an interaction that was available as a natural experiment (two closely timed stand-replacing fires). Our inferences have been linked to what can be learned from this type of event. We expect that additional fires may occur over the course of the next sere and further affect dead wood as the stand matures or resets. As an example, the dead wood quantities we measured in mature/old-growth forests were lower than the equilibrium levels the successional models project (Fig. 5), suggesting that some other driver, such as low-moderate severity fires, may suppress levels as stands age (Wright et al. 2002, Hudiburg et al. 2009). To date, such effects over many fire cycles have been addressed largely with models (Agee 2002) and are beyond the scope of our empirical example. Nevertheless, our data provide insight into the consequences for ecosystem structure when wide variations in fire interval occur.

In this larger fire regime context, reburns themselves may vary in nature and effects. The reburn interval of 15 yr that we studied is highly relevant given that fire risk and hazard can peak at this time point (Thompson et al. 2011), but impacts could vary with differing intervals, which affect the orientation and decay state of fire-killed wood (*via* snag-fall) as well as the accumulation of live biomass. Also, positive feedback cycles of recurring (i.e., >2) stand-replacing reburns at short intervals may be important in some areas (e.g., Martin et al. 1974) and have been posited as important determinants of system shifts (Odion et al. 2010, Stephens et al. 2012). Based on our findings, such a sequence could generate unique dead wood dynamics or even eliminate much legacy wood (*cf.* Brown and Johnstone 2011)—either through extensive decay, consumption, or both. For example, our data suggest that three successive reburns (assuming similar wood consumption rates) would result in ~80% lower dead wood than following a single fire. Studying these variations will be important to better understand the potential pathways and fates of "legacy" dead wood under fire regimes in which stand-replacing fires are but one component of a mixed-severity regime.

Acknowledgments

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