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Wildfire-initiated dead wood legacies: Post-fire habitat and fuels trajectories in westside Pacific Northwest forests, USA

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ABSTRACT

Managers grappling with questions about dead wood in productive, often high-volume, westside Pacific Northwest forests seek to balance wildlife habitat quality and fire hazard reduction, especially following wildfires which can both consume and generate exceptional quantities of dead wood. Longitudinal analysis of a large ($n = 327$) sample of permanent forest inventory monitoring plots measured before and repeatedly after wildfires and an equal-sized sample of comparable but unburned reference plots revealed post-fire trajectories of dead wood and factors that drive abundance of snags, litter, duff, fine (FW) and coarse down wood (CW) over the first two decades post-fire. Fire generated pulses of snag recruitment in these forests, increasing snag density to an average of 3X that found on unburned reference plots, and 5X where fire burned at high severity. Depending on fire severity, 25–45 % of these snags fell within the first two decades after fire, producing a lagged pulse of new down wood that yielded CW stocks 1.5X those found on unburned reference plots. Fine fuel loads were reduced in burned stands initially post-fire, but FW and litter began to re-accumulate during the first post-fire decade, returning to levels seen on unburned reference plots. Thus fire-initiated reductions in reburn potential were temporary while increases in proportion of forested landscape with the elevated levels of dead wood needed to support high quality wildlife habitat persisted.

1. Introduction

Managing dead wood resources is an increasingly complex challenge for land managers responsible for balancing demands concerning wildlife habitat, fuels, water quality, and soil stability. This task is especially complex in carbon-rich, temperate forests, where dead wood is abundant in snags and down wood (Lutz et al., 2021). We investigate fluxes and drivers of change in dead wood resources in mesic “westside” forests (found west of the crest of the Pacific Northwest’s Cascade Range) affected by wildfire. To address considerations that managers may face when balancing two important goals (maximizing wildlife habitat potential and mitigating risk to valued resources and assets by minimizing potential fire hazard), we explore implications of fire on habitat and potential reburns in these forests.

Post-fire dead wood trajectories in these westside forests have received little attention compared to those in more arid, frequent-fire ecosystems (Becker and Lutz, 2023; Donato et al., 2016; Eskelson and Monleon, 2018; Prichard et al., 2017). Because fires are infrequent in this region, the information that is available typically draws from studies that either characterize a single fire event or lack access to pre-fire data, which restricts inference at broad spatial scales and limits the ability to connect pre-fire conditions to post-fire outcomes. In general, landscape disturbances, particularly wildfire, can increase the abundance of standing and down dead wood by creating snags and generating down wood (Lutz et al., 2020). However, snags and down wood, along with litter and duff, can also be consumed by fire, potentially decreasing their abundance immediately post-fire (Becker and Lutz, 2023; Campbell et al., 2007). Post-fire dead wood and fuel trajectories are varied, shaped as they are by pre-fire forest conditions, fire severity, and regenerating vegetation (Campbell et al., 2007; Eskelson and Monleon, 2018; Lutz et al., 2020).

Owing to the importance of dead wood to forest ecosystems, a

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variety of computational tools and models have been developed to characterize dead wood dynamics, fluxes, and interactions. These tools can provide inventory-based, statistical estimates of dead wood based on predictor variables, or utilize a simulation modeling approach to generate projections of future dead wood based on parameter estimates of dead wood inputs and decay processes (Venier et al., 2015). In the United States, forest managers rely on tools like the Decayed Wood Advisor (DecAID), which synthesizes empirical information on wildlife use of dead wood in the Pacific Northwest from approximately 100 studies, to understand interactions between dead wood abundance and potential wildlife habitat (Marcot et al., 2010). Models like the Fire and Fuels Extension to the Forest Vegetation Simulator (FVS-FFE) or the First Order Fire Effects Model (FOFEM) provide managers with insight into the interactions between fire and dead wood. However, owing to the lack of empirical studies of post-fire dead wood dynamics in westside forests, limited guidance is available for calibrating model estimates or informing decisions about the management of wildlife habitat and fuels in recently burned, post-fire landscapes in this region. For example, DecAID does not contain data for post-fire wildlife habitat in the predominant forest type that exists throughout the westside (Westside Lowland Conifer Hardwood Forest), and it contains limited post-fire information for southwestern Oregon (Southwest Oregon Mixed Conifer Hardwood Forest), which is characterized by a mixed fire regime where fire has historically been more frequent (Mellen-McLean et al., 2017).

With an eye towards prioritizing the information needs of those responsible for the management and ecosystem health of westside forests, the USDA Forest Service Pacific Northwest Research Station organized and funded the Westside Fire Research Initiative – a scientific coproduction framework teaming scientists with land managers. The Initiative identified several questions requiring information about post-fire dead wood trajectories. Some centered on wildlife habitat, with an emphasis on snag creation and longevity, while others focused on post-fire fuel loads and reburn potential. We addressed these questions with a longitudinal analysis of permanent forest inventory monitoring plots located throughout westside forests that have been repeatedly measured prior to and following fire events. We anticipate that this analysis will provide insight regarding the interactions of wildlife habitat, fuels, and dead wood abundance in forests where fire has not been historically frequent and expect that our findings will be broadly applicable to forest managers, particularly those tasked with managing forest resources in mesic temperate forests.

1.1. *Wildlife habitat*

Dead wood is understood to be a critical wildlife habitat component. In the Pacific Northwest, approximately 150 forest-dwelling wildlife species, many of which are targeted with special conservation status, utilize dead wood for nesting, roosting, denning, and foraging (Marcot et al., 2010; O'Neil et al., 2001). Two forest types that are noted for their richness of wildlife species and abundance of dead wood are complex early seral and old-growth successional stages, which are a conservation priority due to their reduced prevalence relative to historical conditions (Reilly et al., 2021; Swanson et al., 2014). Concern over a dead wood deficit relative to historic conditions in this region has led to implementation of snag-generating management activities (such as through girdling, topping, or injection with herbicides) on some forest lands (Rose et al., 2001).

Wildfires alter wildlife habitat potential through snag creation, accelerated snag fall and snag consumption, which, depending on fire severity, can shift snag composition through the loss of large, decayed snags and addition of new, smaller snags (Becker and Lutz, 2023; Lutz et al., 2020). During a wildfire, combustion rates of down wood depend upon decay state and moisture content and are highly variable across fire severity classes (Eskelson and Monleon, 2018). Snag fall is a primary driver of coarse down wood accumulation, and its incidence varies by pre-fire stand condition, tree species, and environmental characteristics (Brown et al., 2013; Dunn and Bailey, 2015; Eskelson and Monleon, 2018; Lutz et al., 2020).

1.2. *Fuels*

Surface fuels are composed of coarse and fine down wood, litter, plus understory vegetation, where present. Litter and fine wood are important drivers of fire behavior in these forests, facilitating fire spread and/or fire intensity. Coarse down wood contributes primarily to fire intensity and residence time (Eskelson and Monleon, 2018; Lutz et al., 2020), so it can affect soil burn severity. Combustion and subsequent accumulation of litter, duff, and fine fuels is purported to be more variable and understudied in low- and moderate-severity fire regimes, which are characteristic of many westside forests (Campbell et al., 2007; Eskelson and Monleon, 2018). Litter is often more readily consumed than duff during a fire, however litter biomass can quickly re-accumulate following a fire, fed by inputs from scorched and burned tree crowns and burgeoning understory regrowth; re-accumulation of duff may not begin until 5–10 years post-fire (Campbell et al., 2007; Dunn and Bailey, 2015; Eskelson and Monleon, 2018; Peterson et al., 2019). In a mature forest in the Sierra Nevada, Lutz et al. (2020) found that surface fuel biomass had returned to pre-fire levels by 6 years post-fire, except for duff, which had not yet begun to accumulate. While total fuel loads were similar, the composition had shifted towards greater loadings of fine wood and lower levels of live fuels. In dry conifer stands of western North America, accumulation of fine wood post-fire is estimated to peak between 6 and 18 years after the fire event (Dunn and Bailey, 2015; Eskelson and Monleon, 2018).

Differences in post-fire surface fuel accumulation rates may influence potential reburn likelihood and severity (Eskelson and Monleon, 2018). In drier systems, rapid post-fire fuel accumulation can be driven by small-diameter tree mortality (Lutz et al., 2020). Reburn severity has been linked to the severity of earlier fires, likely owing to severity-mediated differences in the accumulation of post-fire fuels (Thompson et al., 2007). Along with weather, surface fuels drive predictions of fire behavior and are a common emphasis for fuel reduction activities (Eskelson and Monleon, 2018).

1.3. *Post-fire dead wood legacies*

To address the information needs and priorities of forest managers, we trace multidecadal trajectories of dead wood over a broad range of conditions found in westside Pacific Northwest forests. We highlight patterns of dead wood accumulation in both burned and unburned forest stands, identifying trends and variability occurring across the landscape. We assess changes in snag density (implemented here via basal area), as well as quantities of coarse and fine down dead wood, highlighting factors driving these changes. Acknowledging the complexity of managing dead wood resources, we address implications of post-fire dead wood accumulation through analyses of both potential habitat of non-fish vertebrate taxa and reburn behavior. Through analysis of repeatedly sampled permanent forest inventory monitoring plots we provide insight into the pre-, post-, and post-post-fire dynamics of dead wood in westside forests. Our objectives were to 1) describe post-fire dead wood trajectories, 2) identify the factors that influence those trajectories, and 3) explore how changes in post-fire dead wood abundance affect both potential wildlife habitat and future fire behavior.

2. Materials and methods

2.1. *Field data*

To examine the patterns and drivers of post-fire dead wood dynamics, we utilized a longitudinal fire effects dataset containing repeated measurements of forest inventory plots drawn from a spatially balanced sample of federal forest land in Oregon and Washington consisting of 4544 forested plots established by the Current Vegetation Survey (CVS) monitoring system in the 1990s across National Forest and BLM lands. The 654 plots selected for study were initially surveyed under the CVS monitoring protocol and continue to be surveyed through the National Forest Inventory conducted by the USDA Forest Service's Forest Inventory and Analysis (FIA) program and are located between the Pacific Ocean and the crest of the Cascade Mountains (i.e. the Pacific Northwest "westside", Supplementary Information (SI) Figure 1). Half of these plots, described here as the post-fire trajectories (PFT) set, were encountered by fire between 1994 and 2020 and the other half, which serve as a reference set, were not. PFT plots are located within the footprints of 79 wildfires and 10 prescribed fires that collectively burned 2.5 million acres (USGS Combined wildland fire dataset; Welty and Jeffries, 2021). All inventory plots that were located within a fire footprint and/or had field-recorded observations of fire disturbance were included in the PFT set, conditioned on having at least one measurement visit pre- and post-fire.

Plots within both the PFT and Reference sets were first visited and assessed to collect information on trees, down wood, understory vegetation and site characteristics between 1993 and 2002. All plots were revisited to collect remeasurement data, usually on multiple occasions; the 327 encountered by fire (the PFT set) saw remeasurement visits once (79 plots), twice (141 plots) or 3 or more times (107 plots) following the fire event. 133 plots encountered by fire were assessed between 2004 and 2007 with a remeasurement visit under the CVS protocol. All plots were remeasured once or twice under the annual FIA protocol between 2003 and 2022 to yield a total of up to 4 assessments of these plots over 29 years.

The other 327 comparable unburned plots (the reference set) were selected through Distance-Adjusted 1:1 Nearest Neighbor Propensity Score Matching (DAPSM) to identify reference plots with characteristics comparable to the burned plots, following the distance-adjusted propensity score matching method applied in a forest inventory analysis of wildfire effects on forest carbon in Woo et al. (2021). Distance-adjusted propensity matching utilizes the Euclidean distance between plots in addition to other covariates when pairing observations between the reference set and PFT set (Papadogeorgou et al., 2019). Reference plots were drawn from the same set of plots from which burned plots were selected, except that reference plots had to be free of wildfire over a 40-year period centered on the visit date for that plot for the sampling visit closest in date to the pre-fire sampling visit on the burned plot to which it was matched. The best matched 327 plots were selected from the 3822 plots meeting the fire-free criterion via DAPSM (packages "MatchIt" and "DAPSm") in the R software (Ho et al., 2011; Papadogeorgou, 2016; R Core Team, 2022). Propensity scores were calculated from the first measurement visit to each plot (dates ranging between 1993 and 1997), using live tree density (basal area), forest composition (proportion of basal area composed of conifer species), snag density, average tree size (Quadratic Mean Diameter), Mean Annual Increment (MAI), volume of coarse down wood (CW volume), and site elevation (SI Figures 1–4).

Plot configurations, sample area locations ("plot footprints"), and numerous other aspects of inventory protocol evolved over the time span for which these plots were assessed, with most of these changes taking place during the CVS to FIA Annual inventory protocol design transition in the early 2000s. The 5-subplot CVS design overlaps most completely with the 4-subplot annual FIA design on the area sampled at plot center (subplot 1 in FIA parlance, stake position 1 in CVS parlance; SI Figure 5). In both CVS and FIA protocols, down wood is sampled using line-intersect sampling along transects. However, the number, length, and orientation of these transects vary depending on sampling year, which can introduce uncertainty in estimates of temporal changes, as dead wood is not distributed uniformly across the landscape.

In both protocols, every piece of down wood larger than 7.6 cm diameter at the point of intersection with the transect and greater than 0.3 m in length (coarse down wood, CW), is individually tallied and at minimum, the diameter at the point of intersection is recorded. For fine down wood (FW) pieces, which are less than 7.6 cm diameter at the line intersect, all pieces are group tallied by size class. Medium FW (0.5–2.3 cm in CVS, and 0.6–2.3 cm in FIA) is tallied separately from large FW (2.5–7.4 cm in both protocols). Note that while FIA also samples small FW 0.03–0.6 cm in diameter, that material is not considered here because it was not sampled under the CVS protocol. For statistical analysis, CW cover at plot scale was estimated via Equation 3 of Woodall et al. (2019) and FW biomass was estimated via Equation 5 in the same publication, using bulk density and quadratic mean diameter (QMD) coefficients drawn from estimates by forest type (Woodall and Monleon, 2008). CW biomass was estimated via Equation 2 of Woodall et al. (2019), with species-specific bulk density and decay reduction factors. CVS utilized a 3-class decay condition system versus the 5-class system used by FIA. To apply decay reduction factors, we converted the CVS decay classes per Gray and Whittier (2014). Because duff and litter are sampled under FIA protocols, but not CVS, pre-fire measurements for these parameters are not available. We calculated post-fire and unburned reference duff and litter biomass via equation 6 of Woodall et al. (2019) using bulk density estimates by forest type (Woodall and Monleon, 2008).

2.2. *Modeled fuel loads*

Concern about reburns centers on likely fire intensity, which depends not just on total surface fuel loading but the distribution of fuel quantities among litter and 1-, 10- and 100-hr fuel amounts (0–0.6, 0.6–2.5 and 2.5–7.6 cm diameter fine woody material) on the forest floor, as well as the weather under which reburns occur. To obtain a management-relevant indicator of potential reburn intensity that integrates the observed fuel distribution and representative points on the distribution of weather conditions under which reburns may occur, we relied on the Fire and Fuels Extension (FFE) of the Forest Vegetation Simulator (FVS) to predict surface flame length under moderate and severe weather specified for the FVS variant applicable to each plot (these included Inland California and Southern Cascades [CA], Eastern Cascades [EC], Klamath Mountains [NC], Pacific Northwest Coast [PN], South Central Oregon/Northeast California [SO] and Westside Cascades [WC]). Surface flame length has important implications for fire type (e.g., surface versus crown) and for the tree mortality that will follow a fire event. Down wood sample data were provided as input to FFE's surface fuel model selection algorithm, which relies on down wood loading by size class to choose the best-fitting, weighted combination of two surface fuel models as a basis for modeling potential fire behavior (Rebain, 2010). Using the 40-fuel model system (Scott and Burgan, 2005), we identified two subsets of these 40 fuel models for FVS to utilize as potential fuel models (a dry subset for plots in the CA, EC, NC, and SO FVS variants and a humid subset for those in the PN and WC variants). Guided by observed fuel loading, FVS devises a selection of two fuel models and accompanying weights that, via interpolation, smooths the transition in predicted fire behavior (e.g., surface flame length) across the space of observed fuel loading combinations (e.g., of litter, 1-h, 10-h and 100-h down wood fuels) and follows a set of variant-specific decision trees to assign fuel models when down wood loading is very low. We modeled potential fire intensity under the variant-specific default moderate and severe fire weather scenarios. Both scenarios assume an ambient temperature of 21.1°C but have different wind speeds and fuel moisture, with 9.7 km/hr and "moist" fuels assumed for the moderate scenarios and 32.2 km/hr and "very dry" fuels assumed for the severe scenarios. Assumed fuel moisture values can be found in the variant-specific sections of the FFE documentation (Rebain, 2010).

2.3. *Burn severity classification*

To quantify the extent of fire-induced changes in tree vegetation on each plot, we classified fire severity in relation to field assessments of tree status and the implied changes to live tree density, calculated as the percent change in live tree basal area between the first post-fire (BA_{post}) and last pre-fire plot (BA_{pre}) visits ($\frac{BA_{post}-BA_{pre}}{BA_{pre}}$). Four levels of fire severity were defined: Very Low (increased basal area), Low (0–25 % basal area decrease), Moderate (25–75 % decrease), and High (75–100 % decrease).

2.4. Statistical analysis

2.4.1. Dead wood trajectories

To characterize post-fire dead wood trajectories, we characterized estimated mean abundances of dead wood (basal area [m^2/ha] for snags; percent cover and biomass [T/ha] for CW; biomass [T/ha] for FW) for the pre-fire (the last visit prior to the fire event, mean: 5.4 years pre-fire), post-fire (the first plot visit after the fire event, mean: 3.4 years post-fire), and post-post-fire period (the most recent plot visit following the post-fire visit, mean: 13.4 years post-fire) (Table 1 and SI Figure 6). Unburned reference plot measurements were similarly grouped into "pre-fire", "post-fire", and "post-post-fire" categories for ease of comparison during analyses. In this case, we used the fire year associated with the matched burned plot for each unburned reference plot and then conducted analyses as if that unburned reference plot had experienced fire during that year (SI Methods: Temporal Subsampling Protocol).

We used linear mixed models (R packages "lme4" and "emmeans") to model the response variables of dead wood abundance (snag biomass, CW cover and biomass, and FW biomass), with fire severity, measurement period, and their interaction included as fixed effects (Bates et al., 2015; Lenth, 2023). A random intercept for Plot ID was included in the model to account for the non-independence of repeat measurements. Statistically significant differences between groups (e.g., fire severity category, burn status) were found by applying post-hoc tests of Tukey's Honestly Significant Difference. We used an alpha level of 0.05 for all statistical tests.

2.4.2. Dead wood habitat

The Decayed Wood Advisor (DecAID) is an advisory tool that provides a statistical synthesis of empirical studies documenting wildlife utilization of dead wood in the Pacific Northwest. Based on a meta-analysis of approximately 100 studies and covering 112 terrestrial wildlife species and species groups, DecAID provides information on species' association and use of different types, sizes, and amounts of snags and coarse down wood (Marcot et al., 2010). To synthesize information across studies, DecAID utilizes statistical tolerance levels to characterize the percentage of each species' population that is associated with different density intervals of snags and coarse down wood. These tolerance levels are one-sided tolerance intervals with 0 as the lower limit, which provide estimates of the percent of all individuals in a population that fall within that interval (Marcot et al., 2010). DecAID provides information about the 30, 50, and 80 % "tolerance levels" for each species or taxa group (representing dead wood amounts that are associated with use by 30, 50, or 80 % of a wildlife population) to characterize the variability in how wildlife uses dead wood. The tolerance levels are also rank-ordered by habitat type and forest successional class, providing species curves where the cumulative wildlife tolerance level represents the tolerance level for the species that uses the highest density of dead wood (Marcot et al., 2010; Mellen-McLean et al., 2017). For example, 80 % of Cavity Nesting Birds would be provided adequate habitat with a density of 91 snags per hectare, and as they are the wildlife type associated with the highest density of snags, areas with this many snags (the cumulative wildlife tolerance level) could potentially provide habitat for the greatest number of species.

To assess potential habitat trajectories (changes in the abundance of potential wildlife habitat throughout the landscape over time), we analyzed the proportion of plots meeting or exceeding thresholds based on wildlife tolerance levels identified in DecAID and applied scaling factors to account for the proportion of the landscape that each plot represents (to account for differences in sampling intensity between reserved (typically as wilderness) and unreserved national forest lands). When interpreting dead wood trajectories in the context of wildlife habitat using DecAID, we conducted our analyses utilizing minimum

Table 1
Forest structure and sampling design characteristics of plots within the longitudinal fire effects dataset over time. Values are presented for each fire severity category, all burned plots, and all unburned reference plots. *For unburned reference plots, Time to Fire is determined relative to the fire year of the matched burned plot.

		Fire Severity Category				All Burned	Unburned Reference
		Very Low	Low	Mod.	High		
Average Interval of Time before/since Fire Years	Pre-fire	−5.4	−5.8	−5.2	−5.4	**−5.4**	−5.8*
	Post-fire	+ 3.4	+ 3.5	+ 3.3	+ 3.5	**+ 3.4**	+ 3.6*
	Post-post-fire	+ 12.9	+ 14.2	+ 12.9	+ 13.8	**+ 13.4**	+ 13.2*
Sample Size Number of Plots	Pre-fire	63	76	109	79	**327**	327
	Post-fire	63	76	109	79	**327**	322
	Post-post-fire	46	45	74	48	**213**	198
Stand Age Years Mean ± S.E.	Pre-fire	152 ±11	176 ±8	183 ±7	154 ±9	**169 ±4**	160 ±5
Stand Density Live Basal Area (m^2/ha) Mean ± S.E.	Pre-fire	33.6 ±2.9	49.7 ±2.6	47.6 ±2.1	32.9 ±2.6	**41.2 ±1.3**	44.2 ±1.3
	Post-fire	38.9 ±3.0	44.2 ±2.4	24.7 ±1.4	1.2 ±0.3	**26.4 ±1.3**	47.5 ±1.3
	Post-post-fire	36.5 ±3.6	39.2 ±3.5	20.9 ±1.9	2.7 ±0.7	**24.1 ±1.6**	50.7 ±1.9
Average Tree Size QMD (cm) Mean ± S.E.	Pre-fire	23.3 ±1.4	26.2 ±1.5	27.6 ±1.4	20.1 ±1.0	**24.6** ±0.7	24.4 ±0.6
	Post-fire	29.1 ±1.9	37.9 ±2.1	45.0 ±2.1	41.0 ±6.6	**39.0** ±1.4	25.3 ±0.7
	Post-post-fire	36.5 ±2.7	37.3 ±3.3	39.2 ±2.9	13.6 ±2.8	**34.7** ±1.7	26.4 ±1.0
Live Tree Biomass (T/ha) Mean ± S.E.	Pre-fire	207.3 ±20.7	326.4 ±22.4	305.5 ±17.1	192.8 ±20.0	**265.0 ±10.4**	286.3 ±10.3
	Post-fire	242.7 ±24.3	305.2 ±20.7	170.8 ±11.2	8.5 ±2.3	**177.2 ±9.7**	307.6 ±11.8
	Post-post-fire	256.8 ±34.0	278.2 ±30.7	138.3 ±14.7	9.7 ±4.5	**165.2 ±13.1**	344.7 ±17.9

thresholds of 25.4 cm DBH (diameter at breast height, 1.37 m) and 1.5 m height for snags, and 12.7 cm transect diameter and 0.91 m length for CW to best match the dead wood data utilized in DecAID. However, differences in protocol result in a small discrepancy between the minimum threshold for CW in this analysis versus what was used in DecAID (12.5 cm intersect diameter and 1 m length; Mellen-McLean et al., 2017).

We utilized thresholds based on tolerance levels identified in DecAID for Westside Lowland Conifer Hardwood Forest in the Oregon and Washington Cascade Mountains. Specifically, we identified the proportion of the landscape exceeding 91 snags (≥25 cm DBH) per hectare (the 80 % wildlife tolerance level for snag density) or 13 % cover of CW pieces (≥ 12.7 cm transect diameter and 0.91 m length; the 50 % wildlife tolerance level for down wood cover) (Mellen-McLean et al., 2017). We chose to utilize the 50 % cumulative wildlife tolerance level for CW as stands with cover estimates corresponding to the 80 % tolerance level (24 %) were rare. The CW cumulative wildlife tolerance levels reflect the high usage of dead wood by amphibians, particularly plethodontid salamanders—who utilize moist microclimates under dead wood for refugia, while the tolerance levels for mammalian species are lower (Kluber et al., 2009; Mellen-McLean et al., 2017). For example, the 50 % cumulative wildlife tolerance level (12.7 % CW cover) is based on Dunn's salamander (*Plethodon dunni*) and is similar to the 80 % tolerance level of Douglas squirrel (*Tamiasciurus douglasii*,12.3 % CW cover), but in between the 50 and 80 % tolerance levels of ermine (*Mustela erminea*, 8.4 and 17.8 % respectively) (Mellen-McLean et al., 2017). Detailed analyses of the variability in potential habitat trajectories for selected wildlife species, large snag (≥50 cm DBH) density, as well as the different cumulative wildlife tolerance levels identified in DecAID (30 %, 50 %, and 80 %) are included in the supplemental information (SI Tables 1–4).

2.4.3. *Drivers of variability*

To identify the dominant parameters driving variability among dead wood trajectories over time, we constructed random forest regression models to assess the relative importance of various forest structure, disturbance intensity, and climatic parameters associated with burned and unburned reference plots (see SI Table 8 for descriptions of predictors). We included parameters characterizing specific aspects of stand structure such as tree density by size class, average tree size (Quadratic Mean Diameter, QMD), and total live tree biomass. We also included Old Growth Structure Index (OGSI), a composite index that characterizes forest conditions based on the degree of late-stage successional elements present. We calculated OGSI per Davis et al. (2015), summing metrics of live tree density, diversity of tree sizes, large snag density and CW cover to generate an index where higher values indicate stands that exhibit more late-successional forest characteristics compared to stands with lower OGSI values.

For each model, we used the Boruta function (Kursa and Rudnicki, 2020) to determine which variables were confirmed to influence change in dead wood abundance over time and removed variables that were not confirmed from future analyses (SI Figures 7–12). Next, we selected and removed correlated (r ≥ 0.7) variables among the remaining predictor variables because the importance values generated by random forest are sensitive to highly correlated predictor variables (Hooker et al., 2021). First, we identified the set of variables with the highest pair-wise absolute value correlation and then eliminated the variable with the largest mean absolute correlation, repeating these steps until all correlations were below the threshold (package "caret", Kuhn, 2008). We then used the remaining variables to construct random forest regression models to identify variables that are important predictors of annual change in dead wood, calculated as the change in dead wood quantities observed on the plot between the pre-fire plot visit (DW_{pre}) closest in time to the fire date and to the most recent plot visit (DW_{last}), annualized by time between those visits ($\frac{DW_{last}-DW_{pre}}{YEAR_{last}-YEAR_{pre}}$). We used the randomForest package (Liaw and Wiener, 2002, Cutler et al. 2007) to build 6 separate models predicting change in each of snag basal area, CW cover, and FW biomass response variables for the burned and reference plots. We report variable importance for the most important variables of each model in terms of the % increase in mean standard error (MSE), which compares the change in accuracy of predictions on the out of bag samples when values of the given variable are permuted to the original observations (Liaw and Wiener, 2002). Boruta variable importance results and correlations are included in the supplemental information (SI section 2.4.2).

3. Results

3.1. *Post-fire dead wood trajectories*

3.1.1. *Snags*

Unsurprisingly, post-fire snag density (implemented here via basal area) increases immediately post-fire before decreasing, via attrition during the post-post-fire period (Fig. 1). Per expectations, post-fire snag density is strongly, positively correlated with burn severity, reaching a level of 5X that on reference (unburned) plots at the highest level, and is 3X across all burn severities (Table 2). Snag attrition (the magnitude of snag fall in m^2/ha) and conversion to down wood, a process which continues over the first two decades following fire, also correlated strongly and positively with burn severity. For low, moderate and high severities combined, mean snag basal area dropped to approximately 2X the level on reference plots. Snag densities remained unchanged in unburned reference and very low severity burned stands. The proportion of the landscape exceeding a threshold of 91 snags per hectare that are equal to or larger than 25 cm DBH (the 80 % wildlife tolerance level specified in DecAID) increased in association with burn severity. In fire-affected areas, 6.3 % of the landscape met this threshold pre-fire, versus 30.9 % post-fire and 21.1 % by the post-post-fire visit (Table 3). Stands that burned at high severity saw the greatest change, with 10.0 %, 58.8 %, and 40.0 % of them meeting this threshold pre-, post-, and post-post-fire, respectively. Similar patterns were observed for the distribution of potential large snag habitat across the landscape (SI Table 2).

3.1.2. *Coarse down wood*

Post-fire down wood stocks were also dynamic, decreasing as a result of consumption by fire followed by an increase in both fine and coarse down wood during the interval between the post- and post-post-fire plot visits (Fig. 1). Mean cover of CW in burned stands decreased from 3.7 % to 2.8 % on average for all plots regardless of severity during the initial post-fire period (with post-fire coarse down wood cover of 1.9, 3.0, 3.4, and 2.9 % in stands burned at very low, low, moderate, and high severity) (Table 2). Mean CW cover increased to 5.0 % during the post-post-fire period (with post-post-fire percent covers of 2.9, 5.3, 5.9, and 5.3 % in stands that burned at very low, low, moderate, and high severity, respectively). CW biomass trajectories resembled that of CW cover, with significant increases in burned stands during the post-post-fire period (Table 2 and SI Table 6). CW biomass decreased from 20.2 to 17.4 T/ha on average for all plots regardless of severity during the initial post-fire period (driven by a decrease from 20.0 to 11.4 T/ha in high severity plots). Mean CW biomass increased to 26.4 T/ha during the post-post-fire period (with post-post-fire percent covers of 13.6, 31.2, 33.3, and 24.5 T/ha in stands that burned at very low, low, moderate, and high severity, respectively). The proportion of the landscape exceeding the 13 % cover threshold for coarse down wood (the 50 % wildlife tolerance level specified in DecAID) initially decreased, then increased during the post-post-fire period. Pre-fire, 1.7 % of the landscape met this threshold; post-fire, only 0.8 % did, and by the post-post-fire plot visit, 3.2 % had more than 13 % CW cover (Table 3). Change in potential CW habitat was greatest where burn severity was high, with 2.9 %, 0.0 %, and 6.5 % of the landscape that burned at high severity meeting this threshold pre-, post-, and post-post-fire. Stands

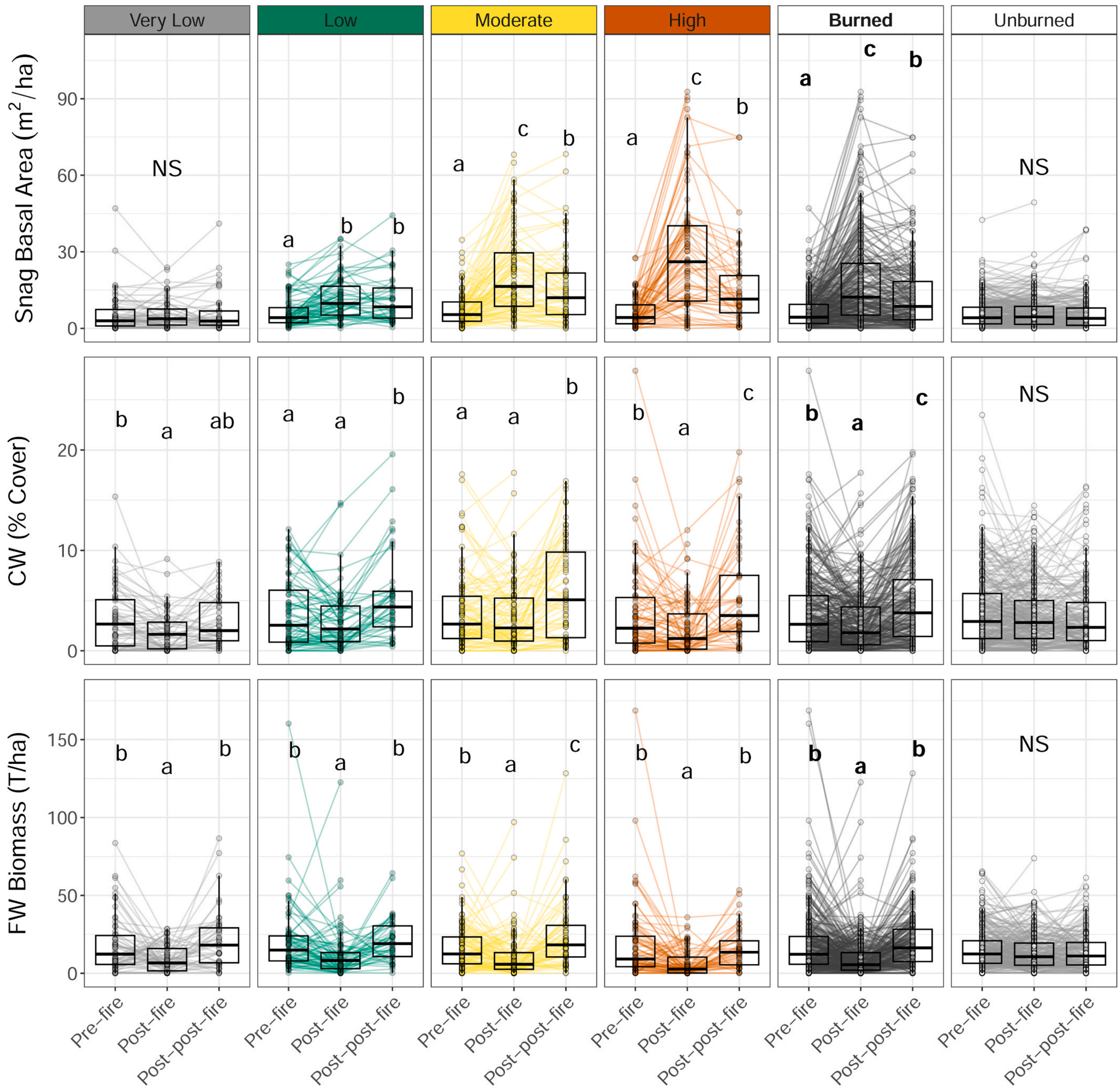

Fig. 1. Dead wood trajectories by burn severity and plot visit timing relative to fire date. Pre-, post-, and post-post-fire mean timings are 5.4 before, or 3.4 or 13.4 years after the fire event, respectively. Values are presented for each fire severity category, all burned plots, and all unburned reference plots. Markers denote dead wood abundance for each plot, with lines connecting measurements from the same plot through time. Differences in lowercase letter (a, b, c) labels in each panel indicate significant differences across time within a burn severity category (post-hoc Tukey HSD), while NS indicates no significant difference across time.

burned at low severity saw similar change from pre- to post-post-fire, with the proportion of area having 13 % CW cover expanding from 0.0 % to 3.5 % of the landscape.

3.1.3. Fine down wood

Fine wood had decreased by the post-fire visit but returned to pre-fire levels during the post-post-fire period. Mean FW biomass decreased from 18.1 to 9.9 T/ha by the post-fire visit (with post-fire fine wood biomass of 9.5, 12.1,10.9, and 6.5 T/ha where fire burned at very low, low, moderate, and high severity, respectively) (Table 2). Mean FW biomass increased to 20.7 T/ha by the post-post-fire visit, with little difference between fire severity classes. While not statistically significant, FW biomass, litter, and duff decreased on the unburned reference plots.

3.1.4. Reburn potential

Modeled potential surface flame length in the case of reburn under moderate fire weather was initially lower for burned plots than for unburned reference plots during the immediate post-fire period (0.66 versus 0.82 m, $p = 0.0141$) (Table 3). Under severe fire weather, flame length was not significantly different between the two plot sets (2.01 versus 1.88 m, $p = 0.4018$), except where high burn severity was observed at the post-fire visit –modeled surface flame lengths in those stands were greater than for all other severity categories (2.70 m versus 1.59–1.95 m). By the post-post-fire visit, modeled surface flame lengths for a potential reburn on burned plots were greater than on unburned

Table 2
Snag, coarse wood (CW), fine wood (FW), litter, and duff trajectories of plots within the longitudinal fire effects dataset over time. Mean abundance ± standard error. Values are presented for each fire severity category, all burned plots, and all unburned reference plots. *Pre-fire litter and duff biomass estimates are unavailable because those measurements weren't in the inventory protocols followed at that time.

		Fire Severity Category				All Burned	Unburned Reference
		Very Low	Low	Mod.	High		
Snag Density Basal Area (m^2/ha)	Pre-fire	5.5 ±1.0	6.2 ±0.7	7.4 ±0.7	6.2 ±0.7	**6.5 ±0.4**	5.9 ±0.3
	Post-fire	5.5 ±0.7	11.8 ±1.0	21.2 ±1.5	29.9 ±2.7	**18.1 ±1.0**	6.0 ±0.3
	Post-post-fire	5.8 ±1.2	11.3 ±1.4	16.0 ±1.7	16.3 ±2.4	**12.9 ±1.0**	5.9 ±0.5
CW Abundance (Percent Cover)	Pre-fire	3.1 ±0.4	3.7 ±0.4	3.9 ±0.4	3.8 ±0.5	**3.7 ±0.2**	4.0 ±0.2
	Post-fire	1.9 ±0.3	3.0 ±0.3	3.4 ±0.3	2.9 ±0.3	**2.8 ±0.2**	3.6 ±0.2
	Post-post-fire	2.9 ±0.4	5.3 ±0.6	5.9 ±0.6	5.3 ±0.7	**5.0 ±0.3**	3.5 ±0.2
CW Biomass (T/ha)	Pre-fire	14.5 ±3.1	19.9 ±3.1	23.8 ±3.2	20.0 ±3.8	**20.2 ±1.7**	19.0 ±1.4
	Post-fire	12.7 ±2.9	17.1 ±2.5	24.7 ±3.2	11.4 ±2.0	**17.4 ±1.4**	19.0 ±1.4
	Post-post-fire	13.6 ±2.2	31.2 ±5.5	33.3 ±3.7	24.5 ±4.2	**26.4 ±2.1**	17.4 ±1.7
FW Biomass (T/ha)	Pre-fire	18.2 ±2.3	20.1 ±2.5	16.4 ±1.4	18.4 ±2.8	**18.1 ±1.1**	15.9 ±0.8
	Post-fire	9.5 ±1.1	12.1 ±2.0	10.9 ±1.4	6.5 ±1.0	**9.9 ±0.7**	14.8 ±1.4
	Post-post-fire	21.0 ±2.8	21.2 ±2.1	23.3 ±2.4	15.9 ±1.9	**20.7 ±1.2**	14.2 ±0.9
Litter Biomass* (T/ha)	Post-fire	6.3 ±0.8	7.2 ±1.4	4.4 ±0.5	4.1 ±0.6	**5.3 ±0.4**	13.0 ±1.0
	Post-post-fire	8.4 ±0.9	7.6 ±1.0	8.8 ±1.0	10.3 ±1.3	**8.8 ±0.5**	10.1 ±0.6
Duff Biomass* (T/ha)	Post-fire	7.0 ±1.3	6.3 ±1.6	6.2 ±1.0	6.5 ±2.0	**6.5 ±0.7**	19.4 ±1.2
	Post-post-fire	6.5 ±1.4	4.7 ±0.9	5.3 ±1.1	4.5 ±1.5	**5.2 ±0.6**	15.7 ±1.2

Table 3
Areal representation of dead wood trajectories relative to wildlife habitat and reburn potential across the landscape. Values presented for each fire severity category, all burned plots, and all unburned reference plots. *Proportion of area exceeding DecAID 80 % Wildlife Tolerance Level of 91 snags per hectare ≥ 25 cm DBH. **Proportion of area exceeding DecAID 50 % Wildlife Tolerance Level of 13 % Cover of CW pieces ≥ 12.7 cm transect diameter and 0.91 m length. ***Sample size for modeled surface flame length: 123 burned and 100 unburned reference plots (sample limited to plots with Post-fire and Post-post-fire visits conducted using FIA protocol due to modeling constraints).

		Fire Severity Category				All Burned	Unburned Reference
		Very Low	Low	Mod.	High		
Proportion of Total Burned Area	-	18.8 %	22.2 %	32.0 %	27.1 %		-
Snag Habitat Proportion of Total Burned Area *	Pre-fire	0.6 %	0.9 %	2.1 %	2.7 %	**6.3 %**	1.7 %
	Post-fire	0.2 %	3.9 %	10.9 %	15.9 %	**30.9 %**	1.0 %
	Post-post-fire	1.0 %	2.3 %	8.2 %	9.6 %	**21.1 %**	1.2 %
Snag Habitat Proportion of Burn Severity Patches *	Pre-fire	3.2 %	4.1 %	6.4 %	10.0 %		
	Post-fire	1.1 %	17.8 %	34.0 %	58.8 %		-
	Post-post-fire	4.8 %	11.5 %	23.2 %	40.0 %		
CW Habitat Proportion of Total Burned Area**	Pre-fire	0.3 %	0.0 %	0.6 %	0.8 %	**1.7 %**	2.1 %
	Post-fire	0.0 %	0.4 %	0.5 %	0.0 %	**0.8 %**	1.0 %
	Post-post-fire	0.0 %	0.7 %	1.0 %	1.5 %	**3.2 %**	2.2 %
CW Habitat Proportion of Burn Severity Patches**	Pre-fire	1.4 %	0.0 %	2.0 %	2.9 %		
	Post-fire	0.0 %	1.6 %	1.5 %	0.0 %		-
	Post-post-fire	0.0 %	3.5 %	2.9 %	6.5 %		
Modeled Surface Flame Length* Moderate Fire Weather (m) Mean ± S.E.	Post-fire	0.68 ±0.08	0.80 ±0.09	0.57 ±0.08	0.65 ±0.11	**0.66 ±0.05**	0.82 ±0.04
	Post-post-fire	0.87 ±0.09	1.01 ±0.11	1.05 ±0.08	1.10 ±0.09	**1.01 ±0.05**	0.82 ±0.04
Modeled Surface Flame Length* Severe Fire Weather (m) Mean ± S.E.	Post-fire	1.59 ±0.21	1.70 ±0.16	1.95 ±0.20	2.70 ±0.28	**2.01 ±0.12**	1.88 ±0.10
	Post-post-fire	2.20 ±0.19	2.10 ±0.18	2.83 ±0.20	3.91 ±0.25	**2.80 ±0.12**	1.82 ±0.08

reference plots under both fire weather assumptions (1.01 versus 0.82 m in moderate weather and 2.80 versus 1.82 in severe weather, $p = 0.002$ and $p < 0.0001$ respectively). Under severe fire weather, modeled surface flame lengths were also greater in stands that had burned at high severity compared to all other categories (3.91 m versus 1.82–2.83 m). For reburns that occur under moderate fire weather, burn severity observed at the post-fire visit was not a significant predictor of surface flame length for reburns in either the immediate post- or the post-post-fire periods.

3.2. Drivers of dead wood trajectory variability

Random forest models explained 56 %, 34 %, and 32 % of the variability in annual change between the pre-fire and most recent plot visits for each of the dead wood pools in burned plots, and 2 %, 33 %, and 25 % of the variability in annual change in unburned reference plots (snag biomass, CW cover, and FW biomass respectively). Fire severity was the most important predictor of change in snag abundance on burned plots (accounting for a 72 % increase in model MSE; i.e., the model MSE would increase by 72 % if this predictor was not included in

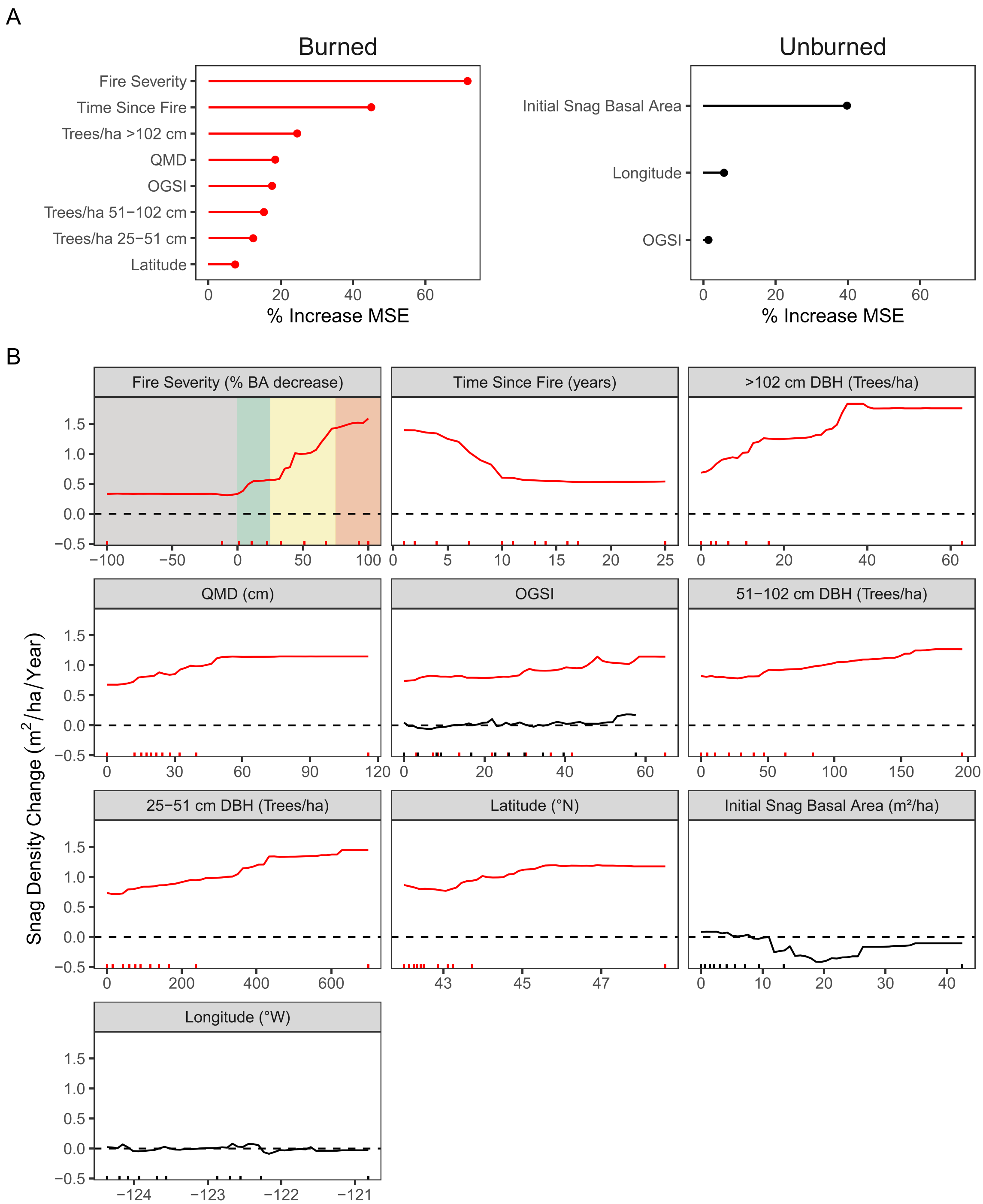

Fig. 2. Drivers of temporal change in snag abundance for burned (red) and unburned plots (black). Panel A: random forest variable importance showing the increase in mean standard error that each variable accounts for within each model (% Increase MSE). Panel B: the partial dependence of annual percent change in snag abundance for the variables included in the random forest models. Positive values indicate a net accumulation of snags from the pre-fire period to the most recent measurement, while negative values indicate a net loss. To visualize the distribution of predictor variables, points along the horizontal axis mark the data deciles. Shaded regions in the fire severity partial dependence plot indicate the four fire severity categories: very low (grey), low (green), moderate (yellow), and high (red).

the model), with partial dependence plots showing greater snag accumulation in more severely burned stands that had higher densities of larger-size trees (25–51, 51–102, and ≥102 cm DBH), greater OGSI values, and were located at higher latitude (Fig. 2). Initial CW cover was the dominant predictor of CW change in both plot sets (accounting for a 43 % increase in MSE for burned and 75 % for unburned reference plots), followed by time since fire (34 % increase in MSE) in burned and latitude in unburned reference plots (30 % increase in MSE) (Fig. 3). In burned plots, partial dependence plots showed an increase in CW abundance with higher density of both small and very large trees (5–25 and ≥102 cm DBH), as well as higher accumulation rates in stands with larger average tree size (QMD). In unburned reference plots, CW abundance decreased more in warmer, drier, stands in the southern portion of the study region. Pre-fire FW biomass was the most influential predictor of FW biomass change (52 % increase in MSE in burned and 84 % in unburned reference plots), followed in burned plots by OGSI and the abundance of large trees (51–102 cm DBH; 17 and 15 % increase in MSE). In unburned reference plots the second most influential predictor was live tree basal area (22 % increase in MSE), which had a non-linear effect on FW change (Fig. 4). Time since fire was an important predictor in each burned plot model (45 %, 34 %, and 8 % increase in MSE for snag, CW, and FW respectively), with partial dependence plots showing a decreasing rate of snag accumulation 5–10 years post-fire, matched with a rapid increase in CW and FW accumulation during this period. Spatially varying predictors (such as latitude, longitude, mean annual temperature, and mean annual precipitation) were frequently included in the random forest models. Dead wood trajectories by EPA Level 3 Ecoregion (areas where ecosystems, biotic, and abiotic phenomena are generally similar) are summarized in the supplementary information (SI Tables 5–7) (Omernik, 1987; U.S. Environmental Protection Agency, 2013).

4. Discussion

4.1. Dead wood trajectories

Dead wood pools were dynamic in the first couple of decades following fire as standing dead wood created by the fire transitioned to the forest floor, replacing initial losses of down dead wood. This pattern occurred throughout the burned landscape, albeit with more substantial changes occurring where higher burn severity produced greater tree mortality. Observed changes in dead wood abundance reflect both additions (mainly post-fire) and subtractions (mainly during fire) of dead wood over a time interval that includes both processes. Post-fire snag density was 3X greater than in unburned reference plots and stands that burned more severely had a higher proportion of snags immediately post-fire (amounting to 5X as many compared to unburned reference). The basal area magnitude of the snag pulse was positively correlated with fire severity, so while the largest initial increase in snags occurred in severely burned stands; these stands were also where snag fall was greatest over the post-fire period. Depending on fire severity category, 25–45 % of snags fell within the first two decades after fire, greatly augmenting large down wood pools in recently burned stands. Despite the attrition of snags, a durable positive association between snag density and fire severity remained evident more than a decade after fire.

High severity burned areas had lower pre-fire live tree biomass, basal area, and average tree size than both unburned reference plots and stands burned at low and moderate severity; consistent with other studies that found higher burn severity in lower stocked, less productive stands affected by the 2002 Biscuit Fire Complex (Azuma and Christensen, 2005; Azuma et al., 2004). Like high severity burned stands, those that burned at very low severity also had lower pre-fire live tree biomass and basal area, although average tree size was similar to that of the other categories. As burned and unburned reference plots had equal distributions of these pre-fire forest conditions (per the Distance-Adjusted Propensity Score Matching process), these differences suggest a tendency for fires in this region to burn at either end of the fire severity spectrum in areas of lower stand density and tree size, likely dependent on the availability of surface fuels and weather when fire arrived.

Coarse down wood cover initially decreased post-fire due to combustion but then increased by a greater magnitude following the snag pulse, ultimately yielding 1.5x the amount of large down wood in low, moderate, and high severity stands relative to unburned and very low severity. While the net change in snag abundance in burned plots was primarily driven by fire severity and the pre-fire density of larger trees, change in CW was also driven by the density of small trees, suggesting that smaller trees may have faster snag turnover times than large diameter trees, a pattern that has been noted elsewhere (e.g. Dunn and Bailey, 2012; Grayson et al., 2019; Becker and Lutz, 2023). OGSI was an important predictor of dead wood accumulation in each of the random forest models except change in FW biomass in unburned stands. Partial dependence plots indicated that fire in stands with higher initial OGSI values (i.e., exhibited more late successional forest characteristics in terms of live tree density, diversity of tree sizes, large snag density and CW cover) typically produced more snags, while down wood production was greatest in stands with mid-range OGSI values.

Initial dead wood abundance was an important predictor in each of the models except for snag density on burned plots. The relative importance and directionality of this predictor follow from how the response variable is defined—as the absolute change between pre-fire and post-post-fire. Initial abundance is negatively associated with dead wood change because plots with little dead wood pre-fire are more likely to gain then lose dead wood, while plots with very high levels of dead wood pre-fire have far greater potential to experience large dead wood loss. Overall, initial dead wood abundance was a more important predictor (accounting for a greater % increase of MSE) for unburned reference plots than for burned plots, confirming the importance of fire effects in predicting dead wood change in burned stands.

While fine down wood, litter, and duff were readily combusted and remained less abundant than on unburned reference plots during the first few years post-fire, FW and litter rapidly reaccumulated. FW reached pre-fire levels by the time of the post-post-fire plot visit (and where burn severity was moderate, exceeded pre-fire levels). Re-accumulating litter matched levels on unburned reference plots by the post-post-fire period; duff levels did not increase during this time, which is consistent with findings in dry forests (Campbell et al., 2007; Dunn and Bailey, 2015; Eskelson and Monleon, 2018; Peterson et al., 2019).

Combustion and re-accumulation of litter, duff, and fine dead wood are thought to be more variable where burn severity is low or moderate (Campbell et al., 2007; Eskelson and Monleon, 2018). We saw this pattern for FW and for the initial re-accumulation of litter but did not see it for duff or for post-post-fire litter amounts. While net change in snag and CW abundance was driven by fire severity and other aspects of pre-fire conditions, change in FW was not. This suggests that post-fire inputs from residual scorched or burned vegetation and burgeoning regrowth drive re-accumulation of fine fuels more than fire severity as this class of woody fuels is readily combusted regardless of fire intensity (Campbell et al., 2007).

4.2. Wildlife habitat

Post-fire increases in dead wood abundance yielded a corresponding increase in potential wildlife habitat. Snag habitat was 5X more abundant across the recently burned landscape, and after a decade of snag fall, remained 3X more abundant than pre-fire. As we expected, high severity patches yielded the most snag habitat (initially increasing from 10 % to 60 % of the area exceeding the 80 % wildlife threshold and then falling to 40 %). Large increases in habitat also occurred where fire severity was low or moderate. The availability of habitat dependent on CW initially decreased post-fire (from 1.7 % to 0.8 % of burned areas exceeding the 50 % wildlife tolerance level), as the proportion of area

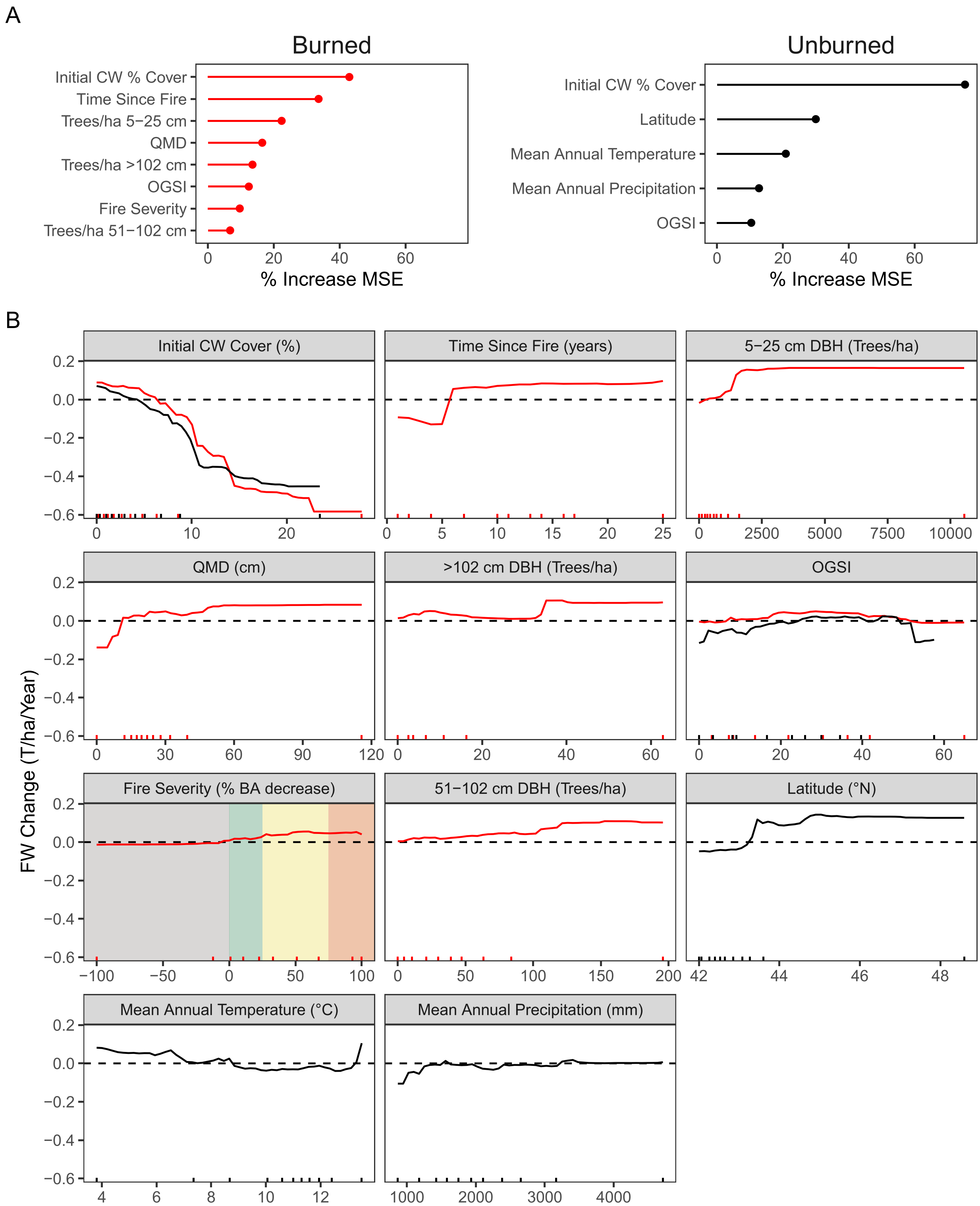

Fig. 3. Drivers of temporal change in coarse down wood abundance for burned (red) and unburned plots (black). Panel A: random forest variable importance showing the increase in mean standard error that each variable accounts for within each model (% Increase MSE). Panel B: the partial dependence of annual percent change in coarse down wood abundance the variables included in the random forest models. Positive values indicate a net accumulation of CW from the pre-fire period to the most recent measurement, while negative values indicate a net loss. To visualize the distribution of predictor variables, points along the horizontal axis mark the data deciles. Shaded regions in the fire severity partial dependence plot indicate the four fire severity categories: very low (grey), low (green), moderate (yellow), and high (red).

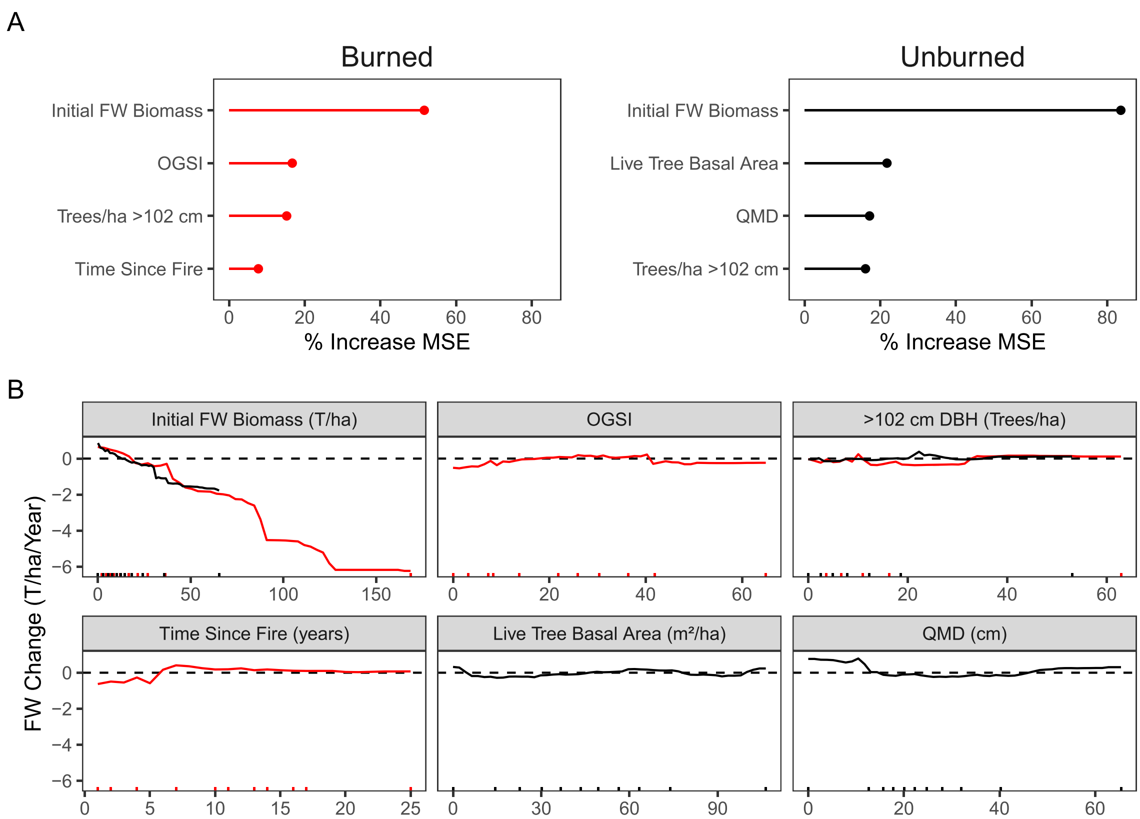

Fig. 4. Drivers of temporal change in fine down wood biomass for burned (red) and unburned plots (black). Panel A: random forest variable importance showing the increase in mean standard error that each variable accounts for within each model (% Increase MSE). Panel B: the partial dependence of annual percent change in fine down wood biomass for the variables included in the random forest models. Positive values indicate a net accumulation of snags from the pre-fire period to the most recent measurement, while negative values indicate a net loss. To visualize the distribution of predictor variables, points along the horizontal axis mark the data deciles.

with high levels of CW decreased, particularly where fire burned at moderate and high severity. CW habitat increased in the decade following fire, ultimately exceeding pre-fire availability by 2X. Large, decayed snags are purported to be susceptible to accelerated snag fall rates post-fire (Becker and Lutz, 2023; Lutz et al., 2020) potentially shifting the composition away from large, decayed snags to new, smaller diameter ones which could have potential implications for utilization by wildlife, but we found a durable increase in the availability of large snags within burned stands. Wildlife associations with dead wood vary within and among taxa. Accordingly, the potential habitat trajectories vary based on the association between each species and their usage of dead wood. For example, Dunn's Salamander, the species associated with the highest cover of CW had an increase of potential habitat from 1.7 % to 4.0 % of the burned landscape pre- to post-post-fire, while ermine increased from 7.5 % to 14.3 % and Douglas squirrel increased from 12.1 % to 20.8 %.

Together, the increases in snags and CW yielded significant increases in the availability of potential wildlife habitat across the burned landscape, substantially increasing the proportion of area with dead wood quantities that support nesting, roosting, denning, and foraging activities. By demonstrating the variability of post-fire dead wood resources and associated implications for potential wildlife habitat, this work points to a need for more research to help inform management of post-fire landscapes. This analysis may help to inform management tools such as DecAID by characterizing the transformations of dead wood pools in the decades following fire. Owing to the importance of dead wood resources to various wildlife taxa and the variety of such uses, an in-depth analysis of post-fire dead wood trajectories and implications for specific wildlife taxa would provide further insight into their multifaceted habitat needs.

4.3. *Reburn potential*

Modeled reburn potential was more sensitive to assumed weather conditions than surface fuel availability, but there was an increase in reburn potential during the decade after fire. Modeled surface flame lengths under moderate fire weather conditions were lower in burned plots (0.66 m versus 0.82 m in unburned reference plots) during the post-fire period (except for plots that had burned at low severity), indicating lower reburn potential during the first few years post-fire. During the post-post-fire period a decade later, modeled surface flame lengths under assumed moderate weather conditions increased and were subsequently greater in burned plots than in unburned reference (1.01 m versus 0.82 m) across all but the very low fire severity categories, implying an increasing reburn potential over time that corresponds with the increase in litter and FW during that period and the still-elevated CW fuel loads. Modeled surface flame lengths were greater when fire weather was assumed severe and were similar for burned and unburned reference plots at the first post-fire visit (apart from stands that burned at high severity, where flame lengths were greater even at the first post-fire visit). A decade later, modeled surface flame lengths under severe weather conditions were significantly greater in previously burned plots compared to unburned reference plots (2.80 m versus 1.82 m), with higher magnitude increases in stands previously burned at higher severities. These patterns reflect a higher reburn potential under severe weather conditions occurring a decade after fire that is likely driven by increased dead wood abundance on burned plots and corresponds with the increase in FW and CW during that period. These findings support hypotheses of a greater likelihood of high intensity reburns occurring following stand replacing fires as surface fuels begin to re-accumulate, a pattern observed in reconstructed tree ring-based fire chronologies within the southern Cascades (Merschel et al., 2024), which may contribute to the development of complex early seral habitats historically found on the westside.

4.4. Considerations

Variations in transect lengths across the protocols by which data were collected yield different estimation errors for dead wood material (DWM) depending on the measurement period associated with a plot. Protocol changes in DWM transect location and direction also contribute to variability in estimated abundances of down dead wood change as dead wood is not distributed evenly across the landscape. Changes in transect length, location, or orientation could result in a sizeable estimation error for individual plots, particularly for large CW pieces, which can have high spatial heterogeneity; however, we do not expect this to significantly impact mean estimates of change across plots. Down wood piece inclination was not included in CW calculations for this study as it is a data item not measured under the CVS protocol, or for most of the FIA data series. Omitting that parameter may yield underestimates of abundance that approximate 5 % (Monleon, 2009). However, this is unlikely to significantly impact estimates of change in CW abundance over time.

While we found increased potential wildlife habitat availability in burned stands, specific habitat requirements extend far beyond just the quantity of dead wood on the landscape and could be affected by fire in myriad ways beyond the scope of this analysis (e.g., post-fire changes in understory vegetation altering the availability of preferred food sources or the provision of cover for prey species) (Smith, 2000). These nuances may alter the suitability of post-fire conditions for certain wildlife species and add additional complexity for managers to consider. For example, patches of high severity fire that create landscapes rich in snags and down wood, but with limited residual live tree canopy, may benefit some species to the detriment of others. Given the varying ranges of dead wood density that individuals, populations, and species utilize, heterogeneity in dead wood abundance and distribution across the landscape may have beneficial consequences (Bunnell et al., 2002).

Comparison of dead wood trajectories for both burned and comparable unburned reference plots selected via Distance-Adjusted Propensity Score Matching provides estimates of fires' effects while accounting for potentially confounding changes in forest conditions (i.e. baseline variability in forest productivity and non-fire disturbances) that may have been occurring over the ~30-year analysis window. Dead wood pools did not significantly change within unburned reference plots, yielding no meaningful changes in potential dead wood habitat or fuels for plots unaffected by fire. In contrast, plots that did burn (but were otherwise similar in all characteristics) saw marked increases in the abundance of dead wood and associated changes in potential wildlife habitat and fuels. Similarly, live tree basal area and biomass steadily increased over time within unburned reference plots but decreased in burned stands – these estimates of changes in live tree stand density may also be useful to managers seeking a better understanding of the changes to expect (Table 1).

These findings underscore the importance and value of maintaining a national forest monitoring system with consistent data collection protocols such as the annual FIA Program. Only by consistently remeasuring plots on a systematic sampling design can statistically representative information be obtained on how the forest responds to, and evolves with, large scale episodic disturbances like wildfire. As future remeasurements of these plots are delivered, this analysis can be extended to address lingering questions about how long elevated dead wood pools persist, and the consequences for reburns as more of those occur. Answers to these questions over the long term will be especially insightful not only from a habitat quality perspective but also to questions concerning how forest carbon dynamics play out under different pre- and post-fire management scenarios.

4.5. Conclusion

Our longitudinal analysis of westside Pacific Northwest dead wood trajectories shows that fire initiates a pulse of snags on the landscape, which subsequently increases both snag and coarse down wood abundance in unmanaged post-fire landscapes (albeit on different time scales). These dead wood accumulations yield significant increases in potential wildlife habitat, along with a measurable increase in reburn potential when fires burn under severe weather (very dry fuel moisture and 32 km/hr wind speeds). Fine down wood, litter, and duff were mostly consumed by fire and were reduced during the initial post-fire period, leading to decreased potential for reburns under moderate fire weather conditions (similar to the application of prescribed fire to reduce fuel loads). In the subsequent decade, biomass of FW and litter began to increase and subsequently approached levels comparable to unburned reference plots, stimulating an increased reburn potential. Through the creation of standing and coarse down wood, fires increased potential wildlife habitat in unmanaged stands where these resources were allowed to remain on the landscape, as they were on all the plots in this study. However, managers may be faced with a potential tradeoff as increased dead wood inputs yield both potential wildlife habitat and elevated fuels.

The changes in dead wood pools initiated by fire, via consumption and movement of biomass between live, standing dead and down wood pools persist and evolve as interwoven and dynamic trajectories of habitat, fire hazard and carbon flux, for well over a decade post-fire even where management is absent. Understanding these baselines will be important for crafting managed forest trajectories that realize the multiple benefits that may be desired and produced as these highly productive forests respond to future fire disturbances.

CRediT authorship contribution statement

Jeremy S. Fried: Writing – review & editing, Supervision, Resources, Project administration, Funding acquisition, Conceptualization. **Oriana E. Chafe:** Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2025.123003.

Data Availability

Most of the data analyzed for this study are available from the USDA Forest Service Forest Inventory and Analysis (FIA) program. CVS initial installation plot data collected by the USDA Forest Service Region 6 are available within the downloadable PNW-IDB Periodic Inventory Database (https://research.fs.usda.gov/pnw/products/dataandtools/tools/pnw-idb-periodic-inventory-database) and data from FIA plot visits are available from the FIA DataMart (https://apps.fs.usda.gov/fia/datamart/datamart.html). CVS remeasurement data may be requested by contacting the Region 6 Biometrician, currently Skye.Greenler@usda.gov, referencing the "historical CVS monitoring data". Exact plot locations and plot identification numbers used to link CVS and FIA data are not publicly available owing to confidentiality requirements under the Food Security Act.

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