



Fuel Dynamics Following Wildfire in US Northern Rockies Forests

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Accumulation of dead woody material is a critical management concern following wildfires, especially given the possibility of subsequent wildfires. Forest structure and fuel accumulation are largely driven by site climatic conditions, so variability in site conditions is important to consider in management beyond the one-size fits all model for post-fire fuel dynamics. Moreover, dead woody material provides important ecological functions for forested ecosystems. Understanding when surface fuels biomass is highest following wildfires and how these dynamics change after repeated fires affords land management agencies opportunities to adjust fuel reduction strategies without removing large quantities of dead trees that are important in these post-fire landscapes for a variety of ecosystem services, including wildlife, water and nutrient retention, and soil stabilization. Here, we examined how surface fuels and standing dead trees change over time, by burn severity, site climate, and with repeated fires at seven different years-since-fire (1–24 years) across 182 sites from ten wildfires in central Idaho. Downed woody fuel loads were higher on moister and cooler sites. Large-diameter woody fuels saw significant increases after about 14 years post-fire while smaller woody fuels had the highest loads at 20 years post-fire. Surface fuel loads varied by burn severity and years since fires, with the highest loadings at longer years since fire and sites burned in stand replacing fire. Litter and duff had not sufficiently accumulated to compensate for continued decomposition, resulting in lower forest floor biomass through 24 years post fire. Density of snags of all sizes generally declined after 7–9 years, but large-diameter snags were still standing 24 years post fire. Repeated fires resulted in >40% reduction in both surface fuels and standing trees compared to once-burned areas at the same years since fire. Burned landscapes can serve a great ecological benefit by providing an infusion of both standing and downed dead woody debris. Additionally, though some sites burned in stand replacing fires exceed recommended surface fuel loadings, these ecological considerations should be weighed against concerns about reburning potential and fire fighter hazards.

Keywords: fuel loading, wildfires, snags, woody debris, resistance to control, repeated wildfires

Abbreviations: CWD, coarse woody debris; DBH, diameter at breast height; dNBR, differenced normalized burn ratio; DWD, downed woody debris; ESP, environmental site potential; FACTS, forest service activity tracking system; FWD, fine woody debris.

INTRODUCTION

The increasing footprint of wildfires across the western U.S. (Moritz et al., 2012; Stephens et al., 2013; Parks et al., 2016) has raised concerns about surface fuel accumulation on post-fire landscapes. Woody fuel accumulation in burned landscapes following large wildfires is a dominant concern when considering how these sites will burn in the event of a subsequent fire, including both the ecological effects and fire suppression issues that may arise. While reburning at low burn severity has been demonstrated as ecologically beneficial and able to achieve restoration objectives (Larson et al., 2013; Stevens-Rumann and Morgan, 2016), repeated high severity fires present multiple management and ecological concerns. A primary concern of U.S. land management agencies related to reburning is the potential for increased “resistance to control” or a compromised ability for fire crews to build fire lines due primarily to increased surface fuel loading (USDA et al., 1976; Alexander, 2000). As fire seasons lengthen (Jolly et al., 2015) and burned areas increase, there is an increased likelihood of extensive reburning, especially of younger forests (Parks et al., 2016; Harvey et al., 2016). However, little is known about long-term woody fuel dynamics and accumulation in forests across a range of climatic regimes and the impact of repeated wildfires in quick succession.

Though standing dead trees (referred to here as ‘snags’) and woody surface fuels are a safety concern for fire fighters during a wildfire event, these dead trees also serve valuable ecological functions. Snags and downed wood provide wildlife and insect habitat and influence soil moisture, nutrient content, and stabilization (Harmon et al., 1986; Graham et al., 1994; Bull et al., 1997; Jurgensen et al., 1997; Brown et al., 2003). In an attempt to identify areas that had sufficient surface woody fuel to provide these ecological functions without presenting excessive “resistance to control,” Graham et al. (1994) and Brown et al. (2003) developed the concept of “optimum” woody fuel loading for conifer forests of the western U.S. Further, recommendations and guidelines across many national forests address retention of snags during logging operations to promote snag density suitable for bird habitat (e.g., Hutto, 2006). One of the potential ecological benefits of areas burned with high severity is the number of snags, particularly when such snags are large. However, the time period that snags remain standing as opposed to contributing to the surface fuels varies geographically (Everett et al., 1999; Fornwalt et al., 2018).

Previous wildfires can influence subsequent burn severity, but these effects are somewhat short-lived, likely due to gradual accumulation of both surface fuels and live biomass (Parks et al., 2016; Harvey et al., 2016; Stevens-Rumann et al., 2016). While shifts in fire behavior, especially on extreme fire weather days, can be driven by climate and weather factors, the other dominant driver of fire behavior in many western U.S. forests is the quantity of both surface and canopy fuels (Parks et al., 2016; Harvey et al., 2016). Thus, it is critical to understand fuel accumulation in the years following wildfires to identify periods after wildfires where fuel loads are highest and of most concern.

While the US northern Rockies have experienced many large fires in recent decades, due to both climate (Westerling, 2016)

and past forest management (Hessburg et al., 2016), there are no regional studies on post-fire fuels across a range of conditions and time since fire, and little is understood about how fuels change following a subsequent fire. Researchers have examined post-fire fuel dynamics through time in other regions, such as the northwestern (Everett et al., 1999; Dunn and Bailey, 2015) and southeastern U.S. (Sah et al., 2006), California (Eskelson and Monleon, 2018), southern Rockies, and southwestern U.S. (Passovoy and Fulé, 2006; Roccaforte et al., 2012; Fornwalt et al., 2018). In the western U.S., researchers documented surface fuel loads peak 6–18 years post fire (Roccaforte et al., 2012; Dunn and Bailey, 2015; Fornwalt et al., 2018; Eskelson and Monleon, 2018), indicating the first 20 years after an area burns with high severity may be of most concern for managers. In one of the only repeated measures studies in the region, Nelson et al. (2016) found surface fuel accumulation 24 years post-fire in lodgepole pine forests (*Pinus contorta*) of Yellowstone National Park had exceeded levels needed for initiation of a subsequent high severity surface fire and/or active crown fire on more than 63% of plots measured.

Most studies focus on fuel variability in a single forest type within a region (e.g., Dunn and Bailey, 2015; Eskelson and Monleon, 2018; Fornwalt et al., 2018). Very few have examined how fuels complexes vary across severity, forest types, and climate space within a region. Here, we analyzed data collected from 10 large wildfires burned between 1989 and 2007 in forests of the US Northern Rockies to better understand the dynamics of woody fuel accumulation and snag retention following wildfires across a range of mixed-conifer forests. We asked the questions: (1) How do canopy and surface fuels change through time following wildfires at sites burned with different severities and with different site climates in mixed-conifer forests of the northern Rockies? and (2) How are woody fuel dynamics influenced by repeated wildfires in a short period of time (<20 years)?

MATERIALS AND METHODS

Standing and downed fuels data were collected between 2008 and 2016 in Idaho (Figure 1). In all studies, fuels sampling was confined to forested sites with elevations of 1,000 to 2,400 m. Dominant overstory tree species on lower elevation sites included ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*), while higher elevation sites transitioned to dominance by lodgepole pine (*Pinus contorta*), Engelmann spruce (*Picea engelmannii*), and grand fir (*Abies grandis*). Common understory shrubs included ninebark (*Physocarpus malvaceus*), common snowberry (*Symphoricarpos albus*), spirea (*Spiraea betulifolia*), and thinleaf huckleberry (*Vaccinium membranaceum*), with shinyleaf ceanothus (*Ceanothus velutinus*) particularly occurring in areas burned at high severity. Common forbs included common yarrow (*Achillea millefolium*), heartleaf arnica (*Arnica cordifolia*), pearly everlasting (*Anaphalis margaritacea*), and forbs of the genera *Fragaria*, *Lupinus*, and *Penstemon*. Fireweed (*Chamerion angustifolium*) occurred frequently on high severity burned sites. Common grasses were Idaho fescue (*Festuca idahoensis*), Sandberg bluegrass (*Poa secunda*), pinegrass (*Calamagrostis rubescens*), and blue wildrye (*Elymus glaucus*).

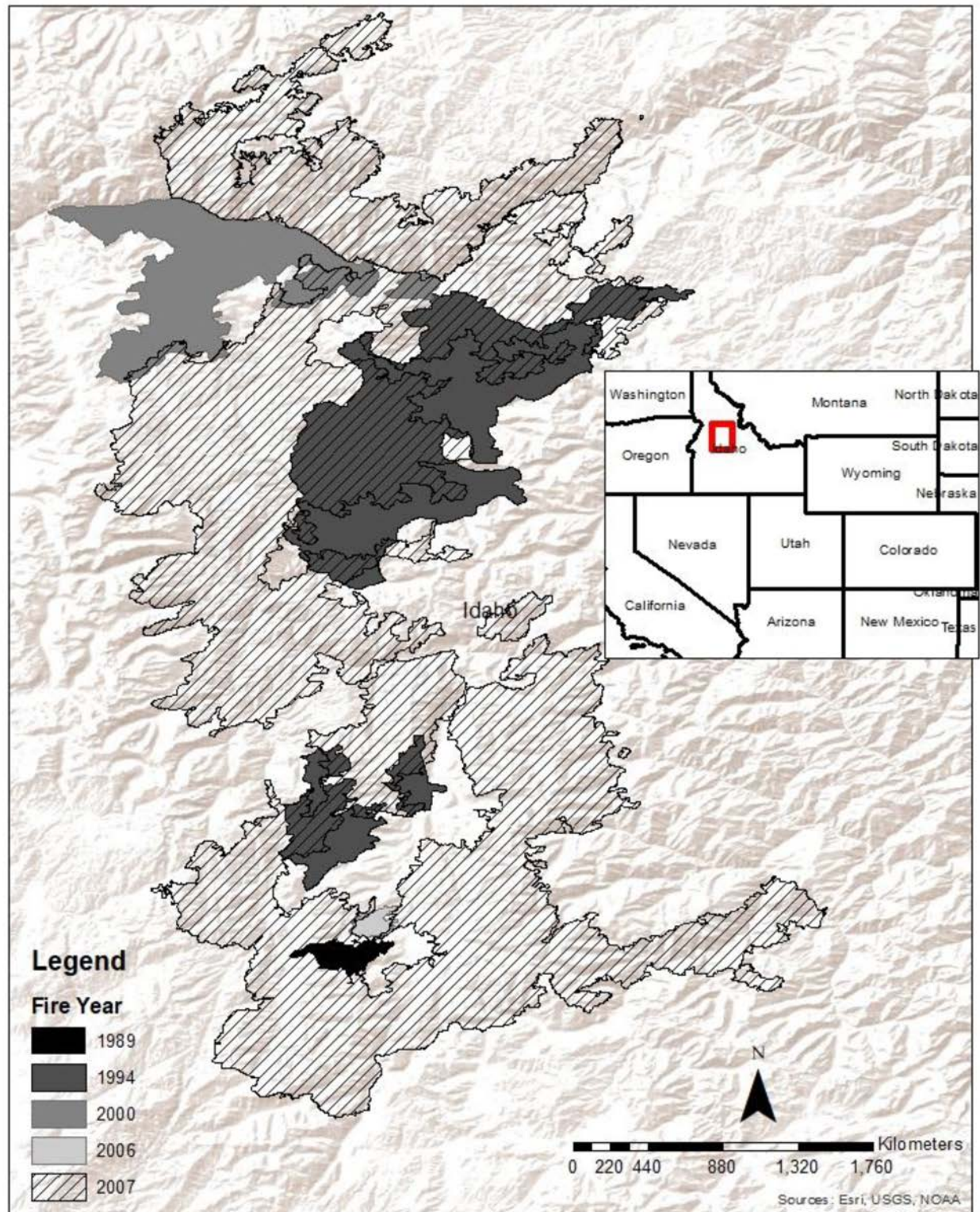


FIGURE 1 | Gray-scale perimeters indicate burn areas we sampled in Idaho, United States. These large fires burned between 1989 and 2007 according to the MTBS database (<http://www.mtbs.gov/mtbs.gov>).

Sedges (*Carex* spp.) were common across burn severity levels. Large, highly flammable shrubs are not a large component of these systems and the large wildfires studied here were largely wind driven wildfires during extreme fire weather conditions (Hudak et al., 2011). We verified the burn severity by considering tree mortality and seedling age at each plot in the field. While our sampled sites likely burned before the 20th century, no large wildfires were recorded since 1950 until our study fires occurred, according to local history of large fires compiled by Gibson et al. (2014). We excluded all sites that had logging prior to data collection based on management action records provided by the National Forests, both through personal communications and the FACTS database (U.S. Department of Agriculture, Forest Service, 2016). Our data was compiled from multiple initial studies and more recent unpublished field work; for a full description of the original studies, see Hudak et al. (2011), Stevens-Rumann et al. (2015), Stevens-Rumann and Morgan (2016), and Strand et al. (2019).

In all sites ($N = 182$), we established a plot at each randomly located point within a stratification of stand replacing fire or non-stand replacing fire excluding sites too close to the road (> 30 m), in order to preclude edge effects. For the purposes of figure display and summary statistics, we grouped burn severity into two categories: “non-stand replacing fire” ($< 90\%$ tree mortality) and “stand replacing fire” ($> 90\%$ tree mortality). Plot size varied slightly: on 158 plots, a 0.04-ha circular plot was established, while on 24 plots measured 9 years post-fire, a 0.02-ha circular plot was established (Table 1). On 138 of the 0.04-ha circular plots, we tallied all trees > 7.62 cm DBH (1.37 m above the ground) as alive or dead, including fallen trees, to estimate percent tree mortality. On 20 additional 1-year-post-fire plots, trees were tallied in 2008 (also 0.04-ha plot), but the minimum DBH was 12.7 cm and no fallen trees were quantified. On 24 plots, each 0.02-ha and circular, we tallied live and dead trees ≥ 10 cm, but no fallen trees were tallied. For all plots, we classified a “large snag” as any dead tree > 30 cm DBH. Live tree basal area and total basal area was calculated; total basal area included both live and dead basal area on each plot. We also determined burn severity based on percent tree mortality. While we estimated tree species to the best of our ability, with the long time-since-fire for some sites, we were unable to determine definitively the species of long-dead trees, thus species-level analyses were not performed. We also could not distinguish reliably between standing dead trees present pre-fire and those created by the wildfire itself.

To quantify downed woody fuels, methods varied slightly by study. We tallied FWD, litter depths, and duff depths following methods outlined by Brown (1974) for most plots and used Keane and Dickinson (2007) methods in 2016 for 24 plots. Dead down woody debris were tallied by size classes (0–0.64, 0.65–2.54, 2.55–7.62 cm diameter) to correspond with 1-, 10-, and 100-h time-lag fuel classes, along 4–6 m, 4–6 m, and 4–10 m of one 30 m fuels transect through plot center, respectively. On 24 1-year-post-fire plots tallied in 2008, all three fuel size classes were tallied along the 7–9 m section of two 15-m fuel transects. These tallies were converted to biomass (Mg ha^{-1}) using Brown’s (1974) allometric equations, and we additionally grouped 1-, 10-, and 100-h time-lag fuel classes together into FWD (< 7.62 cm). Litter and duff

depths were measured 5–10 times at points along the central transect within each plot and averaged for each site; litter and duff depths were also combined for a forest floor depth metric. Litter and duff biomasses were estimated in Mg/ha from the litter and duff depth measures following Eq. (2) in Stephens et al. (2004) for white fir litter ($10.394 \cdot \text{litter_depth_cm}^{1.352}$) and a bulk density of 42.473 kg/m^3 for lodgepole pine duff in Table 1 of Brown et al. (1982).

Slightly different methods were employed between various studies for 1,000-h fuels or CWD (> 7.62 cm). Briefly, on 125 plots, 1,000-h fuels were tallied and diameters taken following methods outlined by Brown (1974) along two 15-m transects (Hudak et al., 2011) or one 30-m transect (Stevens-Rumann and Morgan, 2016). On 24 circular plots, 100 m^2 in size, photoload methods (Keane and Dickinson, 2007) were employed to estimate CWD biomass 9 years post fire. On 13 plots, 6 years post fire, CWD was measured in a 100-m^2 circular subplot located within the center of each plot. Within this subplot the length and diameter at both ends of each piece was measured for every CWD fuel and the fuel load was estimated following Keane et al. (2012). Downed woody debris (DWD) was also calculated as the combined loading of FWD and CWD.

Climate variables indicative of average climate conditions for each site were derived from 30-year (1961–1990) climate normals using thin-plate splines of Hutchinson (2000) per Rehfeldt et al. (2006), available at <http://charcoal.cnre.vt.edu/climate/>. We examined fifteen climate variables (Table 2), choosing for the final models only those that contributed to the best predictive model of the responses. We also considered as predictors topographic slope (%) and aspect; aspect was transformed from degrees to range between 0 (wettest aspect at 30°) and 1 (driest aspect at 210°) following Roberts and Cooper (1989). As a final predictor to indicate burn severity, we considered the dNBR calculated from pre-fire and 1-year post-fire Landsat satellite imagery. Though we used dNBR, Scan Line Corrector (SLC) data gaps specific to Landsat 7 imagery did exist on imagery of some of our fires, therefore, we employed a 5×5 pixel ($150 \text{ m} \times 150 \text{ m}$) window to assign the nearest neighbor dNBR value within 150 m of any sites located in a SLC data gap.

Data Analysis

Pre-fire tree density was reconstructed from standing and down trees post-fire. Variability in pre-fire tree density was tested across years since fire using an Analysis of Variance (ANOVA) to understand if sites within each fire year varied significantly prior to each wildfire.

To answer our first question, we fit generalized linear models (glm function in the stats package in R) using the Gamma (link = “log”) family object (R Development Core Team, 2011), which is recommended for right-skewed, continuous response variables (McCullagh and Nelder, 1989), to predict the fuel and stand attributes using the 15 climate variables, years since fire, slope, transformed aspect, and dNBR as potential predictors. Total sample size across all once burned sites was 182, however, the number of sampled sites at each year-since-fire ranged from $n = 13$ to $n = 39$ (Figure 2). The predictor variables were chosen by initially performing

TABLE 1 | List of studies, with number of sites and surface fuel methods described.

Study	# of plots	Years since fire sampled	Surface fuel methods	Tree overstory plot size	Forest type	Plots with stand replacing fire	Plots with non-stand replacing fire
Hudak et al. (2011)	20	1	Brown (1974)	0.04 ha	Lodgepole pine/Engelmann spruce dominated	16	4
Stevens-Rumann et al. (2015)	13	6	Brown (1974) (FWD) Keane and Dickinson (2007) (CWD)	0.04 ha	Ponderosa pine/douglas-fir dominated	13	0
Stevens-Rumann et al. (2016)	125	7–25	Brown (1974)	0.04 ha	50 ponderosa pine/douglas fir dominated 75 lodgepole pine/Engelmann spruce dominated	65	60
Hudak et al. (2018)	24	9	Keane and Dickinson (2007)	0.02 ha	Lodgepole pine/Engelmann spruce dominated	16	8

All sites listed here burned once and were only used in the GLM analyses.

TABLE 2 | Climate variables derived from Rehfeldt (2006) climate models.

Variable	Description	Units
d100	Julian date the sum of degree-days > 5°C reaches 100	Date
dd0	Degree-days < 0°C	Days
dd5	Degree-days > 5°C	Days
fday	Julian date of the first freezing date of autumn	Days
ffp	Length of the frost-free period	Days
gsdd5	Degree-days > 5°C accumulating within the frost-free period	Days
gsp	Growing season precipitation	Mm
map	Mean annual precipitation	Mm
mat	Mean annual temperature	Celsius
mmax	Maximum temperature in the warmest month	Celsius
mmin	Minimum temperature in the coldest month	Celsius
mmindd0	Minimum degree-days < 0°C	Celsius
mtcm	Mean temperature in the coldest month	Celsius
mtwm	Mean temperature in the warmest month	Celsius
sday	Julian date of the last freezing date of spring	Date
smrp	Summer precipitation: (jul + aug)	mm
smrpb	Summer precipitation balance: (jul + aug + sep)/(apr + may + jun)	Unitless
sprp	Spring precipitation: (apr + may)	mm
winp	Winter precipitation: (nov + dec + jan + feb)	mm

a regression subsets procedure using the leaps package in R. The continuous response variables invariably had right-skewed distributions, and so were natural log transformed before fitting and comparing best subsets models using ordinary least squares multiple linear regression (lm function in the stats package in R). Those selected predictor variables comprising the best subsets model, as defined by the smallest Akaike Information Criterion (AIC) statistic, were then transferred to GLMs specified for a gamma distributional fit and a natural log link function. Finally, the GLMs were tested to see if they could be improved slightly, by either dropping a weakly significant predictor or adding a new predictor that was sufficiently highly correlated to the model residuals, to minimize the AIC statistic between competing GLMs. Model residuals were examined for randomness and normality, and in the case of the surface fuel attributes, compared between the planar transect (Brown, 1974) vs. photoload

(Keane and Dickinson, 2007) fuel sampling methods to confirm that there were no methodological artifacts (i.e., only negligible differences).

To answer our second question regarding the impact of repeated fires on fuel dynamics, only sites burned in the same year as the later fire (for those sites burned twice) were used. Sites were visited 7 years post-fire. In this case, $N = 94$ with $n = 46$ for once-burned sites and $n = 48$ for twice-burned sites; only sites burned in stand replacing fire were considered. Years between fires, which ranged from 1 to 24 years, was previously analyzed and found to have no significant relationship in fuel loading following the second fire (Stevens-Rumann and Morgan, 2016). Twice burned sites are not shown in **Figure 2** but burned in the second fire 7 years prior to sampling. We performed a two-sample t -test to understand the impact of repeated fires versus a single fire on fuel loading and stand structure.

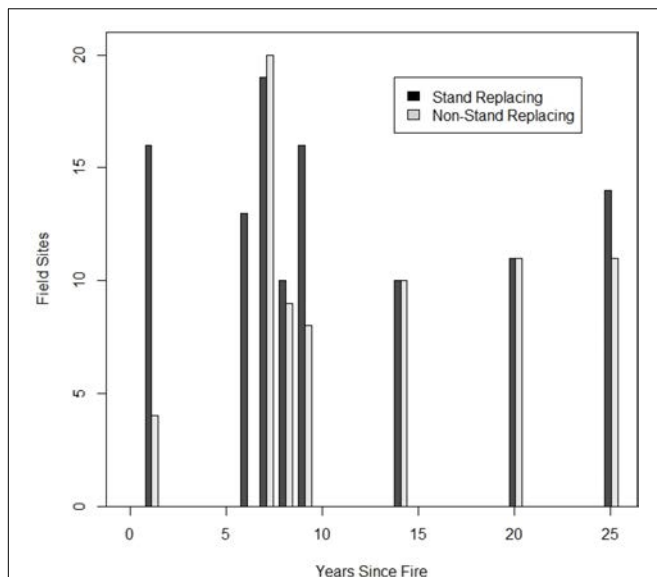


FIGURE 2 | Number of field plots sampled versus years since fire, separated by sites burned in stand replacing fire versus those burned in non-stand replacing fire. All shown plots only burned once in the last >50 years.

RESULTS

Surface Fuels

Total DWD, CWD, and FWD each had a strong positive relationship with years since fire (**Figure 3** and **Table 3**). Further, the biomass of these three woody fuel classes trended toward a significant correlation with dNBR. The strongest climate metrics varied slightly by fuel category, but all three had summer precipitation balance as a significant positive predictor. Also significant across these three surface woody fuel variables was some metric of extreme climate conditions, with all three variables having a negative relationship to the maximum temperature on the hottest month. Thus, those sites that were extremely hot or exposed did not experience as much surface fuel accumulation as those on sites with more moderate climates. Additionally, some metric of the growing season length was significant; for DWD and CWD frost-free period was significant, while Julian date of the first freeze in Autumn significantly predicted FWD (**Table 3**).

Forest floor, litter, and duff biomass were negatively correlated with years since fire and dNBR (**Figure 3** and **Table 3**). Metrics of moisture availability were generally negatively correlated with all three variables.

Canopy Fuels

Pre-fire tree density was not significantly different among the subset of sites for which we were able to quantify this variable ($N = 138$, $F = 1.24$, $P = 0.56$). Post-fire, total basal area was significantly related to years since fire, but large snag density was only marginally related, and live tree density was not related (**Table 4** and **Figure 4**). Total basal area was highest in post-fire years 1–9, with much lower basal area in years 14–25

post-fire (**Table 4** and **Figure 4**). Though total basal area was consistent among sites in early post-fire years, sites burned in stand replacing fire diverged from the rest, especially in later years, due to the live tree component. Total basal area was negatively associated with spring and summer precipitation and positively associated with number of degree days less than 0°C. Mean density of large-diameter snags was also higher in years 1–9, with a reduction in the number of large snags after 9 years post-fire. Large snag density was positively correlated with dNBR. Live tree density and live tree basal area had similar significant predictor variables and both were strongly influenced by dNBR but not years since fire. Years since fire was selected for the GLM model for total basal area because including it lowered the model AIC, even though its contribution to the model was not statistically significant. Live tree density also had a significant positive relationship with spring precipitation and with maximum temperatures in the warmest month (**Table 4**).

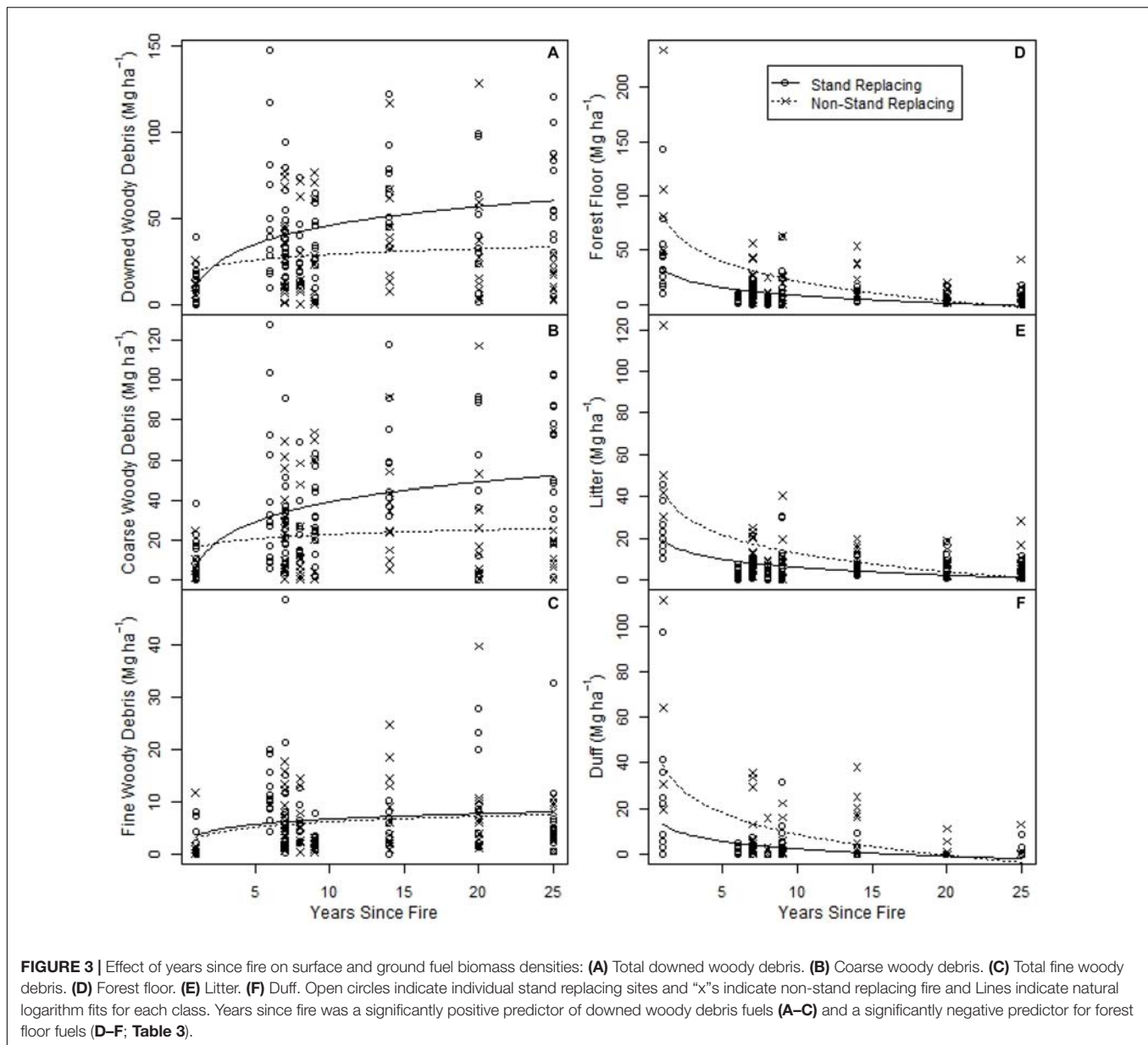
Repeated Fires

FWD and CWD were significantly less on repeated high severity burns than on sites only burned once ($F > 8.0$, $P < 0.01$; **Figure 5**). FWD and CWD did not vary significantly between once- and twice-burned sites at other combinations of burn severity (e.g., high then low, or low then high). Total basal area and large snag density were also significantly lower on sites repeatedly burned with high severity than those sites burned once at high severity ($F = 5.2$, $P = 0.02$ and $F = 6.1$, $P = 0.001$, respectively).

DISCUSSION

Surface fuels increased and snag density declined substantially as years since fire increased, regardless of dominant tree species, and were strongly influenced by climate across the large spatial extent (250,000 ha) within central Idaho. Surface woody fuel accumulation was highest on our sites between 14 and 24 years after the previous fire, suggesting that these years were those with the highest resistance to control of forest fires (Peterson et al., 2015; Harvey et al., 2016; Nelson et al., 2016; Parks et al., 2016), especially following stand replacing fire. Sites with higher precipitation and more moderate climatic conditions had higher surface loadings post-fire than colder and/or drier sites, indicating a relationship between fuel accumulation and site productivity (Schoennagel et al., 2004). Forest floor, litter, and duff dynamics demonstrate that less biomass accumulation and retention occurred on moister sites perhaps due to more decomposition, while warmer sites had more total forest floor biomass than colder sites, perhaps due to higher production and/or less decomposition.

Higher live tree densities persisted on sites burned in a non-stand replacing fire, where little to no delayed tree mortality in the years following fires was observed. Seedlings began to contribute to the overstory basal area on some high severity burned sites at 20–24 years post-fire. Large-diameter snags that provide critical habitat began to decline in density around 7–9 years post-fire, but following this initial decline, the density



of large-diameter snags remained relatively constant through 24 years post-fire. Snags were most abundant in areas burned with stand replacing fire (Roccaforte et al., 2012), and large snags that remain standing after this period may have some longevity on these landscapes, especially on lesser slopes (Everett et al., 1999) and on sites that experience less extreme temperatures; although, these variables were only strong predictors of total basal area, not large snags alone.

By comparison, sites burned twice at high severity had significantly lower surface fuels, tree density, and basal area than sites burned only once in 24 years. This demonstrates that resistance to control is low in these reburned areas, but these sites also lack similar post-fire habitat due to low snags and low surface fuel loading. All these reburned sites burned twice at high severity, thus there is a continued and persistent

lack of accumulated fuel. These attributes, when combined with low understory plant cover and limited tree regeneration, may indicate long-term ecosystem shifts away from similar forest structures (Donato et al., 2016; Stevens-Rumann and Morgan, 2016). However, it is important to distinguish between sites burned repeatedly at high severity, versus areas burned repeatedly at low to moderate severity which has shown to maintain or enhance historical forest structure in many western U.S. forests (Larson et al., 2013; Stevens-Rumann and Morgan, 2016).

High Surface Woody Fuels in More Years Since Fire

The high CWD biomass 14–24 years post-fire combined with the FWD biomass demonstrate that this post-fire time period has

TABLE 3 | Results of generalized linear models (GLM) of site climate variables, topography, years since fire, and severity (dNBR) to predict total downed woody debris, coarse woody debris, fine woody debris, forest floor biomass, litter biomass, and duff biomass.

Variable	Estimate	SE	t-value	P-value
Down woody debris				
(Intercept)	1.26E+02	3.02E+01	4.185	<0.001
Years since fire	5.97E-02	1.03E-02	5.782	<0.001
d100	-5.65E-01	1.27E-01	-4.446	<0.001
dd0	1.31E-02	3.43E-03	3.818	0.0002
ffp	3.21E-01	1.38E-01	2.332	0.021
gsdd5	-2.39E-02	9.07E-03	-2.633	0.009
mmax	-2.43E-01	5.17E-02	-4.704	<0.001
smrpb	1.60E-01	6.34E-02	2.527	0.012
dnbr	4.41E-04	2.34E-04	1.89	0.060
Coarse woody debris				
Years since fire	6.21E-02	1.20E-02	5.197	<0.001
d100	-6.29E-01	1.47E-01	-4.271	<0.001
dd0	1.49E-02	3.97E-03	3.767	<0.001
ffp	3.69E-01	1.59E-01	2.319	0.021
gsdd5	-2.79E-02	1.05E-02	-2.66	0.009
mmax	-2.67E-01	5.99E-02	-4.457	<0.001
smrpb	1.46E-01	7.34E-02	1.984	0.049
dNBR	4.49E-04	2.70E-04	1.659	0.099
Fine woody debris				
(Intercept)	24.62730	18.14544	1.357	0.176
Years since fire	0.042484	0.011076	3.836	<0.001
fday	-0.11118	0.073997	-1.503	0.135
mmax	-0.21717	0.06408	-3.389	<0.001
mtwm	0.31741	0.093101	3.409	<0.001
smrpb	0.213877	0.064503	3.316	0.001
dNBR	0.000437	0.000252	1.734	0.085
Forest floor biomass				
(Intercept)	-1.23E+01	2.78E+00	-4.447	<0.001
Years since fire	-3.37E-02	1.26E-02	-2.676	0.008
dd0	-2.55E-02	3.92E-03	-6.523	<0.001
gsp	-1.17E-01	1.83E-02	-6.394	<0.001
mmindd0	1.38E-02	2.23E-03	6.189	<0.001
sprp	3.26E-04	5.04E-05	6.464	<0.001
Slope	-1.88E-02	8.61E-03	-2.182	0.030
dNBR	-9.29E-04	2.66E-04	-3.49	<0.001
Litter biomass				
(Intercept)	-1.15E+02	3.60E+01	-3.181	0.002
Years since fire	-2.31E-02	1.08E-02	-2.144	0.033
d100	4.01E-01	1.49E-01	2.69	0.008
dd0	-3.84E-02	1.34E-02	-2.862	0.005
dd5	3.40E-02	1.04E-02	3.289	0.001
ffp	-1.95E-01	7.42E-02	-2.622	0.010
gsp	-1.38E-01	2.41E-02	-5.729	<0.001
map	-3.69E-02	1.38E-02	-2.665	0.008
mmindd0	2.44E-02	1.12E-02	2.174	0.0311
sprp	5.80E-04	1.11E-04	5.23	<0.001
Slope	-1.03E-02	7.33E-03	-1.404	0.162
dNBR	-6.44E-04	2.52E-04	-2.556	0.011
Duff biomass				
(Intercept)	1.29E+02	2.02E+01	6.372	<0.001
Years since fire	-5.85E-02	1.63E-02	-3.595	<0.001

(Continued)

TABLE 3 | Continued

Variable	Estimate	SE	t-value	P-value
fday	−4.47E-01	7.83E-02	−5.701	<0.001
mat	1.20E-01	2.57E-02	4.683	<0.001
smrpb	−5.10E-01	8.95E-02	−5.7	<0.001
Slope	−2.79E-02	1.21E-02	−2.31	0.022
dNBR	−8.66E-04	3.70E-04	−2.343	0.020

A single GLM predicted all 10 responses. P-values of significant predictors are italicized.

TABLE 4 | Results of generalized linear models (GLM) of site climate variables, topography, years since fire, and severity (dNBR) to predict total basal area, live basal area, large snag density, and live tree density.

Variable	Estimate	SE	t-value	P-value
Total basal area				
(Intercept)	−5.37E+01	2.24E+01	−2.401	0.017
Years since fire	−2.06E-02	6.86E-03	−3.007	0.003
dd0	−1.68E-02	6.15E-03	−2.727	0.007
fday	1.57E-01	7.12E-02	2.203	0.028
gsp	3.67E-01	1.31E-01	2.793	0.006
mmindd0	1.34E-02	4.63E-03	2.897	0.004
smrp	−7.60E-04	2.67E-04	−2.843	0.005
sprp	−4.18E-04	1.57E-04	−2.656	0.009
Live basal area				
(Intercept)	−3.47E+02	9.59E+01	−3.615	<0.001
dd0	−4.61E-02	1.31E-02	−3.529	<0.001
fday	2.42E-01	1.44E-01	1.681	0.095
gsp	−2.49E-01	7.17E-02	−3.473	<0.001
mmax	4.89E-01	1.70E-01	2.881	0.004
mmindd0	4.98E-02	1.39E-02	3.57	<0.001
smrpb	6.76E-01	2.92E-01	2.311	0.021
sprp	1.04E-03	2.68E-04	3.872	<0.001
dNBR	−2.19E-03	3.06E-04	−7.151	<0.001
Large snag density				
(Intercept)	7.48E+00	2.20E+00	3.397	<0.001
Years since fire	−3.40E-02	1.94E-02	−1.755	0.081
winp	−1.33E-05	8.54E-06	−1.559	0.121
dNBR	1.23E-03	4.64E-04	2.658	0.009
Live tree density				
(Intercept)	−3.39E+02	1.17E+02	−2.904	0.004
dd0	−7.34E-02	1.72E-02	−4.263	<0.001
fday	5.21E-01	1.93E-01	2.697	0.008
gsp	−3.00E-01	4.04E-02	−7.43	<0.001
mmax	3.44E-01	2.27E-01	1.52	0.130
mmindd0	6.61E-02	1.86E-02	3.56	<0.001
sprp	1.08E-03	2.21E-04	4.893	<0.001
Slope	−2.87E-02	1.57E-02	−1.832	0.069
dNBR	−4.08E-03	4.40E-04	−9.28	<0.001

A single GLM predicted all 10 responses. P-values of significant predictors are italicized.

the highest susceptibility to burning again in a stand replacing fire, as others have found (Thompson et al., 2011; Parks et al., 2016; Harvey et al., 2016). During this time, the abundant coarse fuels are especially susceptible to prolonged smoldering with high biomass consumption, heat release, and soil heating (Byram, 1959; Rothermel, 1972; Sikkink and Keane, 2012).

At year one post fire, higher mean CWD biomass was present on sites burned in non-stand replacing fire, likely due to lower fuel consumption of downed material during the wildfire compared to sites burned with high severity. However, by 14 years post fire, CWD loadings on plots burned with stand replacing fire significantly surpassed the CWD loadings in areas burned

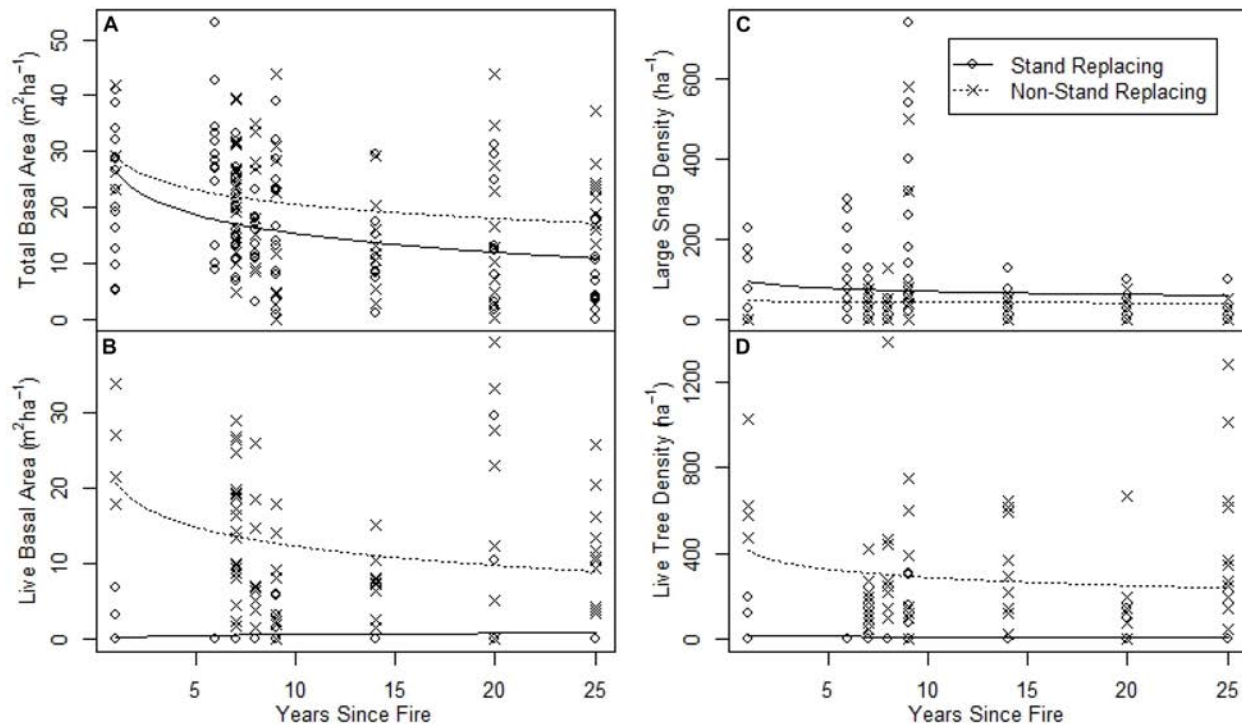


FIGURE 4 | Effect of years since fire on trees. **(A)** Total basal area (live and dead standing trees). **(B)** Live tree basal area. **(C)** Large-diameter snag density. **(D)** Live tree density. Open circles indicate individual stand replacing sites and "x"s indicate non-stand replacing fire and lines indicate natural logarithm fits for each class. Years since fire is a significant predictor of total basal area but only marginally significant for large-diameter snag density and not significant for live tree density (Table 4).

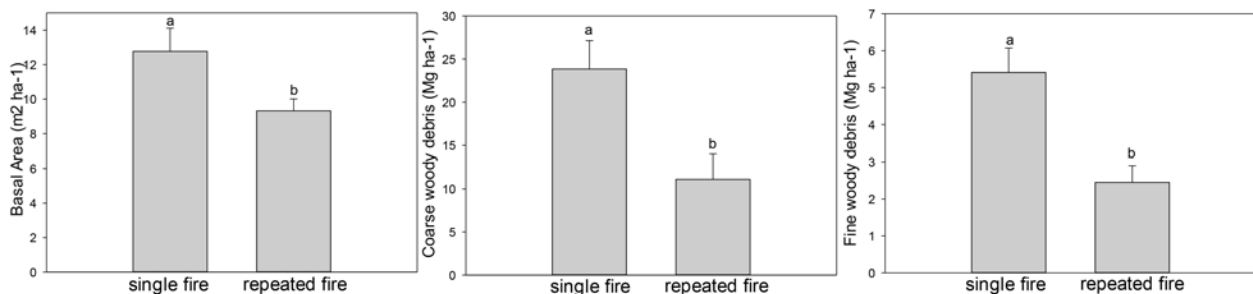


FIGURE 5 | Effect of a single wildfire compared to repeated wildfires measured 7 years following the most recent fire on tree basal area (includes both live and dead trees), coarse woody debris, and fine woody debris. Letters indicate significant differences at the $\alpha = 0.05$ level.

with non-stand replacing fire, and CWD loading remained high through 24 years post fire. The higher CWD loadings in stand replacing fire areas compared to sites burned with non-stand replacing has been observed elsewhere (Stevens-Rumann et al., 2012; Fornwalt et al., 2018) and can largely be attributed to high tree mortality rates and snag fall rates through time, especially following stand replacing fire.

Downed woody debris is important for numerous ecosystems services such as wildlife habitat and improved nutrient cycling in post-fire environments (Harmon et al., 1986; Graham et al., 1994), however, this benefit must be weighed against the risk

presented by having heavy flammable fuels that increase the resistance to control and promote potential soil heating in subsequent wildfires (USDA et al., 1976; Sikkink and Keane, 2012). Our greatest annual mean CWD biomass of 70.4 Mg ha^{-1} in year 14 and FWD biomass of 10.1 Mg ha^{-1} in year 20 on stand replacing fire sites were well above those found by Roccaforte et al. (2012) in Arizona and in dry conifer forests, and more similar to the loadings observed by Dunn and Bailey (2015) and Peterson et al. (2015) in eastern Oregon and Washington. In sites burned with stand replacing fire, for 14–24 years post fire, CWD loadings exceeded the desired

CWD threshold suggested by Brown et al. (2003). Thus, these areas warrant special consideration in the event of subsequent fires and careful monitoring in subsequent years if loadings remain high. This concern about high fuel loading is likely restricted to those areas with high tree mortality, as loadings were significantly lower in areas burned with non-stand replacing fires, as others have found (Stevens-Rumann et al., 2012; Fornwalt et al., 2018). In cases of mixed-conifer forests burning at lower severities, a single fire may not warrant concern for excessive fuel accumulation, resistance to control, and the need for post-fire harvesting (Ritchie et al., 2013; Campbell et al., 2016). In areas of high CWD loading, prescribed fire during times of moderate to high fuel moistures could help reduce the overabundance of fuel without the soil heating effects and other issues arising from reburning these areas in wildfire conditions (Graham et al., 1994).

Interestingly, the surface woody fuel accumulation through time contrasts with the decline in litter and duff biomass through time. Without the establishment of a dominant overstory, litter and duff appear to be continuing to decompose at faster rates than new litter deposition into the system, particularly where trees are absent (i.e., stand replacing fire), resulting in a negative relationship through time. Over longer time periods one would expect this relationship to switch with the establishment of woody shrubs or trees that provide litter input, but on these sites even 24 years post-fire, the lack of an overstory continues to result in low litter and duff biomass. A similar result was recently found in drier pinyon-juniper woodlands (Wozniak et al. unpublished).

High Snag Fall by Seven to Nine Years Post Fire

Total basal area, comprised of both living and dead standing trees, gradually declined over time as trees fell (Roccaforte et al., 2012; Dunn and Bailey, 2015), with both stand replacing and non-stand replacing fires resulting in a reduction of standing trees in the first decade post fire. This pattern was especially pronounced and continued throughout the 24-year chronology in areas burned with stand replacement. Large-diameter snags are of particular interest post disturbance and may remain standing much longer than smaller-diameter trees (Graham et al., 1994). We observed a decline in large snags around 7–9 years post fire, similar to the pattern observed by others for Douglas-fir and ponderosa pine (Everett et al., 1999; Russell et al., 2006; Ritchie et al., 2013). However, after 7 years post fire, large snag basal area did not continue to substantially decline, unlike total tree basal area. We observed 15–36% of fire-killed snags still standing 20 years post fire, unlike the ~90% snag fall rate at about 20 years observed by Dunn and Bailey (2015). This may be a result of both the low quantity of large snags (<10–15% of the standing trees above 30-cm DBH) and site conditions that may allow for higher snag retention after the initial loss of both small and large-diameter, fire-weakened trees. Though large-diameter snags decreased with year since fire, even 25 years post fire many of the largest snags (mean of 17 trees ha⁻¹ of trees 30 cm or greater) are still standing and may serve as habitat

for cavity-nesting birds and raptors. Some snags may persist longer in low to moderate severity burned sites due to less wind induced fall patterns as a result of a less open stand, thus the high variability in large snag density 9 years post-fire. There is also likely variability by species with some species remaining standing for longer periods of times than others (Ritchie et al., 2013), though we did not examine species differences in part because long dead snags (10+ years) is difficult to definitively identify to species in mixed conifer forests. This highlights great regional variability in surface fuel and snag dynamics, as our results vary from those in other regions.

Site climate strongly controlled surface fuel accumulation and smaller snag fall rates. In many ways, this is a proxy for forest type and site productivity. On sites with higher productivity, such as those with higher precipitation and less extreme temperatures, we observed higher surface fuel accumulation and greater retention of large snags. This is often expected with short-term plant responses post fire (e.g., DeSiervo et al., 2015; Stevens et al., 2015), but has not been previously quantified for fuel complexes.

Reduced Fuels in Twice-Burned Sites

While fires in areas burned once both consume and create dead wood, subsequent wildfires only consume woody material and do not contribute to the creation of large woody fuel (Donato et al., 2016). In our study, this was so regardless of the time since previous fire or time between wildfires. Areas reburning with high severity 1–23 years after an initial high severity fire reduced CWD biomass by 47% and FWD biomass by 45% compared to a single high severity fire in the same year, similar to what others have found (Brown and Johnstone, 2012; Donato et al., 2016). However, this reduction in fuels was only observed on sites that experienced consecutive fires that burned with high severity (Stevens-Rumann and Morgan, 2016). Repeated high severity fires in quick succession (<24 years apart) were and continue to be a relatively small portion of the landscape. Of the 12 study fires presented here, repeated high severity fires comprised 3% of the area within these fires (Stevens-Rumann et al., 2016), as others have found across the western U.S. (Teske et al., 2012; Harvey et al., 2016; Morgan et al., 2017). Thus, across landscapes, only small areas will experience this significant drop in surface fuels and standing snags. Conversely, a subsequent low severity burn following a high severity burn likely results in sustained high fuel loads due to minimal consumption of large woody fuels and the contribution of small-diameter regenerating trees that following the second fire add to the dead biomass (Nelson et al., 2016; Stevens-Rumann and Morgan, 2016). Thus, sustained high fuel loads 14–24 years post-fire are likely regardless of a single or repeated wildfire event.

The basal area and densities of large snags were significantly lower on repeated burn sites 7 years post fire compared to once-burned sites at these same years since fire. This demonstrates a change in the longevity of standing fuel following repeated wildfires. This is likely due to both consumption in the second wildfire and the weakening of the already dead trees that will likely, if not consumed, fall more readily post fire (Donato et al., 2016). The reduction in both standing and down wood fuel

in repeated burn sites corresponds to changes in ecosystem services observed in repeatedly disturbed ecosystems (Harmon et al., 1986).

Limitations

Post fire fuels were highly variable, as many prior studies have found. Multiple researchers have presented limitations of the Brown (1974) fuels methods and photoload methods (e.g., Keane et al., 2012; Vakili et al., 2016). In the future, improved sampling techniques that better account for the heterogeneity of fuels will improve our understanding of long-term surface fuel dynamics (Vakili et al., 2016). Given that our primary concern was understanding the comparative differences and not the absolute loadings, there is some evidence that the Brown (1974) methods may be adequate (Keane et al., 2012), but do not align well with other methods for absolute comparisons. The limitations of the Brown (1974) methods were an unavoidable issue in this case, as the first data collection occurred in 2008 prior to the publication of many of these methodological critiques and before improved methods were available. However, when isolating data only collected with similar methods, dominant drivers of biomass and stand structure characteristics did not change.

Further, we used multiple fires across a large geographic area comprised of multiple forest types for a broad range of fuel and stand conditions from which to assess the effect of years since fire. The clustering of our sample site distribution temporally, from 6 to 9 years post fire, made our dataset particularly sensitive to this transition period from predominantly standing snags to predominantly fallen logs. Nevertheless, we were limited temporally to large fire years and may have missed some of the fuel dynamics that occurred during those peak CWD and FWD loading years (9–20 years post-fire), during which time period we only had observations from 14 and 20 years post-fire. While our chronosequence of fires can help inform managers on fuel dynamics, repeated measures studies of the same plots is a more robust way to verify snag fall rates, surface fuel accumulation, and fuel decay (Ritchie et al., 2013; Nelson et al., 2016). Finally, it is not well known how burned fuels age and transition between fuel size classes, decay and, ultimately, contribute to litter and duff. This may also be highly variable between species and is not well understood. We can observe these patterns of change somewhat within this 25-year post-fire window, but much longer intervals are needed to truly understand burned wood decay.

CONCLUSION

Surface fuels and thus resistance to control peaks 14–20 years post-fire as snags fall and CWD and FWD are at their highest. Repeated wildfires alter woody fuel dynamics, reducing surface fuels by more than 40% and basal area by 27% compared to forests burned once. In these mixed-conifer forests of central

Idaho, representative of much of the northern Rockies, fuel accumulation appears to only exceed recommended surface fuel loadings for several years, however, longer-term studies are required to fully understand fuel decay until the time at which fuels don't present excessive risk. The ecological value of both the standing and downed dead material is critical to forested ecosystems for numerous ecosystem functions and should be considered prior to any post-fire fuels reduction action. As area burned continues to increase in the coming decades, it is important to consider when and where these previously burned stands will aid in suppression efforts, and when and where resistance to control and potential for severe reburn arises.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are publicly available. This data can be found here: <https://www.frames.gov/catalog/25470>.

AUTHOR CONTRIBUTIONS

CS-R led data collection for half of the presented data, analyzed part of the dataset, and wrote the manuscript. AH led data collection for the other half of the presented data, analyzed part of the data set, and assisted in the manuscript writing and editing. PM provided funding for original data collection efforts and assisted in the manuscript writing and editing. AA assisted in data collection and manuscript editing. ES assisted in data collection and manuscript editing.

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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