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Forest structure and biomass reflects the variable effects of fire and land use 15 and 29 years following fire in the western Cascades, Oregon

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ABSTRACT

The mixed severity fire regime of western Oregon forests creates a complex post-fire landscape mosaic with patches of low, moderate and high overstory tree mortality. Conversion of old-growth forests into plantations and post-fire salvage logging are widespread land uses that dramatically affect structure, biomass and carbon stocks. Few studies, however, have quantified the complex responses to wildfire and land management (i.e. logging and post-fire salvage logging) over long time periods. We quantified total aboveground biomass and composition in forest stands following low, moderate, and high severity fires 15 (2002 Apple Fire) and 29 years (1991 Warner Creek Fire) following fire in low elevation, old-growth forests dominated by Douglas-fir (*Pseudotsuga menziesii*). We also sampled post-fire responses in forest plantations (harvested prior to fire) and salvage-logged sites (harvested after fire) in the same fires. Fire severity had dramatic effects on the partitioning of total aboveground biomass (TAGB). Most of the TAGB in high severity fires was sequestered in dead trees (> 43%) and downed wood (> 29%) while live trees comprised the largest component of TAGB (> 62%) in low severity fires. In spite of differences in overstory mortality, there was no significant difference in the TAGB between the low, moderate and high severity fires 15 years following fire (Apple Fire). Similarly, there was no significant difference between the low and high severity burns 29 years following fire (Warner Creek Fire). Managed forests (salvage and plantations) had significantly lower post-fire aboveground biomass and carbon storage that the natural forests. The TAGB of salvage logged sites was 49% and 42% that of the high severity sites at the Apple Fire and Warner Creek Fire, respectively. The mean TAGB of plantations was lowest of all fire and land use scenarios. At the Warner Creek Fire, TAGB of the plantations were < 30% of that of the high severity fire sites (e.g. 326 and 984 Mg ha^{-1}, respectively). This equates to a difference in aboveground carbon in the managed compared to the natural stands of 553 Mg CO_2e ha^{-1} at the Apple Fire and 781 Mg CO_2e ha^{-1} at the Warner Creek Fire. This research highlights the management tradeoffs involving values relating to carbon storage and wood harvest following fires.

1. Introduction

Some of the world's most productive forests are found in the low to mid-elevations of the Cascade Mountains of the Pacific Northwest (Keith et al., 2009). Climate, topography and vegetation combine to create a complex mixed severity fire regime that characterizes these landscapes (Agee and Huff, 1980; Perry, 2011; Tepley et al., 2013). The post-fire landscapes of forests in mixed severity regimes are typified by a mosaic of stands of varying fire-induced mortality and successional stage and hence high landscape heterogeneity. For example, on the 3723 ha Warner Creek Fire 40% of the burn area was stand-replacing fires with > 90% crown mortality (high severity), 36% was understory burned or not burned at all (< 25% crown mortality - low severity) and 24% had some intermediate level of crown mortality (moderate severity; Fig. 1; Brown et al., 2013). The range of fire effects imposed by mixed-severity fires is an important factor influencing structural diversity in these forests of the Pacific Northwest (Dunn and Bailey, 2016). This pattern results in a varied succession of both vegetation and fuels and has also been hypothesized to result in a differential reburn potential. Interactions between fire and land-use likely further constrain regional carbon storage in the Western Cascade Mountains (Hudiburg et al., 2009). Land management, fire suppression practices,

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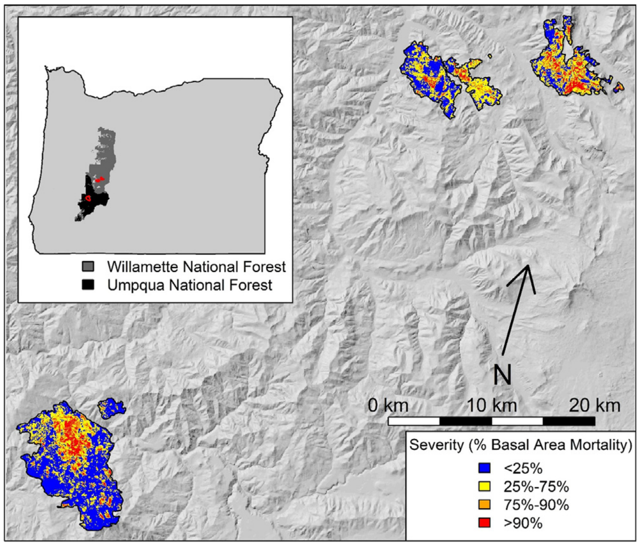

Fig. 1. The fire perimeters and burn severity (% live tree basal area loss) of study sites in mixed severity fire regime/forest landscapes of the Willamette and Umpqua National Forests, Oregon (Reilly et al., 2017).

and climate change have altered the natural structure, function, and resiliency of forests in the Cascades (Perry, 2011).

A number of studies have examined short term differences in biomass responses to fire in both natural and managed forests with mixed severity regimes (Brown et al., 2013; Donato et al., 2006, Campbell et al., 2016). However, data are lacking on the dynamics of change associated with differences in fire severity and management alternatives in the decades following fire (but see Peterson et al., 2015, Dunn and Bailey, 2016). Few studies have examined mid-term dynamics (i.e., 15–29 years post fire) of fuels or vegetation and reburn potential in the region, especially in the wet forest types (e.g., the *Tsuga hetrerophylla* zone) of this study (Halpern, 1989). In forests with no previous history of logging, we quantified composition and biomass 15- and 29-years following fire of differing severities. In these same fire areas, we also sampled managed forests that were plantations at the time of fire or salvage-logged sites which were burned forests that had been logged shortly following the fires.

Our objectives were to (1) quantify how fire severity influences vegetation succession and biomass partitioning 15 and 29 years following fires; and (2) test for differences in post-fire succession and fuel dynamics between managed forests (plantations and salvage logged sites), and natural forests subjected to varying fire severities. We hypothesized biomass composition would reflect the differences in fire severity – the majority of biomass in high severity fires would be comprised of dead vegetation (snags and downed wood) while the majority of the biomass in the low severity sites would be live vegetation (large conifers). The proportion of live large trees in moderate severity-burn forests would be intermediate to that of the low and high severity sites. We also hypothesized that trajectories of total aboveground biomass, revegetation patterns, and residual large live and dead trees will be influenced by fire severity, pre-fire management (plantations vs. natural stands) and post-fire management (salvage logging).

2. Methods

2.1. Study site(s)

The study was located within three large complex fires in the Cascade Mountains, Oregon on the Willamette and Umpqua National Forests (Fig. 1). All fires burned in forests that were predominantly located in the *Tsuga heterophylla* potential natural vegetation zone (Franklin and Dryness, 1973). The *T. heterophylla* zone is the most extensive vegetation zone in western Washington and Oregon and the most important in terms of timber production. These forests develop in wet, mild, Maritime climates with a precipitation range of ≈1500 – 3000 mm (Franklin and Dryness, 1973). Prior to the fires, sites were mostly dominated by *Pseudotsuga menziesii* and *T. heterophylla*.

The first study area is a complex of two fires that occurred in close proximity to each other near the town of Oakridge, Oregon, USA (Fig. 1). The Warner Creek Fire burned 3631 ha on 10–23 October 1991, and the Shady Beach Fire burned 3180 ha in 1988. Salvage logging was widespread within the perimeter of the Shady Beach Fire while no salvage occurred within the Warner Creek Fire. Given similar composition and environments, these two fires provided a paired experiment to test effects of management on long term post-fire composition and structure. Hereafter, we refer to data from these fires as the Warner Creek Fires. The Maritime climate produces wet, mild winters and warm, dry summers. Approximately 90 percent of precipitation occurs October through March, mostly as rain, with snow accumulating at higher elevations (Brown et al., 2013).

The second study area was the Apple Fire, which burned 7745 ha within the Umpqua National Forest from 16 August to 8 September 2002. Within the Apple Fire, 15% had burned at high severity, occurring in both plantations and natural forest stands. Post-fire management included some salvage logging thus also allowing us to sample this land use at this site. Precipitation ranges from ≈1500 - to 1900 mm/year and falls mostly as rain from October through May (PRISM Climate Group, 2004). Our two study areas represent both wetter (Warner Creek Fires) and the drier (Apple Fire) conditions found within the *T. heterophylla* vegetation zone.

2.2. Field sampling design

At both the Warner Creek Fires and the Apple Fire, we determined forest composition, tree density, basal area and total aboveground biomass (TAGB or fuel mass) in unlogged late successional forests (old-growth structure *sensu* Davis et al., 2015) that had burned under three fire severities based upon overstory mortality: (1) low severity sites where < 25% of the overstory had been killed by fire; (2) moderate severity where 25–75% of the overstory had been killed by fire and (3) high severity sites where > 75% of the overstory had been killed by fire. Prior to fire, all of the natural forest sites had no history of logging (old growth forests). Also, we sampled sites that were young plantations (clear-cut logged 11–34 years before the fire, but had burned during the fire and replanted shortly thereafter). Finally, we sampled sites that had been old growth forests at the time of fire but were harvested shortly following fire (salvage logged sites). Site selection was based upon stratifying sites based upon the fire severity maps for each of the fires. (Reilly et al., 2017, Fig. 1). Upon arriving to the stand, and prior to sampling, we confirmed the site was the targeted fire severity or land use class based upon overstory structure (percentage of live and dead trees), and for the low severity sites, the presence of char on the surviving boles. At each of the 2 sites, we sampled 5 different stands of each of these fire/land use scenarios except for the Apple low severity and Apple salvage cover types where we only sampled 3 and 4 stands, respectively. The lower sampling effort is an artifact of large wildfires occurring near our study sites at the time of sampling necessitating an evacuation of the study area before we could complete the field sampling. At the Warner Creek Fires, we sampled 6 high severity burned sites and 4 salvage-logged sites.

2.3. Field measurements

At each plot, vegetation composition and total aboveground biomass including live and dead vegetation, were determined in a manner similar to that used in the USDA Forest Service Forest Inventory and Analysis (FIA) program (Bechtold and Patterson, 2005) and other studies that have quantified forest structure and biomass throughout the

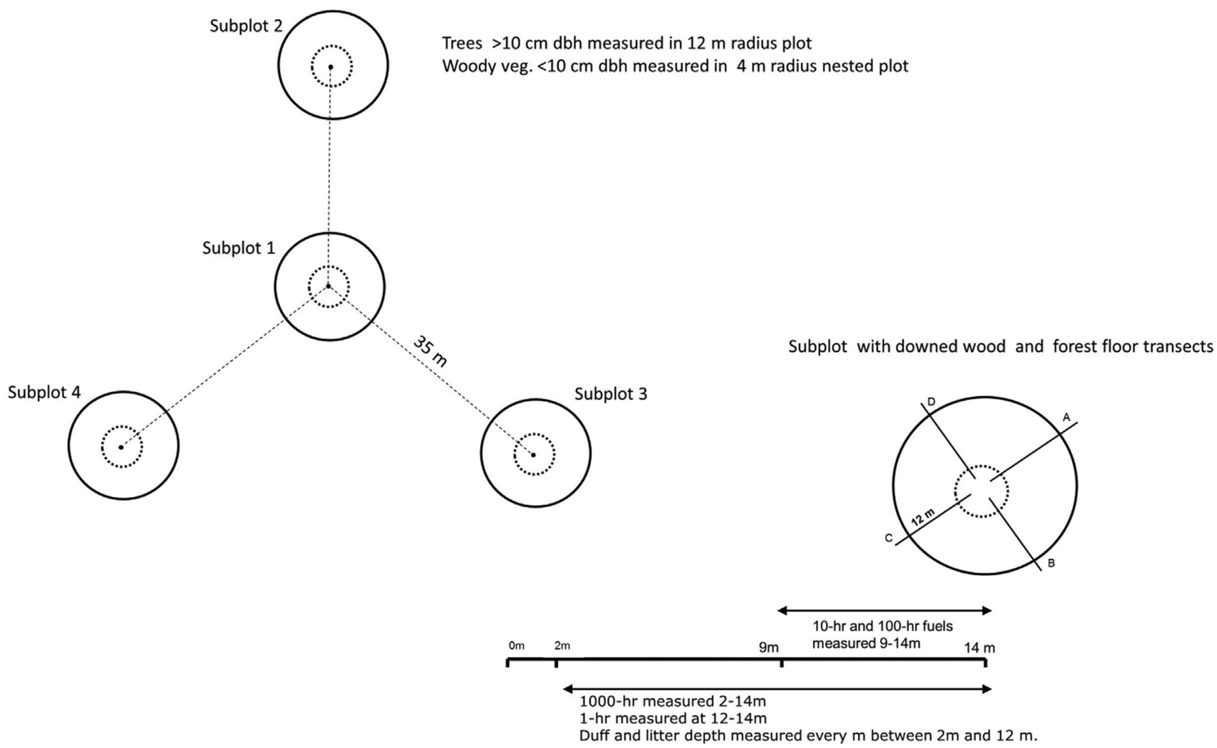

Fig. 2. Experimental plot layout for quantification of species composition and total aboveground biomass of forests. Components include live and dead trees (measured in circular plots depending upon size), downed wood (measured in 4 transects per subplot, and litter and duff mass (measured along fuel transects).

region (Smithwick et al., 2002). Within each sampled forest stand, live and dead woody vegetation were measured in 4 nested circular plots (Fig. 2).

2.3.1. Downed wood

Downed wood was sampled using the planar intersect technique (Van Wagner, 1968; Brown, 1974). Within each subplot of each sampled stand, 4 downed wood transects were systematically established to ensure sample dispersion through the sampled stands. (Fig. 2). The diameter classes used to partition downed wood were 0–0.64 cm, 0.65–2.54 cm, 2.55–7.5 cm, and sound and rotten classes for particles that were > 7.6-cm; Table 1). Lengths of the sampling plane varied among the wood size classes: 2 m for wood particles < 0.64 cm diameter, 5 m for wood 0.65–2.54 cm diameter; 5 m for wood 2.55–7.6 cm diameter; and 12 m for all coarse wood > 7.6 cm diameter. The diameter of each piece of wood > 7.6 cm diameter intercepting the plane was measured to the nearest half centimeter and was determined to be either sound or rotten (decayed). For the three downed wood size classes < 7.6 cm diameter, a quadratic mean diameter (Table 1) was utilized for equations through measurement of 50–100 particles of each size class at each site. Thereafter, for these classes, we counted the number of particles that intersected the sampling plane. Thirty to 50 randomly collected samples of each size class were measured for specific gravity (particle density; Table 1). The slope of each transect was measured in the field using a clinometer to correct for slope (Brown, 1974).

2.3.2. Trees and shrubs

Trees were stratified based upon the diameter of all trees at 1.3 m height (dbh). Trees < 10 cm dbh (which predominantly established following fire) were measured in 4 subplots with a 4 m radius. Trees > 10 cm in diameter are those that were predominantly present at the time of burning and were measured in larger circular plots with a 10 m radius. To be included, trees had to have > 50% of their rooted base within the plot. For live trees, field measurements included species identification and diameter at breast height. Biomass of trees and shrubs were determined by applying species specific allometric equations from BIOPAK (Means et al., 1994) and Chojnacky et al. (2014; Supplementary Table 1). We also determined density and basal area of the live trees from these measurements.

We partitioned standing dead trees into three classes based upon their degree of decay. Biomass of each dead tree decay class was calculated differently. Class 1 dead trees were those that had recently died with fine branches still attached. This only included trees that had died well after the fire events. Biomass was calculated by subtracting the predicted mass of live foliage from the predicted total mass. Then, we corrected for wood decay by multiplying the Class 1 dead tree mass by 0.844 to account for decay (i.e., we determined the mean wood density

Table 1
Specific gravity (g cm^{-3}), mean diameter (cm) and the quadratic mean diameter (cm) of dead wood samples in low to high severity fires, Warner Creek Aug 2017 (29 years following fire).

Particle diameter range (cm)	Specific gravity (g cm^{-3})	N	Mean diameter (cm)	N	Quadratic mean diameter. (cm)
< 0.64 cm	0.58 ± 0.02	34	0.38 ± 0.01	99	0.40
0.64–2.54 cm	0.52 ± 0.02	25	1.40 ± 0.05	100	1.48
2.54–7.6 cm	0.41 ± 0.03	27	3.99 ± 0.13	83	4.15
> 7.6 cm sound	0.38 ± 0.02	52	–	–	–
> 7.6 cm rotten	0.30 ± 0.01	48	–	–	–

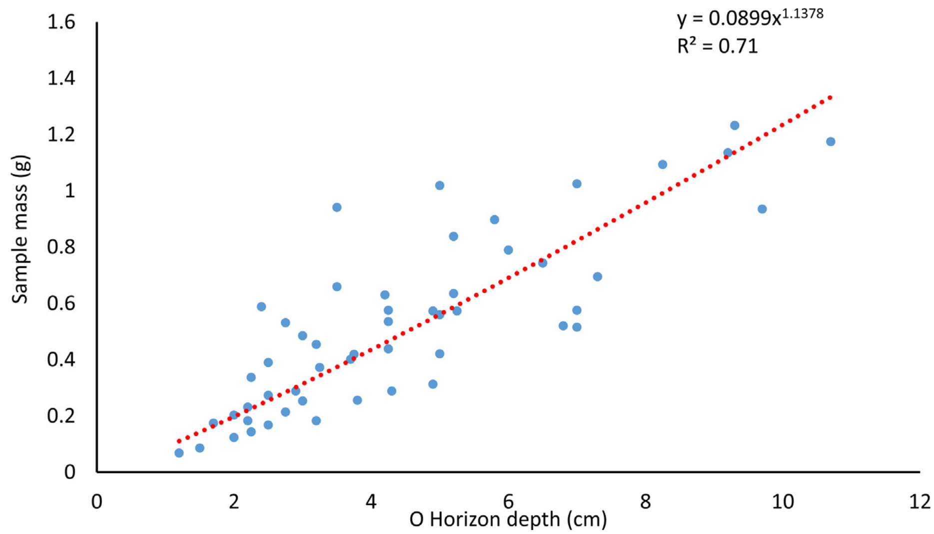

Fig. 3. The relationship of the depth of the organic horizon (cm) to its mass (g cm^{-2}).

of the standing dead trees was 0.38 g cm^{-3} compared to 0.45 g cm^{-3} for live trees). For Class 2 trees where only the main stem and large branches were present (but the mainstem had not yet been broken), we subtracted the predicted mass of both foliage and fine branches. We corrected for wood decay by multiplying the Class 2 dead tree mass by 0.844 to account for decay. Dead Class 3 trees (including stumps) were those where the trunk had been broken (fragmented) and only part of the mainstem remained. Biomass of this class was determined by multiplying the trunk volume (basal area times remaining trunk height) by dead wood specific gravity. The density and basal area of dead trees were also determined from these measurements. Total tree mass, density and basal area was calculated as the sum of both the live and dead trees.

2.3.3. Organic horizons (litter and duff mass)

Determination of organic horizon (litter and duff) mass required the development of an equation utilizing depth as the predictor variable. We collected 53 random samples of depth and bulk density in the low to high severity burn units, (Supplementary Table 2). The mean bulk density of the organic horizon was 0.522 ± 0.04 g cm^{-3}. We found a power equation provided the best fit of the data (Fig. 3):

$$y= 0.0899x^{1.1378}$$

where y = mass of the organic horizon (g cm^{-2})
x = depth of the organic horizon (cm)

$R^2 = 0.71$

Depth of the organic horizons was then measured each m for 10 m along each fuel transect (n = 40 per sampled forest) and from these data we estimated mass of the organic horizons. Understory herbaceous and epiphytic vegetation constituted only a very small proportion of the total forest mass (Smithwick et al., 2002) and was not measured.

2.3.4. Statistical Analysis

Within each fire, we tested whether fire severity and/or management scenario led to differences in forest biomass components (i.e., trees, downed wood etc.) using Analysis of Variance (ANOVA). When the P value of the ANOVA was < 0.05 we conducted a Fisher's Least Significant Difference test to determine which groups were significantly different. We also tested for differences in biomass data between the natural forest/fire scenario (the three fire severities combined) and managed landscapes (salvage logging and plantations combined) using ANOVA and these same multiple comparison procedures.

3. Results

3.1. Species composition

Twenty-nine years after fire in the Warner Creek sites, *P. menziesii* comprised > 98% of the surviving overstory tree mass in the moderate and high severity-fire stands. In low severity fire stands *P. menziesii* accounted for 85% of surviving overstory trees suggesting that the less fire resistant trees survived in greater numbers in the low severity fire. Similarly, 15 years following the Apple Fire, *P. menziesii* comprised 100% of the surviving trees in the high severity sites and 94% of the live overstory tree mass in the moderate and low severity burned forest areas.

There were significant differences in overstory structure due to varying fire severity. At Warner Creek, live trees in low severity stands comprised 94% of the total basal area compared to only 11% in the high severity stands (Table 2). Similarly, live overstory tree density was 8 ha^{-1} in high severity stands compared to 297 ha^{-1} in the low severity stands. Similar patterns of basal area and density were measured at the Apple Fire (Table 2).

There were 29 woody species that established following fire at the Apple Fire and 22 species that established post-fire at the Warner Creek Fire (Fig. 4). Among the post-fire established cohorts, *P menziesii* was the most widespread species and was present in 93% of the sampled stands. Other common species (and their occurrences) were *Acer circinatum* (68%), *Alnus rubra* (48%) and *Thuja plicata* (48%). At the Apple Fire there were 6 species that occurred in > 50% of the sampled stands. *P. menziesii* occurred in 86% of the stands. Other widely occurring species included *Ceanothus veluntinus* (68%), *Holodiscus discolor* (64%) and *Arbutus menziesii* (55%) (Fig. 5).

Land use (salvage logging and plantations) as well as fire severity strongly influenced post-fire succession and composition. The understory composition in the Warner Creek low severity fires were a diverse mix of naturally established species with *P. menziesii* and *A. circinatum* accounting for 39 and 30% of the post-fire established tree biomass, respectively. In contrast, *P menziesii* accounted for 82 and 92% of the understory tree mass in moderate and high severity fire stands, respectively. Plantations and salvage logged sites were dominated by a diverse mix of woody species with planted *P. menziesii* accounting for 64% of the post-fire mass in plantations and 45% in salvage logged sites at the Warner Creek Fire (Fig. 4).

At the Apple Fire, *P. menziesii* was also the most abundant understory species in high severity stands (66%), moderate severity stands (49%), plantations (41%) and salvage logged sites (34%) (Fig. 5). The

Table 2
Basal area and density of standing live and dead trees partitioned by large trees (prefire establishment and > 10 cm dbh) and small trees (post fire establishment and < 10 cm dbh). Numbers are mean with standard errors in parentheses. The P values are results of the Analysis of Variance testing for differences between the land use/fire cover types. Different superscripted letters note a significant difference between the land use/fire cover types.

Large Trees (pre-fire establishment)

	Dead basal area (SE)	Dead density (SE)	Live basal area (SE)	Live Density (SE
Apple Low	17 (10)^{a}	82 (14)^{a}	41 (9)^{b}	172 (81)^{b}
Apple Moderate	29 (9) ^{a}	144 (45)^{a}	26.6 (10.0)^{b}	48 (19)^{a}
Apple High	41 (10) ^{a}	220 (64)^{a}	2.9 (2.9)^{a}	6 (6)^{a}
Apple Salvage	43 (9)^{a}	236 (32)^{a}	0.0 (0.0)^{a}	0 (0)^{a}
Apple Plantation	29 (5)^{a}	167 (53)^{a}	1.4 (1.3)^{a}	13 (6)^{a}
P-value	0.28	0.37	0.0002	0.0032
Warner Low	9 (3)^{b}	37 (20)^{b}	76.2 (12.7)^{c}	207 (28)^{b}
Warner Moderate	39 (10)^{a}	102 (8)^{ab}	22.5 (5.5)^{ab}	132 (43)^{ab}
Warner High	63 (11)^{a}	137 (33)^{a}	3.6 (2.4)^{a}	8 (4)^{a}
Warner Salvage	43 (13)^{a}	111 (16)^{ab}	24.0 (6.3)^{ab}	233 (47)^{bc}
Warner Plantation	45 (10)^{a}	70 (17)^{ab}	26.3 (9.4)^{b}	374 (86)^{c}
P-value	0.0136	0.046	0.0000	0.0003

Small Trees (Post-fire establishment)

	Dead Basal Area (SE)	Dead Density (SE)	Live basal area (SE)	Live Density (SE)
Apple Low	0 (0)^{a}	0 (0)^{a}	0.5 (0.3)^{a}	1376 (190)^{a}
Apple Moderate	0 (0)^{a}	40 (21)^{a}	2.4 (1.2)^{a}	3094 (1157)^{ab}
Apple High	0 (0)^{a}	179 (143)^{a}	2.9 (1.4)^{a}	3382 (1246)^{ab}
Apple Salvage	0 (0)^{a}	87 (87)^{a}	2.2 (1.1) ^{a}	3407 (623)^{ab}
Apple Plantation	0 (0)^{a}	477 (477)^{a}	2.7 (2.0)^{a}	7062 (2757)^{b}
P-value	0.76	0.69	0.57	0.24
Warner Low	0 (0)^{a}	139 (104)^{a}	1.6 (0.5)^{a}	2099 (779)^{ab}
Warner Moderate	0 (0)^{a}	338 (212)^{a}	7.9 (2.2)^{b}	5560 (1751)^{a}
Warner High	0 (0)^{a}	456(340)^{a}	7.6 (2.8)^{b}	4949 (1930)^{ab}
Warner Salvage	1 (1)^{a}	671 (365)^{a}	1.3 (0.5)^{ab}	3817 (1114)^{ab}
Warner Plantation	0 (0)^{a}	126 (102)^{a}	1.3 (0.1)^{a}	1701 (690)^{b}
P-value	0.60	0.77	0.0237	0.26

conifers in the burned stands had naturally established whereas the plantations and salvage logged sites had been planted. *Pinus ponderosa* trees were also abundant in the salvage logged stands (29%) and plantations (11%). Broad-leaved species were more abundant in the post-fire composition of the Apple Fire compared to the Warner Creek Fire. *A. macrophyllum* was the most abundant species in the post-fire understory of the low severity fires (77%) at the Apple Fire (Fig. 4).

In addition to compositional differences, there were significant differences in the understory forest structure as manifested in basal area, density and mass. At the Warner Creek Fire, post-fire established tree basal area was almost 7-fold greater in the high (7.6 m^2 ha^{-1}) and moderate (7.9 m^2 ha^{-1}) severity fire sites compared to the plantations (1.3 m^2 ha^{-1}; Table 2). Similarly, density of post-fire established trees in both the high and moderate sites exceeded 4,949 ha^{-1} compared to 1,701 ha^{-1} in the plantations and 3,817 ha^{-1} in the salvage logged sites (Table 2). The mean mass of post-fire-established *P menziesii* (largely naturally established) was 23 Mg ha^{-1} in the high severity sites and was significantly greater than the plantation and salvage logged sites (largely planted; < 3.7 Mg ha^{-1}). Few *P. menziesii* established in the low severity fires where most of the late successional overstory remained intact.

3.2. Forest mass and fuel loads

There were large differences in the total aboveground biomass (TAGB) of the two fire sites. The TAGB of the low severity stands of the Apple Fire were 561 Mg ha^{-1} compared to 984 Mg ha^{-1} at the Warner Creek Fire. This is reflective not only of the time since fire (29 compared to 15 years) but also due to the soil and climatic differences at these two forested landscapes. The mass of each component of the ecosystem was typically higher at the Warner Creek forest stands compared to the Apple Fire (Tables 3 and 4).

Even though there were inherent differences in forest biomass, both locations responded similarly to fire and to land use (Tables 3 and 4). In spite of differences in overstory mortality, there was no significant difference ($P > 0.05$) in the TAGB between the low, moderate and high severity fires 15 years following burning at the Apple Fire. Similarly, there were no significant differences ($P \leq 0.05$) between the low and high severity burns 29 years following fire at the Warner Creek Fires. In contrast, TAGB in salvage logged and plantation sites were significantly lower than the burned forests. For example, the TAGB of the salvage-logged sites was 49% of that of the high severity sites at the Apple Fire and 42% of that of the high severity fires at the Warner Creek Fires. The mean TAGB of plantations was lowest of all fire and land use scenarios. At the Warner Creek Fires, TAGB of the plantations were < 30% of that of the high severity fire sites (e.g. 326 and 984 Mg ha^{-1}, respectively; Table 3).

While there was no significant difference in the TAGB of the low, moderate and high severity sites, there were dramatic differences in where the biomass was sequestered. As would be expected, the mass of live overstory trees declined with increasing fire severity. For example, the large live tree mass was 349 Mg ha^{-1} in low severity sites and 27 Mg ha^{-1} in high severity-burned stands at the Apple Fire (Table 4). Similarly, live tree mass in high severity burns of the Warner Creek Fires was only 5% of that of the low severity sites (705 and 35 Mg ha^{-1}, respectively; Table 3).

In contrast to live trees, the dead tree mass comprised 54% of the TAGB in the high severity stands, 34% in moderate severity stands and only 3% in the low severity stands at the Warner Creek Fires (Table 3). Dead tree mass of the salvage logged and plantation sites was only different from that of the high severity sites ($P \leq 0.05$). At the Apple Fire, dead large tree mass in the plantations and salvage-logged stands were not significantly different ($P = 0.8$) from that of the natural fire areas. While there were some live overstory trees even in high severity burns at the Apple Fire, there were no live overstory trees in the sampled salvage logged sites suggesting both live and dead tree removal during post-fire harvest at this site.

Organic horizon mass ranged from 31 to 60 Mg ha^{-1} at the Apple Fire and from 77 to 119 Mg ha^{-1} at the Warner Creek Fires; Tables 3 and 4). Salvage logged sites and plantations did not have a reduced mass of organic horizons compared to burned forest areas ($P > 0.53$). Organic horizons comprised a large proportion of the TAGB in plantations and salvage logged sites compared to the burned forest stands. For example, organic horizons comprised 8–11% of the TAGB in the natural forest stands and 22 and 29% of the TAGB in the salvage logged and plantations, respectively, at the Warner Creek Fires.

The largest quantities of downed wood were found in the high severity burns. At both fire sites total downed wood increased with increasing fire severity. For example at the Warner Creek Fire downed wood in the natural forests ranged was 124 Mg ha^{-1} in the low severity fire, 146 Mg ha^{-1} in the moderate severity fire and 269 Mg ha^{-1} in the high severity fire (Table 3). Sites that had been harvested (managed sites) had lower quantities of downed wood than the natural forests (Fig. 6). There was significantly ($P < 0.05$) more downed wood in the high severity burn stands than salvage logged or plantations.

Large downed wood (> 7.6 cm diam.) comprised a significant proportion of the TAGB especially in high severity fires. The large wood component in high severity fires comprised 27% of the TAGB at the Warner Creek Fires and 36% of the TAGB at the Apple Fire. In low severity fires, large wood comprised 12% at the Warner Creek Fires and 11% at the Apple Fire (Tables 3 and 4). The source of the downed wood

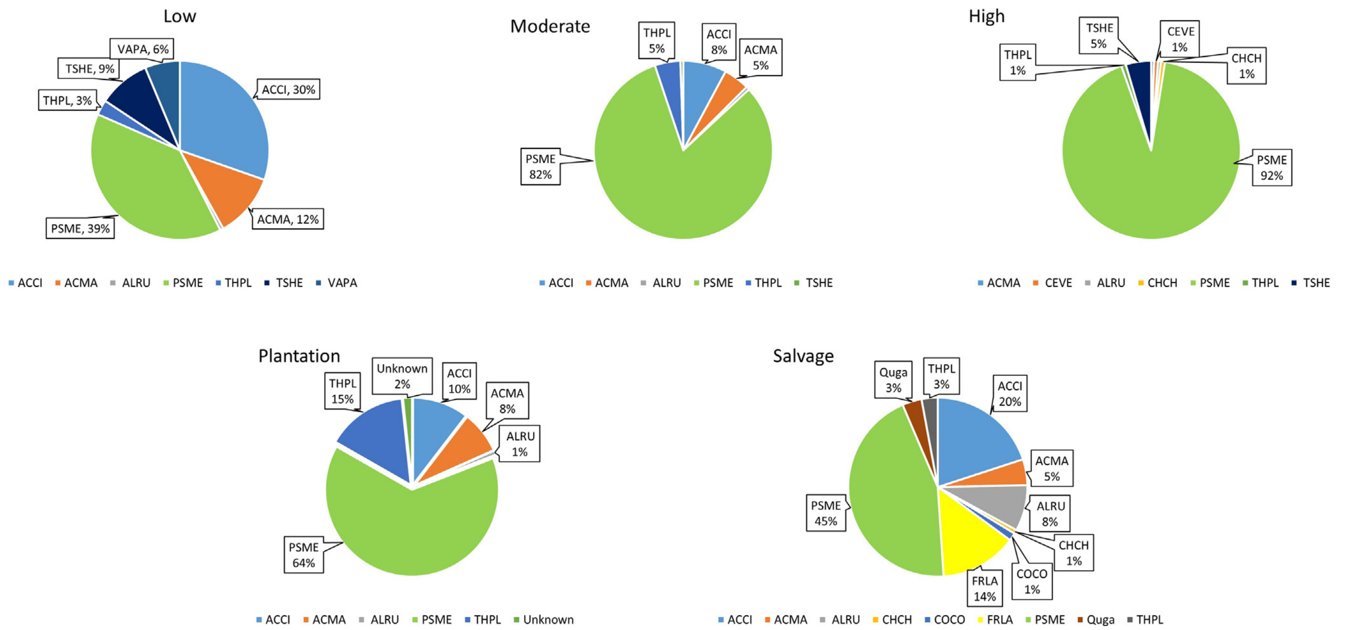

Fig. 4. The composition of woody vegetation that established following fire at the Warner Creek. Fires. Data are the percent of the total live biomass that is comprised by each species. Species abbreviations are: ACCI - *Acer circinatum*, ACMA - *Acer macrophyllum*, ALRU - *Alnus rubra*, CEVE - *Ceanothus velutinus*, CHCH - *Chrysolepis chrysophylla*, COCO – *Corylus cornuta*, FRLA – *Fraxinus latifolia*, PSME – *Pseudotsuga menziesii*, QUGA – *Quercus garryana*, THPL - *Thuja plicata*, TSHE - *Tsuga heterophylla*, and VAPA - Vacciniun parviflorum.

was largely from dead treefall and fragmentation (breakage). At the Apple Fire, 61% and 71% of the dead trees had fragmented at the time of sampling in the moderate and high severity burn sites. At the Warner Creek Fire, > 74% of the trees had fragmented. These fragmentation rates are slightly higher than those observed by Dunn and Bailey (2016) who had also sampled fragmentation 7 years before our study.

Sound large wood comprised the majority of the downed wood component in all sampled sites at the 15-year-old Apple Fire. In contrast, at the 29 year old Warner Creek Fires, the rotten large wood component comprised a larger proportion of the TAGB especially in the high severity stands as well as in the plantations and salvage logged sites (Tables 3 and 4).

Fine wood (wood < 2.54 cm diam) was only a minor component of the TAGB. The fine wood component comprised < 1.5% of the TAGB

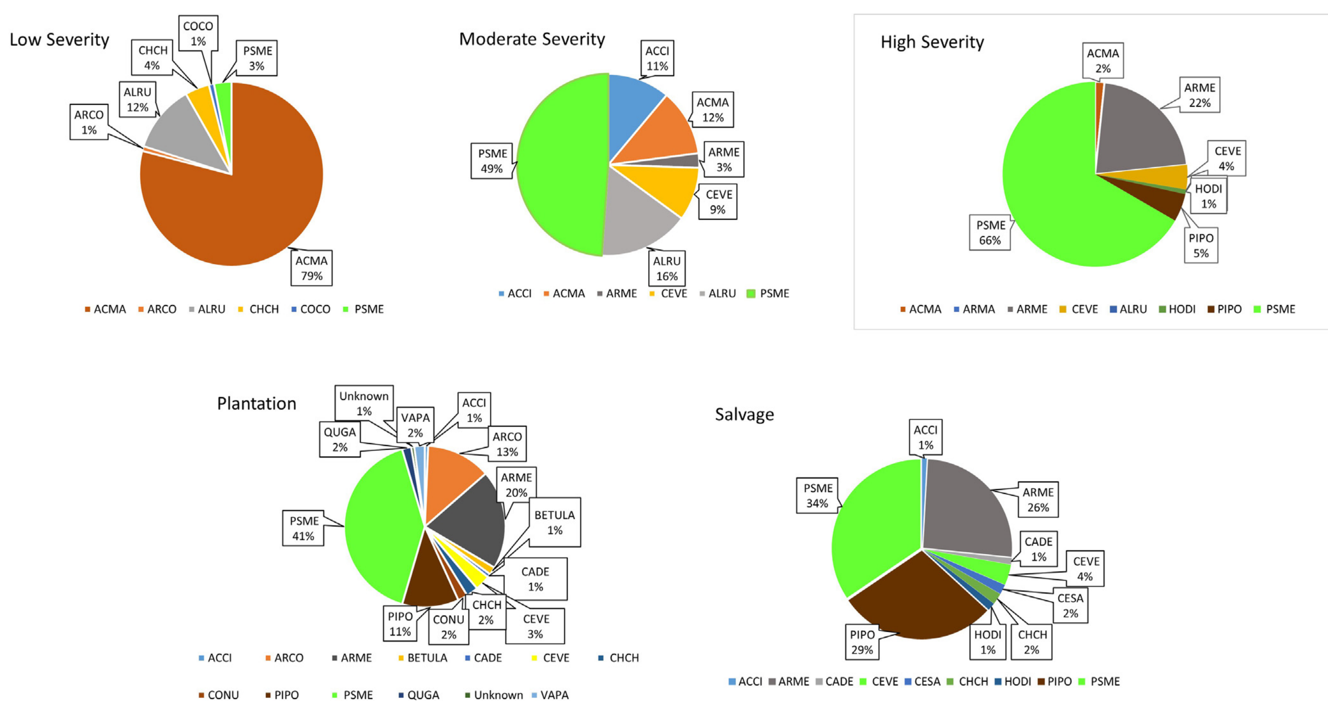

Fig. 5. The composition of woody vegetation that established following fire at the Apple Fire, 2017. Data are the percent of the total live biomass that is comprised by each species. Species abbreviations are: ACCI - *Acer circinatum*, ACMA - *Acer macrophyllum*, ALRU - *Alnus rubra*, ARCO - *Arctostphylos Columbiana*, ARME - *Arbutus menziesii*, BETULA - *Betula sp*, CADE - *Calocedrus decurrans*, CESA - *Ceanothus sanguineas*, CEVE - *Ceanothus velutinus*, CHCH - *Chrysolepis chrysophylla*, COCO – *Corylus cornuta*, CONU - *Cornus nutallii*, HODI - *Holodiscus discolor*, PIPO - *Pinus ponderosa*, PSME – *Pseudotsuga menziesii*, QUGA – *Quercus garryana*, and VAPA - *Vacciniun parviflorum*.

Table 3
Forest biomass 26–29 years following fire at the Warner Creek Fires, Willamette National Forest, Oregon. Numbers are means (Mg ha^{-1}) and one standard error (in parentheses). P values are results of the Analysis of Variance testing for differences between the land use/fire cover types. Different superscripted letters note a significant difference between the land use/fire cover types.

Site	Dead Large tree	Live lg tree	Tot large tree	Live small tree	Dead sm tree	Total small tree
Warner Creek Fires						
P value	0.0045	0.0000	0.0163	0.0349	0.4100	0.1427
Low	31.3 (18.0)^{b}	705.1 (141.8)^{a}	736.3 (132.9)^{a}	3.8 (1.3)^{c}	0.0 (0.0)^{a}	3.8 (8.3)^{c}
Moderate	226.3 (88.7)^{b}	202.2 (50.9)^{b}	428.4 (129.2)^{b}	21.4 (8.2)^{ab}	0.2 (0.1)^{a}	21.6 (8.3)^{ab}
High	505.5 (137.7)^{a}	35.1 (24.3)^{b}	540.5 (139.6)^{ab}	25.0 (7.4)^{a}	0.4 (0.3)^{a}	25.4 (1.2)^{a}
Salvage	98.0 (65.8)^{b}	165.1 (53.5) ^{b}	263.2 (50.1)^{bc}	8.3 (0.5)^{bc}	1.5 (1.3)^{a}	9.8 (1.5)^{bc}
Plantation	9.7 (2.3)^{b}	87.8 (7.5) ^{b}	97.5 (8.1) ^{c}	3.3 (1.2)^{c}	0.6 (0.6)^{a}	3.7 (1.1)^{c}

Site	Duff mass	> 7.6 cm Sound	> 7.6 cm Rotten	0–0.64 cm	0.65–2.54 cm	2.55–7.6 cm	Total wood mass	Total mass
Warner Creek Fires								
P value	0.7493	0.2941	0.0082	0.3670	0.1741	0.0116	0.0019	0.0045
Low	119.3 (29.3)^{a}	63.4 (20.5)^{ab}	50.9 (3.3)^{b}	1.5(0.4)^{a}	2.9(0.6) ^{a}	5.6 (1.4)^{ab}	124.3 (19.8)^{bc}	983.7 (140.5)^{a}
Moderate	77.4 (8.6)^{a}	72.8 (32.7)^{a}	63.6 (8.0)^{b}	1.1 (0.3) ^{a}	2.8 (0.5) ^{a}	5.8 (1.0)^{ab}	146.2 (35.9)^{b}	673.6 (144.4)^{bc}
High	93.3 (22.4)^{a}	91.0 (24.7)^{a}	165.3 (36.4)^{a}	1.2 (0.2) ^{a}	3.2 (0.5) ^{a}	8.2 (1.3)^{a}	269.0 (40.4)^{a}	928.2 (136.0)^{ab}
Salvage	93.3 (17.8)^{a}	8.7 (8.7)^{b}	44.2 (23.8)^{b}	0.8 (0.1) ^{a}	1.6 (0.1) ^{a}	1.3 (0.4)^{c}	56.7 (22.2)^{c}	422.9 (44.0)^{cd}
Plantation	93.3 (22.0)^{a}	52.5 (21.5)^{ab}	71.0 (21.1)^{b}	0.9 (0.2) ^{a}	1.8 (0.5) ^{a}	4.5 (1.2)^{b}	130.8 (15.1)^{bc}	325.7 (27.1)^{d}

Table 4
Forest biomass 15 years following fire at Apple Fire, Umpqua National Forest, Oregon. Numbers are means (Mg ha^{-1}) and one standard error (in parentheses). P values are results of the Analysis of Variance testing for differences between the land use/fire cover types. Different superscripted letters note a significant difference (P ≤, 0.10) between the different land cover types (fires and land uses.)

Site	Dead Lg tree	Live Lg tree	Tot lg tree	Live small tree	Dead sm tree	Total small tree
Apple Fire						
P value	0.759	0.0011	0.0180	0.60	0.4625	0.5977
Low	110.7 (94.2)^{a}	349.2 (91.3)^{a}	459.9 (23.6)^{a}	1.3 (1.2)^{a}	0.0 (0.0)^{a}	1.3 (1.2)^{a}
Moderate	126.8 (47.6)^{a}	256.3 (89.7)^{a}	383.1 (74.8)^{ab}	5.3 (3.1)^{a}	0.2 (0.2)^{a}	5.3 (3.1)^{a}
High	216.4 (106.4)^{a}	27.1 (27.1)^{b}	243.5 (97.5)^{bc}	8.2 (5.3)^{a}	0.0 (0.0)^{a}	8.2 (5.3)^{a}
Salvage	132.4 (73.5)^{a}	0.0 (0.0)^{b}	132.4 (73.5)^{c}	4.6 (1.8)^{a}	0.0 (0.0)^{a}	4.6 (1.8)^{a}
Plantation	85.1 (46.7)^{a}	12.2 (11.5)^{b}	97.4 (47.7)^{c}	4.2 (2.0)^{a}	0.1 (0.1)^{a}	4.2 (2.1)^{a}

Site	Duff mass	> 7.6 cm Sound	> 7.6 cm Rotten	0–0.64 cm	0.65–2.54 cm	2.55–7.6 cm	Total wood mass	Total Mass
Apple Fire								
P value	0.5386	0.1261	0.0537	0.3178	0.1005	0.2321	0.0401	0.0033
Low	31.0 (4.4)^{a}	42.2 (15.1)^{c}	18.2 (10.9)^{bc}	1.2 (0.1)^{a}	3.5 (0.6)^{ab}	4.0 (0.7)^{a}	69.2 (23.1)^{bc}	561.4 (50.6)^{a}
Moderate	48.0 (5.9)^{a}	96.7 (27.5)^{ab}	51.8 (11.4)^{a}	1.2 (0.1)^{a}	3.6 (0.5)^{a}	7.7 (0.2)^{a}	161.1 (33.0)^{ab}	597.9(39.2)^{a}
High	59.7 (14.7)^{a}	139.6 (34.1)^{a}	46.4 (13.6)^{ab}	1.2 (0.1)^{a}	2.9 (0.1)^{abc}	7.3 (1.2)^{a}	197.3 (32.0)^{a}	508.7 (94.4)^{a}
Salvage	42.0 (12.2)^{a}	74.1 (49.1)^{ab}	17.2 (11.0)^{c}	0.8 (0.1)^{a}	2.1 (0.6)^{c}	5.0 (1.4)^{a}	99.1 (59.8)^{bc}	278.2 (106.8)^{b}
Plantation	55.2 (12.8)^{a}	32.2 (10.6)^{c}	14.2 (4.3)^{c}	1.1 (0.3)^{a}	2.4 (0.4)^{bc}	5.2 (1.7)^{a}	55.0 (13.5)^{c}	211.8 (50.8)^{b}

in the natural forests at the Warner Creek Fire and about 2% at the Apple Fire. Fine wood comprised slightly greater proportions of the TAGB in the plantations and salvage sites, but there few differences across fire severities and land uses (Tables 3 and 4).

4. Discussion

Heterogeneous patterns of fire severity and land-use within forests characterized by mixed-severity fire regimes contributed to the substantial variation in post-fire vegetation composition and fuels succession observed in our study. Land-use (plantation and salvage-logging) exhibited the greatest influences on total aboveground biomass and carbon stocks, while fire severity exhibited the greatest effects on the partitioning of live vs. dead trees to the TAGB. These results imply that potentially complex trajectories of forest succession in the western Cascade Mountains are influenced and mediated by feedbacks between disturbance characteristics and human-systems that develop over long time-scales, as has been suggested in recent literature (Peters et al., 2017; Grimm et al., 2017).

4.1. Mass compared to other PNW forests

Beyond a faint char on the bark of the late successional trees and downed logs, there were few readily observable signs of fire in the low severity burns, especially 29 years following fire. In contrast, the residual dead trees and a dense regenerating forest characterized moderate and high severity fires. Assuming that the sampled stands within the low severity fires would be representative of late successional forests of the region, it is apparent that these forests were quite large in structure and biomass. The TAGB of these forests were 562 Mg ha^{-1} at the Apple Fire and 984 Mg ha^{-1} at the Warner Creek Fire (Tables 3 and 4). In Western Cascade old growth forests similar to the Warner Creek Fires, Smithwick et al. (2002) reported the mean TAGB was ≈1108 Mg ha^{-1}. Their estimate of total detrital biomass (all aboveground dead vegetative materials) was similar to the low severity sites of this study. Based upon forest inventories of the region, Law et al. (2018b) reported the mean TAGB to be about 458 Mg ha^{-1}. Their estimates of mean live tree biomass was 350 Mg ha^{-1}, which is lower than the mean of the live trees following low severity fires at the Apple Fire (349 Mg ha^{-1}) and at the Warner Creek Fires (749 Mg ha^{-1}). The lower TAGB estimate in Law et al. (2018) is likely because less productive forests outside of the *T. heterophylla* zone are included. Forests of the Apple and Warner Creek Fires are not out of the ordinary in terms of live biomass for the *Tsuga heterophylla* zone.

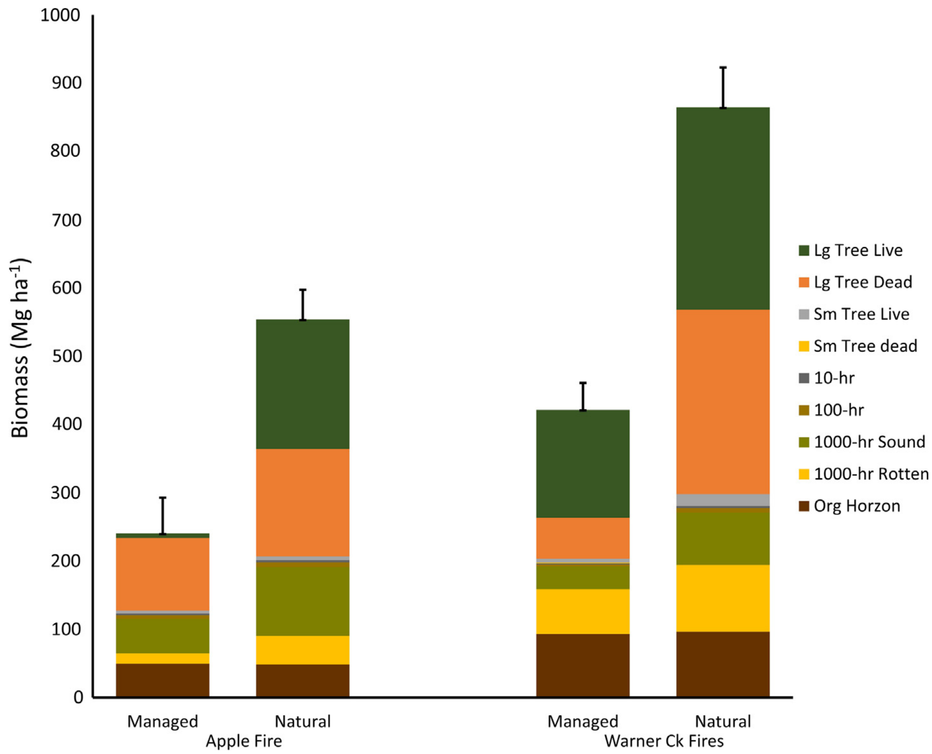

Fig. 6. Total aboveground biomass (Mg ha^{-1}) of managed (plantations and salvage-logged) and natural (burned late successional forests). Vertical small bars are one standard error of the total aboveground biomass.

4.2. Vegetation dynamics and fire

P menziesii was the most abundant plant species following fire in the moderate and high severity fire stands (Fig. 4). This conifer accounted for 82 and 92% of the post-fire established tree mass in moderate and high severity fire stands, respectively. This is in contrast somewhat with longstanding generalizations that planting is necessary following wildfire. Some have suggested that it could take more than 100 years to even establish conifer forests without conifer planting and subsequent shrub control (Sessions et al., 2004). Our results of abundant natural conifer establishment in the decades following fire are similar to results of other studies in *P. menziesii* forests reporting conifer re-establishment occurring primarily during the first few years following fire (Brown et al., 2013; Tepley et al., 2017).

4.3. Fuel dynamics and fire

After 15 and 29 years, salvage-logged sites and plantations had significantly lower quantities of downed wood compared to high severity fire sites, but not to low severity fire sites. The lower mass of wood in salvage-logged sites reported here differs from studies conducted soon after fire in forests of mixed severity regimes where downed wood in the salvage logged sites was higher than burned sites that were not logged (Donato et al., 2013; Campbell et al., 2016). Because of reductions in the mass of fire-killed trees, salvage logging has been suggested as a means of reducing future fire severity (forest mortality) (Peterson et al., 2015). However, the mass of organic horizons can greatly affect plant mortality by fire because of its continuity at the soil surface in close proximity to roots and tree crowns (Varner et al., 2007; Woolley et al., 2011). Salvage logging did not affect the mass of the organic horizons (P = 0.87 when testing between natural and managed forests). In addition, the presence of large downed wood such as that found in the high severity sites may actually decrease the rate of fire spread (Sullivan et al., 2018), though they may increase crowning, torching, and spotting during existing high-intensity wildfires (Brown et al., 2003). Further, the continuity of the live vegetation in the managed forest stands at the time of sampling would be sufficient for sustained combustion under late summer climatic and fuel moisture condition of this region. Zald and Dunne (2018) reported that intensive plantation forestry characterized by young forests and spatially homogenized fuels, rather than total pre-fire biomass, were the most significant drivers of wildfire severity.

4.4. Biomass and carbon loss associated with salvage logging and plantation management

There was no significant difference in the TAGB of salvage logged and plantations at both of the fire sites. As such, we combined data from managed sites within each fire to examine how land use affects TAGB and carbon stocks (Fig. 6). Active forest management resulted in significantly lower TAGB compared to the natural forests (P = 0.002 at the Apple Fire and P = 0.03 at the Warner Creek Fires). At the Apple Fire, the mean TAGB of the managed and natural sites was 241 Mg ha^{-1} and 555 Mg ha^{-1}, respectively. At the Warner Creek Fire, the mean TAGB of the managed and natural sites was 423 and 866 Mg ha^{-1}, respectively (Fig. 6). The most important differences in forest structure and biomass between natural and managed forests lie in significant decreases in total downed wood, live tree mass and dead large trees. These structural differences have important management implications with respect to wildlife habitat, and the functional role of these forests in carbon cycling (Law et al., 2018).

In the time since fire, there have likely been losses in carbon due to decomposition at all sites (Campbell et al., 2016; Acker et al., 2017). This is apparent by a greater proportion of the downed wood at the Apple fire being composed of sound wood and a greater proportion of the downed wood at the Warner fire being composed of rotten wood (Tables 3 and 4). Further, it can be expected that losses via decomposition will continue in the decades to come (Janisch and Harmon, 2002), but this will be offset to varying degrees by forest regeneration. For example forest regeneration in these productive forests is evidenced by a tree mass in the moderate and high severity sites of 60 and 223 Mg ha^{-1} in the 29-year old Warner Fire (Table 3).

Is there a net loss of carbon from the salvage logged sites and plantations? The harvest of biomass in managed forests could come at a cost of a loss in carbon storage and the values of these forests in climate change mitigation. We found that land use (salvage logging and plantations) resulted in significantly lower TAGB than the burned late successional natural forests (Fig. 6). The total aboveground carbon in the managed forests was 553 Mg CO_2e ha^{-1} lower compared to unmanaged forests in the Apple Fire and 781 Mg CO_2e ha^{-1} lower in unmanaged forests of the Warner Creek Fire. This likely reflects a realistic cumulative estimate of the greenhouse gas emissions due to land use at the 29 year old Warner Creek Fires. At 30 y following harvest Smith et al. (2006) assumed that all of the pulpwood (paper products) would be in landfills or emitted by this time and the wood used in modern building, including those of cross-laminated timber, outlive their usefulness and are replaced in about 30 y (Tollefson, 2017; Law et al., 2018). Ingerson (2011) reported that as little as 1% of initial standing-tree CO_2e may remain in products in-use and 13% in landfills at 100 years post-harvest.

Fires are a natural disturbance process in the western Oregon landscape, yet climate change, forest management, and fire suppression are likely changing the fire regime that has shaped these forests for millennia (e.g., Davis et al., 2017; Zald and Dunn, 2018). Managing the large carbon stocks present in the forests described in this study would facilitate balancing greenhouse gas emissions in the western USA (Law et al., 2018). Throughout the successional progression in the decades following disturbances, such as fire, these forests continue to function as significant carbon sinks and stocks (Gray et al., 2016).

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2019.117570.

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