Forest Ecology and Management 391 (2017) 164–175

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Landscape-scale quantification of fire-induced change in canopy cover following mountain pine beetle outbreak and timber harvest

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ARTICLE INFO

Article history:
Received 23 October 2016
Accepted 7 February 2017
Available online 23 February 2017

Keywords:
Wildfire
Burn severity
Fire effects
LiDAR
Eastern Cascades

ABSTRACT

Across the western United States, the three primary drivers of tree mortality and carbon balance are bark beetles, timber harvest, and wildfire. While these agents of forest change frequently overlap, uncertainty remains regarding their interactions and influence on specific subsequent fire effects such as change in canopy cover. Acquisition of pre- and post-fire Light Detection and Ranging (LiDAR) data on the 2012 Pole Creek Fire in central Oregon provided an opportunity to isolate and quantify fire effects coincident with specific agents of change. This study characterizes the influence of pre-fire mountain pine beetle (MPB; *Dendroctonus ponderosae*) and timber harvest disturbances on LiDAR-estimated change in canopy cover. Observed canopy loss from fire was greater (higher severity) in areas experiencing pre-fire MPB (Δ 18.8%CC) than fire-only (Δ 11.1%CC). Additionally, increasing MPB intensity was directly related to greater canopy loss. Canopy loss was lower for all areas of pre-fire timber harvest (Δ 3.9%CC) than for fire-only, but among harvested areas, the greatest change was observed in the oldest treatments and the most intensive treatments [i.e., stand clearcut (Δ 5.0%CC) and combination of shelterwood establishment cuts and shelterwood removal cuts (Δ 7.7%CC)]. These results highlight the importance of accounting for and understanding the impact of pre-fire agents of change such as MPB and timber harvest on subsequent fire effects in land management planning. This work also demonstrates the utility of multi-temporal LiDAR as a tool for quantifying these landscape-scale interactions.

1. Introduction

Climate change is facilitating an increase in large-scale ecological disturbance agents including wildfire (Abatzoglou and Williams, 2016; Littell et al., 2009) and bark beetle outbreaks such as those involving the mountain pine beetle (MPB; *Dendroctonus ponderosae*). MPB is a significant species of concern due to the threat that outbreaks pose to North American forests already stressed by warming temperatures, particularly sensitive species such as whitebark pine (*Pinus albicaulis;* Jenkins et al., 2014; Raffa et al., 2008). Along with timber harvest, fire and bark beetle outbreaks comprise three of the most significant drivers of tree mortality across the western United States (Cohen et al., 2016; Hicke et al., 2016). As such, there is considerable interest in understanding both interactions between these agents of forest change and also their effects on subsequent wildfire behavior and impacts (Turner, 2010). MPB epidemics preceding wildfire have been widely hypothesized to impact both wildfire severity and carbon emissions through alteration of pre-fire fuel loading (Hicke et al., 2012, 2013; Meigs et al., 2009). Mechanical fuel treatments are frequently used to reduce fire behavior metrics that influence severity, such as fire line intensity and flame length (Agee and Skinner, 2005; Reinhardt et al., 2008). Timber harvest activities not specifically targeted to hazardous fuel reduction (e.g., clear cuts) have not been well-assessed in the literature for contributions to subsequent fire intensity and ecological effects, however, the reduction in biomass alone suggests modification of fire behavior would be indirectly achieved (Hessburg et al., 2005, 2015). The intersection of MPB, timber harvest, and wildfire is inevitable across the landscape, so understanding the consequences and uncertainties of their combined impacts is

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http://dx.doi.org/10.1016/j.foreco.2017.02.015
0378-1127/

highly relevant to forest managers seeking to make informed decisions.

While there has been extensive effort made to understand how individual natural and anthropogenic agents influence subsequent fire effects (e.g., Hicke et al., 2012; Hudak et al., 2011; Turner, 2010), there have been limitations to quantifying fire effects in landscapes affected by MPB and/or timber harvest. As Hicke et al. (2012) demonstrate, to assess the impacts of forest agents of change on subsequent fire effects, data should be collected at multiple temporal points, including prior to any disturbance and between each subsequent change event. Researchers would also ideally stratify data collection across gradients of change, including control points without change for each disturbance type (Hicke et al., 2012). Realistically, this sort of controlled experiment is nearly impossible to design because of the impossibility to intentionally apply MPB and fire to landscapes. As such, most field studies of interactions have arisen from opportunistic sampling and analysis where fires occurred following known MPB outbreaks and timber harvest, where there is no multi-temporal data, or they are simulated (Hicke et al., 2012).

Passive remote sensing platforms such as Landsat provide both multi-temporal data and large or even complete populations (i.e., the number of samples is very high), allowing for the full range of interacting disturbance effects to be assessed. For analysis to be carried out, however, ecological metrics must be inferred from spectral reflectance, preferably with field data (Lentile et al., 2006). For inferring fire effects, the delta Normalized Burn Ratio (dNBR; Key and Benson, 2006; Lopez-Garcia and Caselles, 1991) and the Relative dNBR (RdNBR; Miller and Thode, 2007) are the most widely utilized spectral indices in the United States (US), primarily because dNBR and RdNBR raster products are produced and distributed by the Monitoring Trends in Burn Severity (MTBS) project (Eidenshink et al., 2007). Perhaps the most frequently used field sampling method used to relate field ecological observations to these spectral indices is the Composite Burn Index (CBI; Key and Benson, 2006) or variants such as the Geometrically Structured CBI (GeoCBI; De Santis and Chuvieco, 2009). However, CBI has not performed well in some ecosystems (e.g., Kasischke et al., 2008), and dNBR and RdNBR, which were originally developed through empirical correlation with CBI, are being analyzed across fires without being mechanistically tied to specific physiological metrics or function (e.g., Baker, 2015; Meigs et al., 2016), introducing considerable uncertainty into exactly what is being measured (Kolden et al., 2015).

Even when spectral indices are strongly correlated to quantitative field measurements, the vast majority of studies lack pre-fire observations and instead rely upon subjective reconstruction of the likely pre-fire conditions, thus failing to objectively capture the true magnitude of change (Roy et al., 2013; Smith et al., 2010, 2016). It is also difficult to address landscape fire effects using field measurements alone, as many wildfires are in remote areas with difficult terrain and few access roads. This often limits field data to parts of the fire that are safely accessible (Cansler and McKenzie, 2012; Hoy et al., 2008; Hudak et al., 2007; Key and Benson, 2006), making it very difficult to control for only the agents of change in an experimental framework and potentially causing a source of bias. Using spectral data to stratify potential field sites for validation can also be problematic. While studies may be able to capture the dynamic range of post-fire effects as indicated from the spectral data (e.g., Landsat) in the accessible sampled area (e.g., Hudak et al., 2007), the observed variation in the spectral indices may not be the best measure of the true variability in post-fire effects of interest (e.g., understory regeneration, soil stability). This may also lead to bias in the process of calibrating spectral remote sensing with field data (McCarley et al., 2017).

Although recent studies have explored the effects of antecedent MPB outbreaks on fire effects, there remain significant limitations in understanding their interaction with fire effects. Most have only analyzed a limited number of ground-based observations and have not demonstrated the scaling up of these observations and/or their relationship to remotely-sensed indices (Agne et al., 2016; Harvey et al., 2014a, 2014b; Schoennagel et al., 2012; Simard et al., 2011). Prior landscape-scale studies describe changes in reflectance rather than specific fire effects (Kulakowski and Veblen, 2007; Meigs et al., 2016), or been limited to calibrating reflectance using post-fire measurements only (i.e., CBI; Bond et al., 2009; Prichard and Kennedy, 2014).

There are many more studies examining the effect of fuel treatments on subsequent wildfire effects than on timber harvest activities not specifically intended to reduce hazardous fuels (e.g., Kennedy and Johnson, 2014; Moghaddas and Craggs, 2007; Ritchie et al., 2007; Safford et al., 2009, 2012; Stephens et al., 2012). Studies that do address timber harvest are primarily based on theoretical fire behavior (Graham et al., 1999; Keyes and O'Hara, 2002; Stephens, 1998; Stephens and Moghaddas, 2005) and/or a finite number of sampling plots rather than landscape-scale synoptic measurements (Cram et al., 2006; Lezberg et al., 2008; Omi and Kalabokidis, 1991; Weatherspoon and Skinner, 1995). These field observations also face a common challenge inherent to assessments of fire effects, namely the lack of pre-fire data. Even among fuel treatment studies, few have measurements of treated and untreated stands before and after a fire (Raymond and Peterson, 2005), and those that do rely on the assumption that their plots represent the range of heterogeneity in severity seen across the fire. Only Prichard and Kennedy (2014) have evaluated the effect of harvest treatments (alongside fuel treatments) at the landscape scale, albeit using spectral reflectance (i.e., RdNBR) validated with post-fire field data (CBI). A few others have performed similar analyses addressing the effect of fuel treatments (Finney et al., 2005; van Leeuwen, 2008; Wimberly et al., 2009).

To our knowledge, no prior studies exist that quantify changes in specific forest structure or ecophysiology metrics across a wildfire that follows either MPB outbreak or timber harvest. High-density return Light Detection and Ranging (LiDAR) data offers a practical method for performing this task. LiDAR is a proven forest measurement tool, able to detect structural changes in canopy cover, height, vertical distribution of canopy material, volume, biomass, and gap size (Hudak et al., 2009; Kane et al., 2013; Lefsky et al., 2002). Numerous studies have employed LiDAR for characterizing post-fire areas (e.g., Casas et al., 2016; Goetz et al., 2010; Kane et al., 2013, 2014, 2015; Wing et al., 2010), while only a few have utilized multi-temporal LiDAR datasets to asses change caused by fire (Bishop et al., 2014; McCarley et al., 2017; Reddy et al., 2015; Wang and Glenn, 2009; Wulder et al., 2009). Increasing availability of LiDAR has resulted in several cases where repeated LiDAR data acquisitions capture the canopy changes affected by wildfires, MPB damage, and timber harvest, although until now no study has capitalized on these incidents to address the intersection of disturbance agents. This study utilized multi-temporal LiDAR acquired following the 2012 Pole Creek Fire in central Oregon, where post-fire LiDAR data were opportunistically acquired in 2013 to resample an area flown in 2009. The fire followed many decades of silvicultural treatments by the Deschutes National Forest and an extensive MPB outbreak in the early 2000s.

The objective of this study was to determine the influence of antecedent MPB outbreak and timber harvest treatments on subsequent burn severity (defined here as the change in LiDAR-estimated percent canopy cover following McCarley et al., 2017). This study quantifies the change in canopy cover inferred from multi-temporal LiDAR for four stratifications: (1) by type of change agent both inside and outside of the fire perimeter, (2) by MPB outbreak intensity within the fire, (3) by forest harvest treatment type within the fire, and (4) by forest harvest treatment age classes within the fire.

2. Material and methods

2.1. Study area

The Pole Creek Fire (10,800 ha) occurred in September 2012 in the eastern Cascade Mountains of central Oregon (Fig. 1). One-third of the area within the fire perimeter falls within the Three Sisters Wilderness Area. Of the remaining two-thirds, timber harvest with scenic view restrictions was allowable on 2536 ha, and on an additional 3403 ha without scenic view restrictions (Deschutes National Forest, 1990). A MPB outbreak caused widespread mortality in lodgepole pine (*Pinus contorta*) between 2000 and 2006, particularly throughout the wilderness area (Agne et al., 2016; Fig. S1). The area of the Pole Creek Fire had not experienced a large (>404 ha) fire since 1984, as there were no previous fires in the MTBS database (Eidenshink et al., 2007), however, there may have been unrecorded smaller or older fires within the study area. The fire burned across a wide elevational gradient (1200–2100 m) and through a variety of forest types, with dominant species including lodgepole pine, ponderosa pine (*Pinus ponderosa*), mountain hemlock (*Tsuga mertensiana*), and grand fir (*Abies grandis*). However, in the study area lodgepole pine is often interspersed with other species rather than occurring primarily in homogenous stands as it does in other parts of central Oregon. Between 1981 and 2010, the mean July minimum and maximum temperatures were 6.3 °C and 23.1 °C, and the mean January minimum and maximum temperatures were −6.0 °C and 2.1 °C. Precipitation averages across the study area were highly dependent on topography, with the highest elevations receiving 2310 mm, primarily as snow, and the lowest elevations receiving only 412 mm (PRISM Climate Group, 2015).

2.2. LiDAR data

Pre- and post-fire LiDAR data were acquired by Watershed Sciences, Inc. (Corvallis, OR) using a Leica ALS50 at an altitude of 900 m above ground with a 28° field of view (±14° from nadir), at least 50% side-lap, a maximum of four returns per pulse, and an average pulse density of 8 pulses m^{-2}. All other collection settings were comparable between LiDAR datasets. The pre-fire LiDAR was flown between 7 and 11 October 2009. The post-fire LiDAR was acquired between 8 and 11 October 2013. Data were post-processed by the vendor for geometric accuracy. Height normalization was accomplished by subtracting the 1-m digital terrain model provided by the vendor in the delivery from the LiDAR point cloud using the USDA Forest Service's FUSION software package (McGaughey, 2014). Pre-fire LiDAR data were re-projected using LAStools (Isenburg, 2013) from state plane into UTM to match the post-fire LiDAR data.

Next, each LiDAR point cloud was binned to 30-m voxels and percent canopy cover was calculated per voxel for pre- and post-fire scenes (Fig. 2a and b). Canopy cover was selected to represent fire effects based on it being the structure metric most strongly related to the spectral indices used to measure fire effects in other studies of multiple interacting forest disturbance agents (McCarley et al., 2017); this was critical as it facilitated the comparison of results to prior studies. LiDAR-estimated canopy cover is defined as the number of LiDAR returns above a certain height-above-ground as a percentage of all LiDAR returns. Returns can be comprised of live or dead material. Based on the surface fuel definition from (Brown et al., 1982), 1.8 m was used as the threshold for calculating canopy cover. This threshold also mitigated the presence of less than 10 cm of snow on the ground during the 2009 acquisition as recorded by the Three Creek Meadows SNOTEL site; no snow was present during the 2013 acquisition. Once percent canopy cover was quantified, the data were converted to a raster grid and a differenced image was calculated in ArcGIS (Fig. 2c; ESRI, 2014). Since pre- and post-fire canopy cover are reported as a percent, the unit for change in canopy cover is a difference in percent.

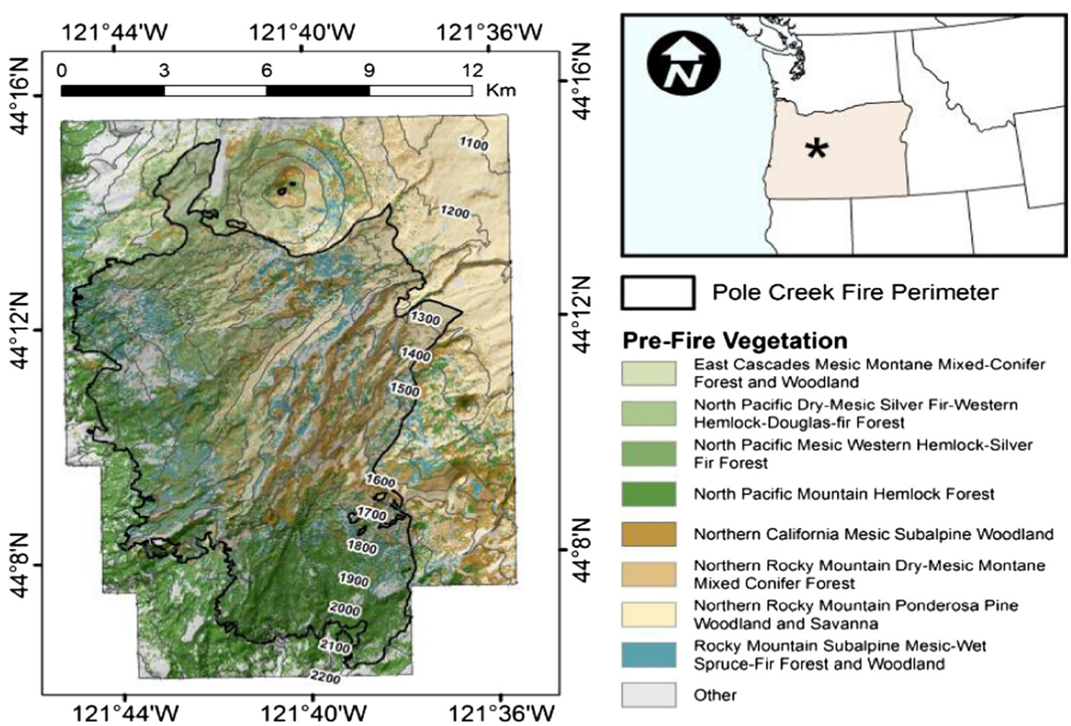

Fig. 1. Pole Creek Fire location and elevation contours (derived from post-fire LiDAR estimated ground elevation) overlaid on pre-fire existing vegetation type (EVT) data (LANDFIRE, 2010) clipped to the multi-temporal LiDAR acquisition area. EVTs less than 1000 ha in extent are included in "Other".

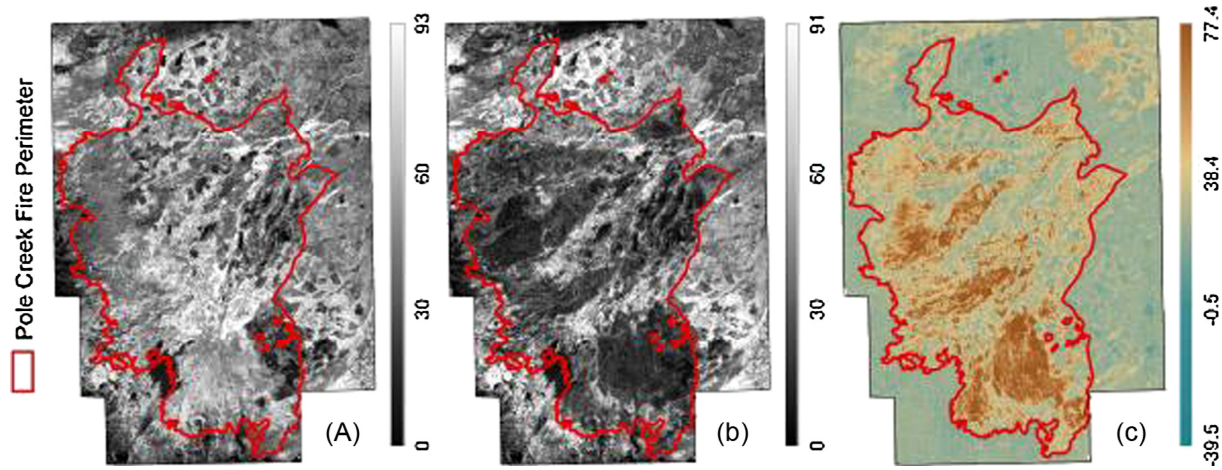

Fig. 2. LiDAR-estimated percent canopy cover for (A) pre-fire (B) post-fire and (C) the difference between pre- and post-fire (change in percent canopy cover).

2.3. MPB data

Aerial detection survey (ADS) data, which includes manual mapping of damage caused by insects, disease, weather, and mammals, were acquired from the USDA Forest Service Pacific Northwest Region (USDA Forest Service, 2015a). Mapped ADS MPB mortality estimates were converted to trees ha^{-1} (TPH; Fig. 3a). The ADS data showed that tree mortality from the MPB outbreak occurred primarily between 2000 and 2006, peaking in 2004 (Fig. S1); therefore, the sampled data were limited to those years. For MPB-impacted lodgepole pine, needle loss typically happens within 3–5 years following beetle attack (Hicke et al., 2012), which is 2–4 years after when ADS records mortality. Given that the pre-fire LiDAR was collected in 2009, it is unlikely that there were very many trees which still retained needles at the time of the LiDAR acquisition, and even fewer for which the needles subsequently dropped prior to the 2012 fire. This would be represented erroneously as fire-caused change in canopy cover, and while this is a source of uncertainty in the analysis, most MPB mortality occurred more than four years before the LiDAR flight (Fig. S1). As such, the contributions of this scenario to overall error are minimal.

ADS polygons were converted into 30-m gridded data such that the output raster cells aligned with those of the 30-m LiDAR-derived raster. However, polygons are sketched at a map-scale between 1:250,000 and 1:50,000 depending on the flight (McConnel et al., 2000), meaning the distance between two 30-m pixels could be as little as 0.12 cm on paper. The unit of observation, and thus effective minimum mapping unit according to some sources, is 0.4 ha (Johnson and Ross, 2008). Due to the noise in the original ADS data, a 3×3 moving window was applied to smooth the mortality estimates. Finally, TPH estimates were summed over the 7 year window from 2000 to 2006 to produce cumulative MPB mortality (Fig. 3a).

Multiple sources have suggested that ADS data give only approximate estimates of mortality (McConnel et al., 2000; Meddens et al., 2012; Meigs et al., 2011). This study also recognized that at relatively low TPH values, MPB–induced tree mortality is indistinguishable from background levels of other causes of mortality (i.e., disease, wind throw, drought). McCarley (2016) identified a threshold of 16 TPH, a comparable but slightly more conservative threshold for MPB mortality than that used by Agne et al. (2016) for the same area, to isolate epidemic MPB outbreak. For the remainder of the analysis, MPB-affected areas were defined as those with impacts above background mortality noise (>16 TPH).

2.4. Harvest data

Forest harvest activity history was provided by the Deschutes National Forest (USDA Forest Service, 2015b), which has spatial records of harvest and fuel reduction activities dating back to the 1960 s (Fig. 3b and c). Areas where treatments occurred between LiDAR acquisition dates were excluded from the analysis (1187 ha total; 137 ha inside the fire perimeter). Ten unique combinations of forest treatments were identified inside the fire perimeter (Table 1). Seven of these were assessed for effect of harvest treatment (HARV) type on burn severity (defined here as the change in LiDAR-estimated percent canopy cover), while the remaining combinations were deemed too small to be representative. Total area and patch size characteristics for the selected harvest types are provided in Table S1. The age of the treatments inside the fire ranged from 41 years since treatment (a 1971 overstory removal cutting) to six (2006 pile burning). Treatment age outside the fire ranged from 40 years (a 1962 stand clearcut) through four years (2008 two-aged coppice cut and sanitation cut). In order to assess the impact of harvest treatment age on burn severity, the treatments were classified by decades prior to fire (Table 1).

2.5. Analysis

The effect of MPB and timber harvest on LiDAR-estimated change in canopy cover was evaluated by first comparing the distributions for possible combinations of disturbances inside and outside of the fire perimeter (Fig. 3d). Outside of the fire perimeter, disturbance agents included no agents (Control), MPB, harvest (HARV), and HARV plus MPB (HARV/MPB). Inside the fire perimeter, agents were Fire-only, MPB followed by fire (MPB/Fire), harvest followed by fire (HARV/Fire), and HARV plus MPB followed by fire (HARV/MPB/Fire). Due to the large number of observations, tests of significance (i.e., ANOVA) were redundant. These statistics are useful for inferring differences in populations based on a sample, but as the sample size grows, it gets closer to the population size and there is greater confidence that any non-identical groups are different. This leads to the tests being excessively sensitive to differences between the groups (i.e., all differences are significant). In this case, data for the entire fire was collected, thus providing the entire population for which differences could be observed directly. Therefore, this study simply quantifies and characterizes the differences in the distributions with the understanding that these represent the entire range of values occurring for this fire.

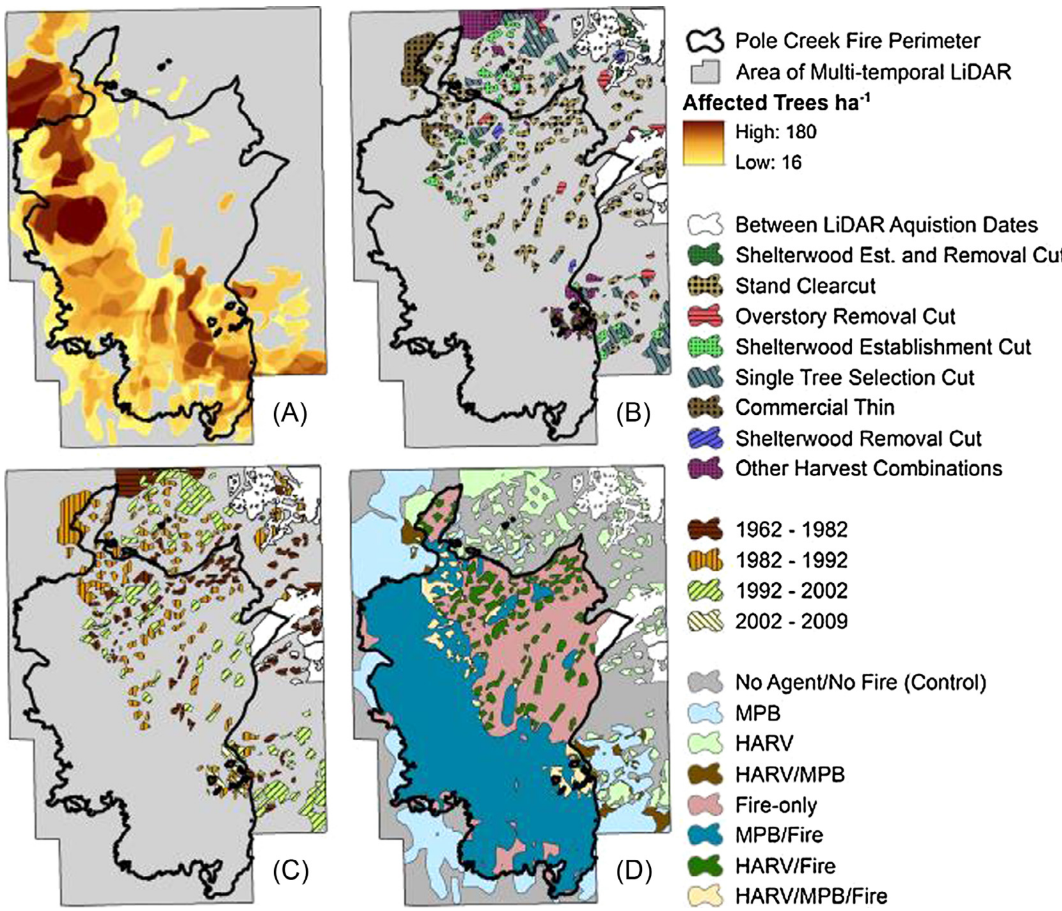

Fig. 3. (A) Cumulative area affected by mountain pine beetle reported by aerial detection survey data during the peak outbreak years of 2000 and 2006. (B) Harvest treatment types and (C) treatment age reported by the Deschutes National Forest. (D) Combinations of forest agents of change represented in the study: Mountain pine beetle (MPB) followed by wildfire (MPB/Fire), harvest followed by wildfire (HARV/Fire), a combination of forest harvest and MPB followed by fire (HARV/MPB/Fire), wildfire having not been preceeded by either MPB or harvest (Fire-only); and areas outside of the fire affected by MPB, HARV, HARV/MPB, and no pre-fire agent (Control). Areas where harvest occurred between LiDAR acquisition dates were excluded from the analysis.

The effect of MPB was further explored by repeating this analysis across levels of pre-fire beetle mortality. Due to the uncertainty associated with ADS mortality estimates, MPB impacts were binned into 24 TPH increments using the range from greater than 16 TPH to the maximum observed cumulative TPH (183). The results visualize the distribution of change in canopy cover values for the binned TPH groups, in order to compare these to HARV/Fire and Fire-only areas. The effect of forest treatment type was assessed by examining the distributions of the seven major HARV types within the fire perimeter in comparison with MPB/Fire and Fire-only areas. Forest treatment age was evaluated by plotting the distribution of the four harvest-age classes, defined by decades prior to the fire.

3. Results

Median values for change in LiDAR-estimated percent canopy cover from pre- to post-fire ranged from a reduction of 19% for the MPB/Fire class to an increase of 1.4% for the HARV/MPB class (Fig. 4 & Table S2). Areas experiencing fire exhibited greater canopy cover loss than areas not experiencing fire. Considering only the areas within the fire perimeter, pre-fire MPB mortality (MPB/Fire) produced the greatest loss of canopy cover (highest severity), followed by Fire-only, HARV/Fire, and HARV/MPB/Fire. In addition to observing higher severity in MPB/Fire than any other combination of forest conditions, severity within MPB/Fire increased with MPB outbreak intensity (Fig. 5 & Table S3).

Results indicated lower severity in HARV than in MPB or Fire-only, with some variation between types (Fig. 6 & Table S4). The lowest average severity occurred in shelterwood removal cuts, followed by commercial thinning, single tree selection, shelterwood establishment cuts, and overstory removal cuts. The highest severities were in stand clearcuts and the combination of shelterwood establishment and removal cuts. Time since treatment affected severity only for the earliest treatments (Fig. 7 & Table S5), with relatively higher severity for treatments between 1971 and 1982, but little difference between treatments in more recent decades.

4. Discussion

4.1. Effect of change agent type

Compared to Fire-only areas, greater severity in canopy cover change was observed in areas of antecedent MPB tree mortality than in areas unaffected by or with background levels of mortality (TPH $\leq$ 16). There has been widespread disagreement in the literature regarding the impact of MPB on subsequent fire effects (Hicke et al., 2012), although recent studies have indicated that MPB may lessen severity or have negligible impacts (Agne et al., 2016; Meigs et al., 2016). The theory of reduced severity in gray phase is based

Table 1
Area affected by all possible forest agent combinations: Fire, timber harvest (HARV), mountain pine beetle (MPB), and for no agents (Control); and by Harvest treatment type, harvest treatment age, and MPB intensity.

	Area (ha)	% of Fire Area
Outside Fire Perimeter	10,033	–
Control	5225	–
MPB	3067	–
HARV	1460	–
HARV/MPB	281	–
Inside Fire Perimeter	10,614	100.0
Fire-only	3332	31.4
MPB/Fire	6055	57.0
16 – 40	*1436*	*13.5*
40 – 64	*1283*	*12.1*
64 – 88	*1249*	*11.8*
88 – 112	*873*	*8.2*
112 – 136	*779*	*7.3*
136 – 160	*316*	*3.0*
160 – 183	*119*	*1.1*
HARV/Fire – Type	816	7.7
Shelterwood establishment and removal cut	*40*	*0.4*
Stand clearcut	*570*	*5.4*
Overstory removal cut	*29*	*0.3*
Shelterwood establishment cut	*19*	*0.2*
Single tree selection cut	*70*	*0.6*
Commercial thinning	*47*	*0.4*
Shelterwood removal cut	*28*	*0.3*
Pre-commercial thin, commercial thin, and pile burning	*6*	*< 0.1*
Two-aged seed-tree and removal cut	*4*	*< 0.1*
Two-aged coppice cut	*3*	*< 0.1*
HARV/Fire – Age	816	7.7
1971 – 1982	*249*	*2.3*
1982 – 1992	*307*	*2.9*
1992 – 2002	*251*	*2.4*
2002 – 2009	*9*	*< 0.1*
HARV/MPB/Fire	411	3.9

on reductions in crown bulk density and canopy base height leading to reduced crown fire behavior, but this study demonstrates contradictory findings.

HARV/Fire areas exhibited lower severity in canopy cover change than Fire-only areas. This finding is consistent with other studies demonstrating lessened fire effects in the forest canopy following a harvest activity and is the predominant conclusion throughout the literature (Cram et al., 2006; Fraver et al., 2011; Graham et al., 1999; Omi and Kalabokidis, 1991; Weatherspoon and Skinner, 1995). Many other studies have demonstrated an increase in fire effects at the surface due to an increase in fuels as a result of mechanical activities (Cram et al., 2006; Donato et al., 2006; Lezberg et al., 2008; Raymond and Peterson, 2005; Stephens, 1998; Thompson et al., 2007). This study examined burn severity in the canopy only and was not able to account for post-harvest treatment of slash. Additional research is needed to understand the complexity of fire effects across forest strata.

LiDAR-estimated change in canopy cover appeared to positively detect fire effects. This was demonstrated by all agent combinations outside the fire perimeter exhibiting substantially lower change in canopy cover than any combinations inside the fire perimeter. However, there was some variation. Areas experiencing only MPB (i.e., no fire) had greater reductions in canopy cover than other agent combinations outside the fire perimeter, which might be explained by post-MPB snag fall, woodcutting, or other mortality agents. These factors may also have contributed slightly to the higher severity observed in MPB/Fire. Outside the fire perimeter, HARV tended to correspond with a slight increase in canopy cover, potentially representing forest growth in these areas.

4.2. Effect of MPB outbreak intensity

A positive relationship was noted between MPB intensity and subsequent burn severity (defined here as the change in LiDAR-estimated percent canopy cover). This trend was observed even when normalizing for pre-fire conditions by using relative change in canopy cover (Fig. S2). These results agree with several prior studies (Table 2); however, the location, time since outbreak, and definition of severity impacts varied. The most notable difference was time since outbreak, as Prichard and Kennedy (2014) and Harvey et al. (2014a) found higher severities only in recently affected stands (0–3 years since attack). This time since outbreak diverges considerably in structure and flammability from the outbreak period observed by this study (7–13 years since attack) due to needle retention (Hicke et al., 2012; Jolly et al., 2012). This study evaluated the entire MPB-affected area uniformly without regard to variation in time since outbreak since all areas were in the same post-outbreak stage.

The results of this study diverge from those finding decreased or no effect from antecedent MPB on burn severity (Table 2). Agne et al. (2016) established and evaluated burn severity at 52 plots within the Pole Creek Fire. They found that their measures of physical burn severity (bole char height, proportion of canopy con-

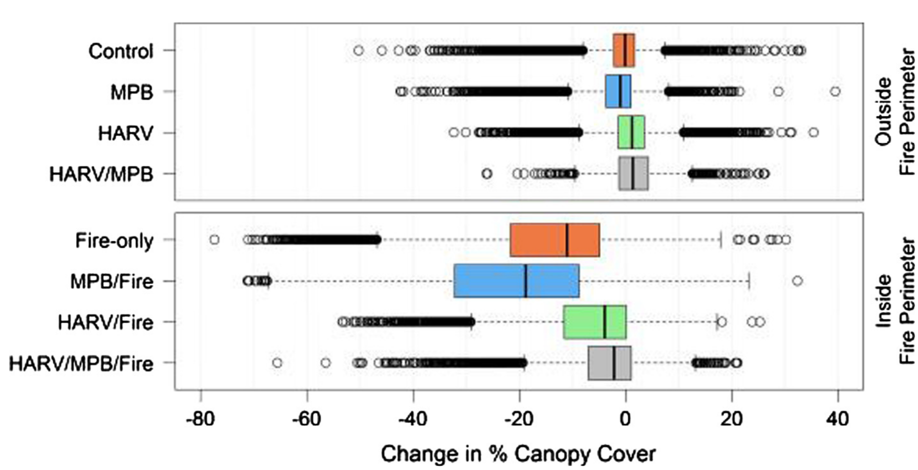

Fig. 4. Distribution of change in percent canopy cover for possible combinations of agents: Control (no agent), mountain pine beetle (MPB), timber harvest (HARV), both HARV and MPB (HARV/MPB), Fire-only, MPB followed by wildfire (MPB/Fire), HARV/Fire, and HARV/MPB/Fire. Solid lines represent medians, boxes show the interquartile range, whiskers extend to 1.5 times the interquartile range, and points are outliers.

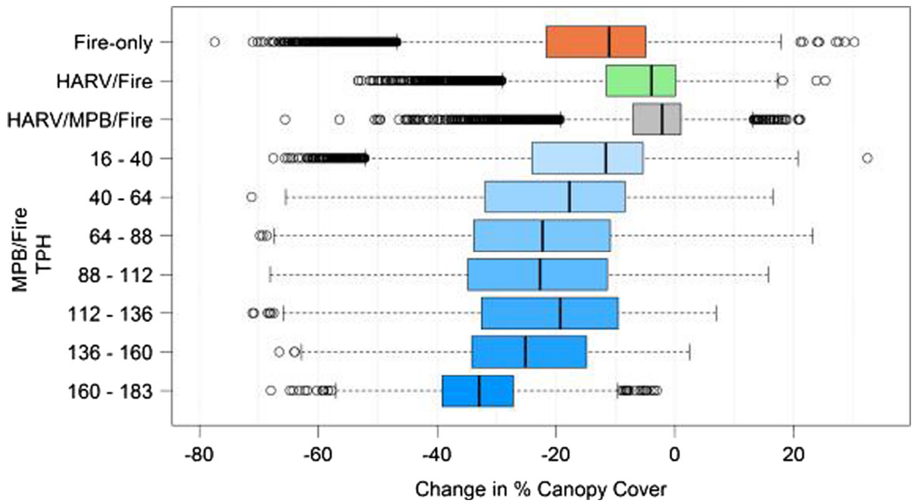

Fig. 5. Distributions of change in percent canopy cover for Fire-only, timber harvest followed by wildfire (HARV/ Fire), HARV and mountain pine beetle followed by wildfire (HARV/MPB/Fire), and MPB/Fire binned at 24 trees ha^{-1} (TPH) by cumulative MPB-induced mortality. Solid lines represent the medians, boxes show the interquartile range, whiskers extend to 1.5 times the interquartile range, and points are outliers.

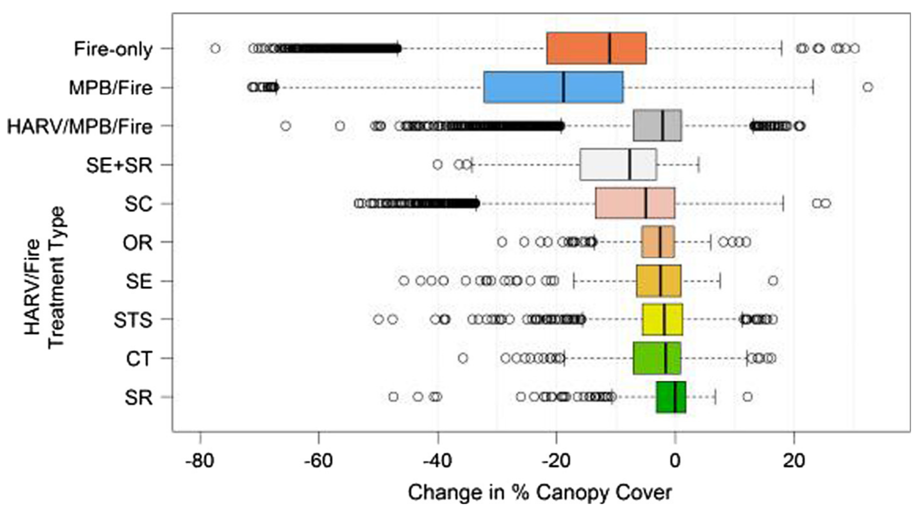

Fig. 6. Distributions of change in percent canopy cover for Fire-only, mountain pine beetle followed by wildfire (MPB/Fire), timber harvest and MPB followed by wildfire (HARV/MPB/Fire), and HARV/Fire treatment types. combination of Shelterwood Establishment and Removal Cut (SE + SR), Stand Clearcut (SC), Overstory Removal Cut (OR), Shelterwood Establishment Cut (SE), Single Tree Selection (STS), Commercial Thin (CT), and Shelterwood Removal Cut (SR). Solid lines represent the medians, boxes show the interquartile range, whiskers extend to 1.5 times the interquartile range, and points are outliers.

sumed, basal area killed, soil char depth, and proportion of ground char) and CBI estimates decreased with increasing MPB intensity. Meigs et al. (2016) evaluated fires across Oregon and Washington, narrowly excluding the Pole Creek Fire due to the time period of analysis. They also observed decreased fire effects (defined as RdNBR) in areas affected by MPB. Certainly, some of the disagreement in both these cases is due to different measures of severity, especially when comparing canopy vegetation to soil effects. However, measures of severity such as proportion of canopy consumed are similar to this study's severity, shedding little light as to why the results of these studies differ. One possibility is in the extrapolation of limited ground data in other studies.

The use of multi-temporal LiDAR here improves on prior studies in several important ways. While Agne et al. (2016) used some quantitative, well-defined measures of burn severity, they lacked pre-fire data. This leads to uncertainty about the accuracy of attributing changes observed post-fire to either fire or MPB (Roy et al., 2013; Smith et al., 2010, 2016). Also, because they established a relatively small number of plots that were all in the wilderness and at high elevation relative to the rest of the fire, Agne et al. (2016) may not have been able to capture the same range of heterogeneity in effects observed by LiDAR over the entire fire. Using multi-temporal reflectance data (RdNBR), Meigs et al. (2016) were better able to capture the change at a landscape-scale; although, there are considerable issues arising from the use of spectral indices alone as a measure of severity, and particularly RdNBR (McCarley et al., 2017). Other studies have already indicated many of these shortcomings (Kolden et al., 2015; Lentile et al., 2009; Roy et al., 2006; Smith et al., 2016).

Certainly, there are limitations to this dataset as well. Without using LiDAR intensity values, live and dead returns are indistinguishable unless tied to crown shape, the consideration of which was beyond the scope of this study. The use of intensity has demonstrated effectiveness in other post-fire environments (Kim et al., 2009; Wing et al., 2015) and for other post-MPB outbreak forests (Bright et al., 2014, 2013), and might be used to focus on how change in living canopy varies across pre-fire agents of change. However, the 2009 LiDAR data did not have the necessary

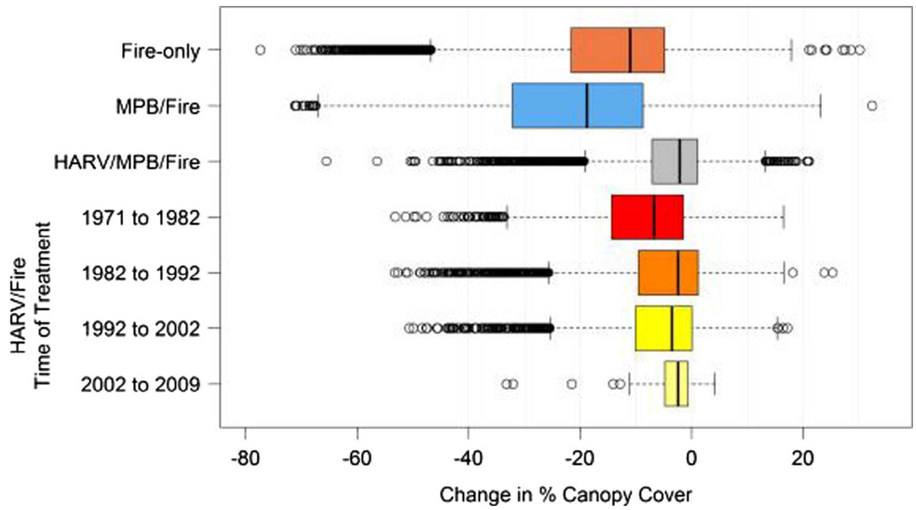

Fig. 7. Distributions of change in percent canopy cover for Fire-only, mountain pine beetle followed by wildfire (MPB/Fire), timber harvest and MPB followed by wildfire (HARV/MPB/Fire), and HARV/Fire binned by treatment age in decades before the fire. Solid lines represent the medians, boxes show the interquartile range, whiskers extend to 1.5 times the interquartile range, and points are outliers.

Table 2
Recent research examining the effect of mountain pine beetle induced tree mortality on burn severity.

Reference	Location	Time since outbreak	Definition of burn severity	Burn severity response
Turner et al. (1999)	Greater Yellowstone Ecosystem	5–17 years	Observational assessment of soil and vegetation	Decreased then increased at higher beetle intensity
Kulakowski and Veblen (2007)	Colorado	0–5 years	dNBR	No effect
Bond et al. (2009)	Southern California	0–1 years	RdNBR	No effect
Kulakowski and Jarvis (2011)	Colorado and Wyoming	Variable	Undefined	No effect
Harvey et al. (2014a)	Greater Yellowstone Ecosystem	0–3 years; 3–15 years	Char height/bole scorch, tree-mortality, depth and char of post-fire litter	Variable, increased (0–3 years), no effect (3–15 years)
Harvey et al. (2014b)	Northern Rocky Mountains	0–2 years; 3–10 years	Char height/bole scorch, tree-mortality, depth and char of post-fire litter	Variable, mostly no effect
Prichard and Kennedy (2014)	Washington	0–3 years	RdNBR	Increased
Agne et al. (2016)	Oregon	8–15 years*	CBI, char height, basal area killed, crown consumed, depth and char of post-fire litter	Decreased
Meigs et al. (2016)	Washington and Oregon	Variable	RdNBR	Decreased
McCarley et al. (*This Study*)	Oregon	7–13 years*	LiDAR-estimated change in canopy cover	Increased

* Differences in reported time since outbreak for the same fire (Pole Creek) were due to Agne et al. (2016) using the year of epidemic initiation at their study plots (Michelle Agne, Oregon State University, Corvallis, Oregon, USA, personal communication), while McCarley et al. (*This Study*) reported time since outbreak using the peak period of mortality across the LiDAR acquisition area (Fig. S1).

information to process and normalize intensity values (Bright et al., 2013). This dataset was also not reconciled with field data of any kind, primarily because no pre-fire data existed. Many of the conflicting studies quantifying the effect of MPB on burn severity continue to use different modes of analysis (i.e., field data, passive remote sensing, active remote sensing), thus it would be beneficial if future work reconciled multiple methods.

There are also uncertainties regarding the MPB dataset. The ADS measure of mortality is based off of the number of observed trees divided by the area of the polygon mortality was observed in (McConnel et al., 2000), and is thus insensitive to stocking rate. It is possible that higher TPH values are indicative of denser stands, which are more likely to burn at higher severity (Agee and Skinner, 2005). However, while stocking rates might affect the increased severity observed at higher TPH levels, this does not fully explain the higher average severities in MPB/Fire even at low TPH levels compared to HARV/Fire or Fire-only. This study also noted that change in canopy cover relative to pre-fire cover did not alter any of the observed trends (Fig. S2). Nonetheless, these uncertainties highlight the importance of using accurate MPB mortality estimates in future work. Other studies have produced alternatives (Meddens et al., 2013; Prichard and Kennedy, 2014) or developed methodologies that apply corrections to ADS data. Meddens et al. (2012) developed a method of adjusting ADS data using species-specific crown area, albeit at a much coarser resolution (1 km). LandTrendr (Kennedy et al., 2010) is an another useful approach, which has demonstrated effectiveness for detecting insect damage at a 30-m resolution (Meigs et al., 2011). However, the use of these approaches was beyond the scope of the study, in part because of the limited amount of cloud-free Landsat data available in the study area for the years of the MPB outbreak and leading up to and following the 2012 fire. Such data is necessary to conduct change assessments involving time series.

Another issue, also observed by Prichard and Kennedy (2014), is that the MPB outbreak occurred primarily in lodgepole pine, which in relatively homogeneous stands burns at replacement (higher) severity (Agee, 1993). Although the lodgepole pine in the study area often occurred in heterogeneous, mixed species stands, this was still a potential confounding variable of this study, since the effect of MPB was not isolated from the characteristics of the vegetation. This issue was explored using LANDFIRE existing vegetation type (LANDFIRE, 2010) data. For each of the eight major vegetation types (Fig. 1), burn severity was compared for areas affected by Fire-only and by MPB/Fire. These eight classes were further delineated into Fire Regime Group. Group 1 generally consists of vegetation types that support high frequency and low severity fire, Group 3 generally consists of mixed severity fire, and Group 5 generally consists of vegetation types supporting stand-replacing fire. Fig. S3 demonstrates that higher median severity generally occurred in MPB-affected areas versus areas not affected for six of the vegetation types and across all three Fire Regime Groups, although the difference were not significant. Although there are other possible confounding factors (i.e., topography, weather, suppression actions), this indicates robustness to the finding of increased burn severity in MPB-affected forests.

4.3. Effect of harvest type

While a large number of studies have assessed the effect of fuel treatments (e.g., Kennedy and Johnson, 2014; Moghaddas and Craggs, 2007; Ritchie et al., 2007; Safford et al., 2009, 2012; Stephens et al., 2012), this adds to a much smaller body of literature describing the influence of antecedent timber harvest. Aside from Prichard and Kennedy (2014), landscape scale analysis of the interaction between timber harvest and fire has been lacking. In general, past work has indicated lower severity for harvested stands, although findings have depended on treatment type (Cram et al., 2006; Graham et al., 1999; Keyes and O'Hara, 2002; Prichard and Kennedy, 2014; Stephens and Moghaddas, 2005).

This was a unique opportunity, as no known studies have evaluated as many treatment types for the same fire. A few other studies have examined treatments equivalent or similar to the ones that burned most severely in this study. Both Omi and Kalabokidis (1991) and Prichard and Kennedy (2014) found clearcut stands to lower severity compared to untreated stands. However, Weatherspoon and Skinner (1995) noted that plantations (approx. 20 years old) burned more severely than untreated stands.

Results indicated that the combination of shelterwood establishment and removal cuts and stand clearcuts had noticeably higher severities than other types, although not higher than areas affected by MPB or Fire-only. These results might be driven partially by time since treatment, as the oldest treatments (occurring 1971–1982) were comprised of the three treatment types with the highest severity (combination of shelterwood establishment and removal cuts, stand clearcuts, and overstory removal cuts). However, there are also differences in stand structure between treatment types that would promote different type of fire behavior. Stand clearcuts and the combination of shelterwood establishment and removal cuts likely result in similar structure, with the exception that the shelterwood treatments use natural regeneration rather than planting seedlings post-treatment. Single tree selection cuts and commercial thinning had among the lowest severities. These treatments are widely thought to promote multistoried stands resilient to crown fires, as many studies have indicated (e.g., Agee and Skinner, 2005; Graham et al., 1999; Stephens et al., 2009). The results of this study tentatively support that conclusion, but it is important to note that these are also the youngest treatments (so there was less fuel regeneration prior to the fire).

4.4. Effect of harvest age

There is considerable interest in understanding how time-elapsed since treatment affects fire behavior, but there has been little research in this area, particularly for silvicultural treatments. Some studies have attempted to model trajectories in severity for fuel treatment areas, suggesting several decades of reduced severity (Stephens et al., 2012; Strom and Fulé, 2007). Using spectral remote sensing, Prichard and Kennedy (2014) showed only weak correlation between treatment age and severity for treatments up to 30 years old. This study represented a unique opportunity to evaluate a wide range of harvest treatments occurring in a managed forest across a longer period (41 years).

The results indicate that treatments older than 30 years tended to burn more severely. The oldest treatment class (1971–1982) had the highest burn severity, but still not as high as Fire-only or MPB/Fire. Among treatments that are more recent (less than 30 years), no notable difference was observed in severity, suggesting that harvest may reduce severity for a few decades while surface and ladder fuels accumulate. Many of the harvested areas fall into Fire Regime Group 1 classification (LANDFIRE, 2012), defined by low or mixed severity fire on a 35 year or less fire return interval. This would suggest that areas harvested more than 35 years pre-fire could be more susceptible to crown fire, consistent with these findings. However, the lack of differences might also suggest unaccounted-for variability in forest treatment type or burning conditions. Future work exploring the relationships between treatment age, type, burning conditions, and vegetation type would be useful to contextualize the results presented here. Another caveat was that there was far less area represented by most recent treatments (2002–2009) than the other three age classes. Timber harvest was not substantially lower; rather, MPB also affected most treatments during this time.

4.5. Effect of harvest/MPB combination

The results of this study show that the combined effect of MPB and HARV produced the lowest burn severity of any tested. This agent combination also tended to result in the greatest increase in canopy cover (i.e., forest growth) outside the fire perimeter. Compared to MPB or HARV individually, there was less area where these agents overlapped (Table 1), perhaps because of the distinct elevational gradient influencing vegetation type (Fig. 1). Much of the preferred MPB host species (lodgepole pine) occurred at higher elevations within the Wilderness Area, while harvest was limited to lower elevation sites dominated by ponderosa pine and grand fir.

Given the other findings of this study, HARV may be the primary driver of the observed decrease in severity for HARV/MPB/Fire areas, since MPB alone tended to increase severity. This would suggest that the influence of HARV is more substantial than that of MPB, perhaps because canopy fuels are reduced more by HARV than MPB. For instance, a stand clearcut effectively removes trees of all type and size, whereas areas devastated by MPB outbreak will retain smaller diameter trees and species less desirable to the beetle. It is also interesting that the HARV/MPB/Fire areas exhibited a lower severity than HARV/Fire alone, suggesting that the MPB contributes to lower severity when interacting with another agent (i.e., HARV). However, areas affected by both MPB and HARV tended to be lower in elevation than areas affected by MPB only, indicating potentially contrasting fire behavior due to different vegetation species. These remaining uncertainties and continued widespread disagreement in the literature (Table 2; Hicke et al., 2012) suggest further work is necessary to understand how the effect of compounding disturbances vary across landscapes.

4.6. Management implications

While the results presented here may suggest questions about potential management implications, we caution against trying to infer anything conclusive. This was not a controlled study specifically designed to test forest harvest impacts on subsequent disturbance, it was an opportunistic exploratory analysis based on the acquisition of multi-temporal LiDAR following wildfire in an area that with extensive prior forest harvest activity and MPB outbreak. What these results do suggest is that there are considerable questions that arise about the robustness of prior studies conducted with less comprehensive data sets, and that the continued acquisition of LiDAR to support management applications can only improve understanding of the interacting effects of landscape disturbances.

5. Conclusions

This study demonstrated the utility of multi-temporal LiDAR for understanding the effects of antecedent MPB and forest harvest treatments on subsequent burn severity, defined here as change in LiDAR-derived percent canopy cover. The use of multitemporal LiDAR differs from previous research, which has been largely limited to observations of the post-fire environment only (i.e., no pre-fire data), or conclusions made from spectral change (i.e., dNBR and RdNBR) without physiological or biometric validation. The results indicate increased burn severity in areas experiencing MPB, increasing with pre-fire MPB intensity. Conversely, timber harvest led to lower burn severity. Some harvest treatment types burned more severely than others did, but not more severely than untreated stands. Time since treatment appeared to play a role, with treatments older than 30 years experiencing higher average severity than more recent ones. These findings demonstrate quantitatively that pre-fire agents of forest change are important drivers of subsequent fire effects on canopy cover. This is a significant step in resolving uncertainties concerning the increasing overlap of wildfire, MPB, and timber harvest. However, considerable work in this area is still needed. Concerning forest managers, who are increasingly looking to practices that can help manage fire effects (Reinhardt et al., 2008), this study presents compelling evidence that existing pre-fire agents of change strongly influence burn severity in the forest canopy. In context with the results of previous work, these findings can contribute to informed management decisions in similar ecosystems affected by past MPB and/or timber harvest.

Acknowledgements

The authors are grateful for valuable feedback from the editor and anonymous external reviewers. This work was partially funded by the US Forest Service under Joint Venture Agreement 13-JV-11261900-072, and by the Department of the Interior Northwest Climate Science Center through Cooperative Agreement G14AP00177 from the United States Geological Survey (USGS). Its contents are solely the responsibility of the authors and do not necessarily represent the views of the NW CSC or the USGS. This manuscript is submitted for publication with the understanding that the United States Government is authorized to reproduce and distribute reprints for Governmental purposes.

Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2017.02.015.

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