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