Forest Disturbance Interactions and Successional Pathways in the Southern Rocky Mountains

Lu Liang^{a,b}, Todd J. Hawbaker^c, Zhiliang Zhu^d, Xuecao Li^e, Peng Gong^{b,e*}

^a Arkansas Forest Resources Center, University of Arkansas Division of Agriculture,
School of Forestry and Natural Resources, University of Arkansas at Monticello,
Monticello, AR, USA

^b Department of Environmental Science, Policy, and Management, University of
California, Berkeley, CA, USA

^c U.S. Geological Survey, Denver, Colorado, USA 80225

^d U.S. Geological Survey, 12201 Sunrise Valley Drive, Reston, VA, USA 20192

^e Ministry of Education Key Laboratory for Earth System Modeling, Center for Earth
System Science, Tsinghua University, Beijing, China

*Correspondence to: liang@uamont.edu

17

18 **Abstract**

19 The pine forests in the southern portion of the Rocky Mountains are a heterogeneous
20 mosaic of disturbance and recovery. The most extensive and intensive stress and
21 mortality are received from human activity, fire, and mountain pine beetles
22 (MPB; *Dendroctonus ponderosae*). Understanding the disturbance interactions and
23 disturbance-succession pathways are crucial for adapting management strategies to
24 mitigate their impacts and anticipate future ecosystem change. Driven by this goal, we
25 assessed the forest disturbance and recovery history over a 13-year time series in the
26 Southern Rocky Mountains Ecoregion with Landsat image stacks. An automated
27 classification workflow that integrates temporal segmentation techniques and a random
28 forest classifier was used to examine disturbance patterns. To enhance efficiency in
29 selecting representative samples at the ecoregion scale, a new sampling strategy that
30 takes advantage of the scene-overlap among adjacent Landsat images was designed. The
31 segment-based assessment revealed that the overall accuracy for all 14 scenes varied
32 from 73.6% to 92.5%, with the mean at 83.1%. A design-based inference indicated the
33 averaged producer's and user's accuracies for MPB mortality were 85.4% and 82.45%
34 respectively. We found that burn severity was largely unrelated to the severity of pre-fire
35 beetle outbreaks in this region, where the severity of post-fire beetle outbreaks generally
36 decreased in relation to burn severity. Approximately half the clear-cut and burned areas
37 were in various stages of recovery, but the regeneration rate was much slower for MPB-
38 disturbed sites. Pre-fire beetle outbreaks and subsequent fire produced positive compound
39 effects on seedling re-establishment in this ecoregion. Taken together, these results
40 emphasize that although multiple disturbances do play a role in the resilience mechanism
41 of the serotinous lodgepole pine, the overall recovery could be slow due to the vast area
42 of beetle mortality.

43

44 **Key words:** Landsat; fire; mountain pine beetle; time series; disturbance; recovery

45

1. Introduction

The vegetation of the Southern Rocky Mountain Ecoregion is a heterogeneous disturbance-derived mosaic, where fires, blow-downs, insect outbreaks and clear-cuts are regarded as the primary disturbance types (Barbour & Billings, 2000; Veblen et al., 1994). Under pressure from a warming climate and human interventions, the spatial and temporal dynamics of disturbance and recovery are undergoing rapid changes in the subalpine zones of the Rocky Mountains (Turner, 2010). Fire is perhaps the most commonly studied disturbance agent in this ecoregion, with a dramatic rise in the total amount of area burned as well as large-fire occurrences in the past decades (Justice et al., 2002; Simon et al., 2004; Calkin et al., 2005; Stephens, 2005). Since the mid-1990s, massive conifer mortality has been related to outbreaks of bark beetles in this ecoregion: primarily mountain pine beetle (MPB), spruce beetles, and western balsam bark (Meddens et al., 2012; Smith et al., 2015). Among them, MPB mortality has expanded in high elevation conifers in the Rocky Mountains reaching epidemic levels not previously recorded (Gibson et al., 2008). The coupling of disturbances with environmental factors determines the various rates and pathways of succession and results in heterogeneous patterns of forest recovery (Pfulgmacher et al., 2013). Disturbance together with recovery can influence forest structure, composition, biomass, dynamics of land surface water and radiation, and affect feedbacks between forest ecosystems and climate (Baldocchi, 2008; Bonan, 2008; Franklin et al., 2002; Gough et al., 2007; Kurz et al., 2008; Liang et al., 2014a). Thus, understanding the patterns of disturbance and recovery as well as their interactions is crucial for assessing their impacts on forest resilience.

The occurrences of multiple disturbances in this ecoregion provide us an opportunity to investigate their interactions - a rising concern in ecology because they may cause abrupt changes in an ecosystem quality, property, or phenomenon (White & Pickett, 1985; Groffman et al., 2006; Turner, 2010). However, our ability to investigate disturbance interactions remains challenging due to the scarcity of quantitative data. A disturbance may be linked with another by changing its extent, severity, or probability of occurrence. For the Southern Rockies, there is substantial interest in how the extensive outbreaks of

bark beetles may affect future wildfire (Turner, 2010). While spruce beetle outbreaks were determined to have little to no overlap with stand-replacing fires in the Colorado Rocky Mountains (Veblen et al., 1994), conclusions regarding MPB and fire interactions remain controversial. Increased (Hopkins, 1909; Schoennagel et al., 2012), decreased (Simard et al., 2011), and unrelated (Bond et al., 2009; Veblen et al., 1994) active crown fire potentials were found following MPB outbreaks, depending on the outbreak stages and burning conditions (Harvey et al., 2014). Additionally, there is no consensus whether or not MPB outbreaks following fires affect forest vegetation. Although it is presumed that fire-injured forests are more susceptible to beetles (Geiszler et al., 1984), some have pointed out that wildfire is an unlikely driver of MPB outbreaks (Powell et al., 2012). Depending on the order of occurrence, we can categorize their interactions into two types: fire occurs after MPB outbreaks, and MPB outbreaks occur after fires. In this study, we will determine their linkages by investigating how the recent bark beetle outbreaks have influenced subsequent fire occurrences, and whether fire has affected the severity of subsequent beetle outbreaks. If such a link exists, they may produce compound effects on ecosystems. For instance, the interactions may amplify severity beyond that of any singular disturbance, or create of novel characteristics that would not normally happen in isolation (Buma & Wessman, 2011), such as the shifting the dominant forest cover types (Johnstone et al., 2010) and reducing regeneration rates (Buma & Wessman, 2011). Understanding the consequences of those interactions is thus of critical importance for conservation and management purposes.

To address the above forest ecology questions, a spatial-temporal characterization of the patterns of forest disturbances is essential and several approaches exist.

Dendrochronology is a useful technique to reconstruct disturbance histories at the tree level but the geographical scope is limited. Aerial surveys, which are conducted by state and federal agencies, provide a record of insect and disease mortality beginning in the mid-twentieth century. However, the data quality has been criticized for its spatial imprecision, its temporal shift between the attack onset and the spectral signature changes of canopy, and its lack of complete coverage (Meigs et al., 2011; Kautz, 2014). Landsat time series stacks that are capable of detecting subtle and chronic changes in forests have

been proven to be useful in assessing the extent and severity of stand-level MPB disturbance over long time spans (Meddens et al., 2013; Liang et al., 2014b; Meddens & Hicke, 2014; Meigs et al., 2015). However, few studies have been conducted to assess their ability in detecting the spatial-temporal pattern of bark beetle-caused mortality at the ecoregion-scale (Meddens et al., 2012; Creeden et al., 2014).

The purpose of this paper is three-fold. First, we quantified the changes of disturbance and recovery patterns over the past decade in the Southern Rocky Mountain Ecoregion with the focus on MPB disturbance. Second, we assessed whether MPB outbreaks and wildfire were linked disturbances in this ecoregion. Finally, we compared the effects of various types of compounded disturbances on lodgepole pine forest regeneration with singular disturbances.

2 Methods

2.1 Study area

The Southern Rockies Ecoregion extends south from southern Wyoming through Colorado to northern New Mexico (lower left: ~35°N, 109°W; upper right: ~43°N, 103°W) (Figure 1). To avoid confusion with other types of insect mortality (e.g. spruce vs. pine beetles), we confined our analysis to the extent of MPB host species in the LANDFIRE Refresh 2001 Existing Vegetation Type data layer, which was developed with *circa* 2001 Landsat imagery (Nelson et al., 2013). The extracted MPB host species mask was smoothed with a 3×3 majority filter to reduce noise and to minimize the reported high omission errors (National c2001 Assessment; Figure 1). The selected MPB host species included lodgepole pine (*P. contorta*; 95.4%), limber pine (*P. flexilis*; 2.9%) and ponderosa pine (*P. ponderosae*; 1.7%). Their total coverage was approximately 60,675 km². Hereafter, these forest types are referred to as ‘forest’.

Figure 1. The coverage of the Southern Rocky Mountains Ecoregion (U.S. Environmental Protection Agency, 2013) and the boundaries of 14 intersecting Landsat scenes used in this study.

2.2 Remote sensing image pre-processing

Landsat time series stacks are well suited to this study, because of their free availability, long time period coverage, broad spatial extent, multispectral characteristics, and temporal continuity. All Landsat images were acquired from the Landsat Surface Reflectance Climate Data Record that was generated by the Landsat Ecosystem Disturbance Adaptive Processing System (URL: <http://earthexplorer.usgs.gov>; accessed: 22 August 2014). We restricted our analysis to images acquired in June through October.

June to September is the most active growing season for vegetation and thus provides the strongest spectral contrast between healthy forests and forests with beetle mortality. Images in October usually have better quality than summer in this mountainous region and were used as substitutions when clouds heavily contaminated all images in the June to September date range. We tried to find images containing less than 20% cloud cover, and if no such data existed, two or more cloudy images with similar phenology characteristics were used as replacements to ensure there was at least one clear observation each year. Fourteen Landsat scenes were intersected with the ecoregion's boundary, and 247 images were collected from 1999 to 2011 in total with approximately 18 images per scene (Figure S1).

2.3 Segment-based random forest classification

Disturbance processes operating at both fast and slow rates on landscapes drive long-term dynamics in forest properties. From the perspective of remote sensing, the processes can be characterized by the time series of spectral value or vegetation indices derived from satellite observations. As a forest experiences multiple events, such as regrowth, abrupt, and chronic disturbance, its time series trajectory will have distinct characteristics corresponding to the disturbances and periods of post-disturbance recovery. Thus, the trajectory can be decomposed into a sequence of straight-line segments, with each segment representing one salient event (Figure 2, Step 1). For example, if a healthy forest was burned and recovered afterwards, its curve can be broke down into three segments (Figure S2 d). The first segment covering the healthy period can be recognized as a flat line, and the second segment for the disturbance period tends to be a declining line. The third segment happening over the recovery period is a line with positive slope. Different disturbance events possess different segment shapes. For instance, clear-cuts can be recognized as lines with rapid and severe spectral value changes, whereas MPB mortality segments tend to have lower rates of change in spectral values that span several years. Based on this idea, the multi-temporal classification can be summarized into two questions: 1) how to best capture the event segment? and 2) how to attribute the segments with the associating events? In this section, we presented a segment based classification

framework to address these two questions and an overall work flow was displayed in Figure 2.

Figure 2. The classification work flow. The time-series vegetation index trajectory extracted from the Landsat image stacks is first decomposed into several segments after noise removal and model simplification (Step 1). A set of features for the focal segment as well as their spatial-temporal neighborhood features are built (Step 2). Training samples are selected based on scene overlaps (Step 3). The features and samples are then used together in a random forest classification to label the event type of the focal segment (Step 4), and finally the accuracy of the classification process is evaluated (Step 5).

Step 1: Temporal segmentation

Temporal segmentation is one useful technique that is designed with a set of parameters to decompose the time-series curve into distinct segments. For this study, we used the temporal segmentation approach implemented in the Landsat time-series in Landsat-

based detection of Trends in Disturbance and Recovery tool (Landtrendr; Kennedy et al., 2010). For the Southern Rockies, the parameters were calibrated with a set of training sites in our previous work (Liang et al., 2014a). The goal of temporal segmentation is to minimize the omission of disturbance detection and maximize signal to noise ratio in the time series. Time series are often constructed using the Normalized Burn Ratio (NBR) spectral index because of its sensitivity to disturbance (Kennedy et al., 2010), its capacity to minimize radiometric differences among images, and its robust performance in reducing omission error of disturbance events (Cohen et al., 2010). To conduct the temporal segmentation, the spikes in often-noisy NBR time series were removed if the proportional difference in NBR values between two adjacent points was less than a preset parameter. Then, a set of linear regressions were applied across the NBR time series at each pixel, and the resulting residual error at each time step aided in the selection of potential vertices that define the starting and ending points of new regression segments. This process was repeated until a statistical threshold was reached (Kennedy et al., 2010). Figure S2 displays a typical NBR time series and the curve fit for four distinct events (Figure S2).

Step 2: Segment feature construction

The temporal segmentation generated one or more fitted segments for each pixel, and the next step was to identify the event labels for these segments. To enable automatic labeling with machine learning techniques, we generated a total of forty-two features to train a random forest classifier. The technical details are explained below:

Segment features: Several key segment features in differentiating various disturbance and recovery events included onset year, magnitude, duration, and slope. A vertex year is the starting year of the event occurrence, i.e., the turning point of a segment. The fitted vertex value is the NBR value at the vertex. Magnitude is defined as the absolute NBR change between the head and tail vertex of the focal segment ($|NBR_{head} - NBR_{tail}|$), which is associated with the percent vegetative cover change in the Pacific Northwest region (Kennedy et al., 2010). Duration denotes the time span from

the head to the tail vertex, and can indicate the most dominant event over the time series for a particular location. Slope is the ratio of magnitude to duration (Figure 3).

Besides the focal segment's features, we also constructed several spatial neighborhood features and temporal neighborhood features. They were designed based on the assumption that disturbance and recovery events tend to be clustered in space and time.

Temporal neighborhood features included duration, magnitude, and slope of the prior and posterior segments of the focal one.

Spatial neighborhood features were calculated based on the slopes and vertex values of the eight spatially adjacent pixels over the duration of the focal segment. The slope set is denoted as N_s and the vertex value set is defined as N_f :

$$N_s = \{(S_{a1}, S_{b1}, \dots, S_{n1}), (S_{a2}, S_{b2}, \dots, S_{n2}), \dots, (S_{a8}, S_{b8}, \dots, S_{n8})\}$$
$$N_f = \{(F_{a1}, F_{b1}, \dots, F_{n1}), (F_{a2}, F_{b2}, \dots, F_{n2}), \dots, (F_{a8}, F_{b8}, \dots, F_{n8})\}$$

Where, S is slope and F is the fitted vertex value. The first subscript a to n indicates the first year a of the time span to the last year n, and the second subscript 1 to 8 represents the first to the eighth neighboring pixel. For example, S_{a1} is the slope of the first neighboring pixel in the first year of the focal segment duration.

Following that, we calculated the maximum value for each subset in N_s to highlight the event with the most significant change (max8slope), and computed the mean and the standard deviation of max8slope to represent the overall condition in the neighborhood system (avg_max8slope; std_max8slope). The same processing routine was applied to N_f , with an additional minimum value calculation (min8fit, avg_min8fit, and std_min8fit).

245 Table 1. Segment features and clarification.

Category	#features	Clarification
Vertex1_year	1	Year of the head vertex
Vertex1_NBR	1	Normalized Burn Ratio (NBR) value of the head vertex
Vertex2_NBR	1	NBR value of the tail vertex
Dur	3	Duration of the segment: {pre_cur, cur_dur, post_dur}
Mag	3	Magnitude of change in NBR between the head and tail vertex: {pre_mag, cur_mag, post_mag}
Slope	3	Magnitude/duration: {pre_slope, cur_slope, post_slope}
max8slope	8	Max slopes over the duration of the focal segment for its eight spatially adjacent pixels: {max(S _{a1} ,S _{b1} ,...,S _{n1}), max(S _{a2} ,S _{b2} ,...,S _{n2}),..., max(S _{a8} ,S _{b8} ,...,S _{n8})}
avg_max8slope	1	Mean of Max8slope: mean {max(S _{a1} ,S _{b1} ,...,S _{n1}), max(S _{a2} ,S _{b2} ,...,S _{n2}),..., max(S _{a8} ,S _{b8} ,...,S _{n8})}
std_max8slope	1	Standard deviation of max8slope: Std {max(S _{a1} ,S _{b1} ,...,S _{n1}), max(S _{a2} ,S _{b2} ,...,S _{n2}),..., max(S _{a8} ,S _{b8} ,...,S _{n8})}
max8fit	8	Maximum fitted NBR value over the duration of the focal segment for its eight spatially adjacent pixels: {max(F _{a1} ,F _{b1} ,...,F _{n1}), max(F _{a2} ,F _{b2} ,...,F _{n2}),..., max(F _{a8} ,F _{b8} ,...,F _{n8})}
avg_max8fit	1	Mean of max8fit: avg {max(F _{a1} ,F _{b1} ,...,F _{n1}), max(F _{a2} ,F _{b2} ,...,F _{n2}),..., max(F _{a8} ,F _{b8} ,...,F _{n8})}
std_max8fit	1	Standard deviation of max8fit: Std {max(F _{a1} ,F _{b1} ,...,F _{n1}), max(F _{a2} ,F _{b2} ,...,F _{n2}),..., max(F _{a8} ,F _{b8} ,...,F _{n8})}
min8fit	8	Minimum fitted NBR value over the duration of the focal segment for its eight spatially adjacent pixels: {min(F _{a1} ,F _{b1} ,...,F _{n1}), min(F _{a2} ,F _{b2} ,...,F _{n2}),..., min(F _{a8} ,F _{b8} ,...,F _{n8})}
avg_min8fit	1	Mean of min8fit: avg {min(F _{a1} ,F _{b1} ,...,F _{n1}), min(F _{a2} ,F _{b2} ,...,F _{n2}),..., min(F _{a8} ,F _{b8} ,...,F _{n8})}
std_min8fit	1	Standard deviation of min8fit: Std {min(F _{a1} ,F _{b1} ,...,F _{n1}), min(F _{a2} ,F _{b2} ,...,F _{n2}),..., min(F _{a8} ,F _{b8} ,...,F _{n8})}

246

Step 3: Sample selection

Before we ran the random forest classification, a necessary step was to acquire training and test samples that contained information on disturbance history. A basic principle in designing an efficient sampling strategy at the ecoregion scale is to select samples that are representative enough to cover most of the population, while at the same time requires the least amount of sampling effort to produce adequate estimates of the population.

To achieve this goal, we developed a new strategy by intensively selecting samples in the Landsat scene-overlapping area (SOA) between images along the same row, which covers 20% to 40% of the entire scene. The intention is to save efforts in image interpretation or on-site data collection in the overlapping areas. Meanwhile, although 20%-40% of the full scene coverage is a good representation of the landscape, to make the final sample set an unbiased indicator of the population, we automatically added spectrally dissimilar samples from the non-overlapping area (NOA). Specifically, our sampling strategy involved the following steps (Figure 3):

- 1) *Clustering*. Each Landsat scene was split into SOA and NOA, and K-Means clustering was applied to the 13-year SOA and NOA image stacks separately. Multi-date clustering is commonly applied in change detection analysis. We used it here to find groups of pixels such that all the pixels in one cluster share similar NBR trajectories, and pixels from different clusters present different spectral characteristics.
- 2) *Calculate spectral distances between cluster centers*. The spectral distance between the center of each SOA and NOA cluster was calculated.
- 3) *Remove spectrally similar NOA clusters*. NOA clusters that shared similar spectral signature with any SOA cluster were treated as duplicated samples and removed. That is, only NOA clusters with the distance of their centers to all SOA cluster centers larger than one standard deviation of the distance set were retained.
- 4) *Stratified random sampling and event labeling*. The combination of all SOA clusters and non-duplicated NOA clusters should cover the majority of NBR

trajectories that could be encountered in the two adjacent images. A total of 300 pixels were drawn from those retained clusters via stratified random sampling, with the sample size proportional to the total number of pixels in each cluster. For each sample, we interpreted its disturbance history with the aid of NBR trajectories, false-color composite Landsat layers, and high resolution Google Earth images.

To better illustrate how the SOA and NOA clusters interact in the spectral space, we used one scene as an example by displaying the cluster centers from both regions in the Tasseled Cap space (Figure S3). In this case, if the centers of the non-duplicated NOA clusters were further away from the SOA cluster, it implies that our sampling strategy can successfully pick those representative samples with low omission error.

Figure 3. Sampling workflow. SOA: Landsat scene-overlapping area; NOA: non-overlapping area; dis: spectral distance; CI: one standard deviation of the spectral distance.

Step 4: Random forest classification for segment labeling

With the derived segment features and the training data, we then applied the random forest to classify the disturbance and recovery events based on their distinct features. Five broad classes that represented major forest health status in the Southern Rockies were classified: healthy, MPB mortality, fire, clear-cut, and regrowth. For sudden canopy removal events, we separated fires from clear-cuts using the U.S. Geological Survey Monitoring Trends in Burn Severity database (Eidenshink et al., 2007). We then applied a plurality vote to the ten random forest outcomes generated from different training partitions and produced the annual time-series maps by assigning the segment label to their corresponding years.

Step 5: Accuracy assessment

We evaluated the classification performance in three ways: segment-based assessment, ground-based assessment, and design-based inference.

Segment based assessment was conducted via 10-fold cross validation on the sample set collected from step 2. The overall accuracy (OA) was calculated by summarizing the percentage of segments that were labeled the same by the random forest classifier and our samples. This assessment was based on the assumption that the temporal segmentation was flawless. It was made because it is difficult to obtain historical, temporally-explicit ground truth data across the whole ecoregion.

Ground-based assessment is a preferred technique for evaluating classifications. A total number of 119 plots with global positioning system coordinates were visited by USGS crew in a field survey in the summer of 2010. Each plot had a 8-m radius, and the information recorded at each plot included forest type, number of dead, sick and healthy trees, disturbance agent, canopy height, live canopy percentage, and composition. A plot

was considered healthy if the total live canopy percentage of healthy trees was more than 90%, otherwise it would be assigned to the main disturbance agent. However, it should be noted that the target of the survey was the MPB infested forest, which might introduce some bias in evaluating classification performance on other disturbance types. A confusion matrix was built for each of the above methods. The overall accuracy, per class user's and producer's accuracies were calculated.

Recognizing that field samples were spatially and temporally scarce, we employed design-based inference to calculate the map accuracy statistics. Time series test samples were acquired from a stratified random sampling on ADS data and National Agriculture Imagery Program (NAIP) imagery. Specifically, we refined ADS polygons that were associated with MPB using the Damage Causal Agent attribute code and clipped the boundary to the extent of the ecoregion for each year. Based on the onset timing detected by ADS, the landscape was partitioned into 12 strata and a total of 200 points were then proportionately allocated to each stratum. In this manner, our test samples were able to cover a complete gradient of MPB mortality. Subsequently, each point was visually interpreted against the 1-m resolution NAIP imagery. Since the acquisition of NAIP imagery is usually at a three-year cycle and varies among regions, the number of test samples that can be interpreted each year was uneven. The total number of validation samples was 126 (year 2005), 54 (year 2006), 194 (year 2009), and 133 (year 2011). Similar to the ground-based assessment, confusion matrices and associated metrics were built.

2.4 Disturbance interactions and disturbance-recovery pathways

To determine the relationships between beetle outbreaks and fires, we first allocated pixels into four groups based on their disturbance history: MPB+Fire (fire happened after beetle outbreaks) and Fire+MPB (fire and subsequent MPB outbreaks) cases referred to the assemblage of pixels containing the process of the two interaction modes. The collection of pixels possessing either beetle outbreaks or fire events (MPB/fire), which were designed to show the whole spectrum of outbreak or burn severity occurring over the study region. We then assessed the magnitude values for MPB and fire events in cases

and controls separately using analysis of variance (ANOVA). The significance test on the means of cases and controls provided a way of examining the effects of one disturbance event on the other. For instance, if no significant difference in the NBR change magnitude of MPB was found between Fire+MPB case and MPB control, fire probably did not play a role in changing the severity of beetle outbreaks. However, it is important to note that the Fire+MPB interactions are highly dependent on species attacked. In this analysis, we included three major MPB host species, separately lodgepole pine, limber pine and ponderosa pine. Forests dominated by the first two species typically experience stand-replacing crown fires which leave few surviving host trees that could support subsequent MPB attack. In contrast, forests dominated by ponderosa pine are more fire-tolerant, which can survive low to moderate intensity fires and are consequently have the potential to support subsequent MPB attacks. To gain a better understanding of the Fire+MPB interactions and which forest types they occurred in, we overlaid the burned pixels with subsequent MPB mortality with the LANDFIRE existing vegetation type map. Finally, we depicted the disturbance-recovery pathway by summarizing the area and proportion of each disturbance event that transitioned to either another disturbance type or to the recovery stage.

3 Results

Some key points of our research results are summarized as following: The classification on the long-term forest disturbance and recovery achieved satisfying performance as reflected from our three independent evaluation processes. Based on this series of remote sensing derived maps, we found that burn severity was largely unrelated to the severity of pre-fire beetle outbreaks in this region, but the severity of post-fire beetle outbreaks generally decreased in relation to burn severity. The recovery rate in the clear-cut and burned areas was much faster than that in MPB-disturbed sites. MPB+Fire interactions could produce positive compound effects on seedling re-establishment in this ecoregion.

3.1 Algorithm assessment

The segment based OA for individual scenes varied from 73.6% to 92.5%, with a mean value of 83.1% (Table S1).

Because our mapping boundary was confined by the extent of MPB host species derived from the LANDFIRE Refresh 2001 Existing Vegetation Type data layer, we found that 37 field samples were located outside of our mapping extent. If those points were excluded, only one healthy plot was misclassified as having beetle mortality, and the overall accuracy was 98.7% (Table S2). But if considering the missing of those 37 samples as a part of the omission errors in classification due to the study area confinement, the omission error for MPB mortality became 32.7% and the overall accuracy was down to 67.2%.

Based on the design-based assessment in years with NAIP imagery, the overall accuracies in year 2005, 2006, 2009 and 2011 were 73.8%, 83.3%, 92.8%, and 93.9%, respectively (Table S3). The averaged producer's and user's accuracies for MPB mortality were 85.4% and 82.45% respectively. By carefully examining the misclassified pixels, we observed that most areas of confusion happened in forests with low canopy cover.

3.2 Disturbance interactions and successional pathways

Despite the fact that the Southern Rockies experienced a diverse range of disturbances, we did not observe a substantial amount of area where a secondary disturbance happened after a primary disturbance within the decade-long time period of this study. Only 2.3% (57 km²) of the area burned by fires had subsequent MPB outbreaks, and that mostly happened in fire-tolerant ponderosa pine forests (31.4 km²) or douglas-fir-ponderosa pine-lodgepole pine mixed forests (18.8 km²), with only a small portion (6.8 km²) occurring in fire-intolerant limber pine and lodgepole pine forests (Table S4, Figure S4). The percentages were even smaller for fires and clear-cuts in MPB infested forests, which were 1.7% and 0.8% of the area affected by MPB mortality, respectively.

Although the probability of disturbance interactions was low in this region, their combined impact could be different than the impacts of individual disturbances. As revealed from the ANOVA test, there was a significant difference in the magnitude of MPB and fire events between the Fire+MPB case and controls, but no significant difference was found between the MPB+Fire case and controls for both events. Their comparison also revealed that stands experiencing interaction were attributed with higher fire magnitude but lower beetle outbreak magnitude than those without (Figure 4). However, it is important to note that the classification of disturbance types was not perfect, and the differences in spectral values may not fully represent the actual disturbance severity, which add uncertainty to these findings.

Figure 4. Boxplots displaying the Normalized Burn Ratio change magnitude of cases and controls. Boxes with solid color represent the case group and those with dash lines represent the random samples selected from the whole population besides the case group. The bottom and top of the box are the first and third quartiles, and the band inside the box is the median. The whiskers represent the 1.5 interquartile range of the lower and upper quartile. MPB: mountain pine beetle.

We outlined the major successional pathways in the South Rocky Mountains Ecoregion (Figure 5). During the 13-year period of this study, the forest area recovering from all kinds of disturbances was 2,539 km², which accounted for 24.2% of the total disturbed area. To understand what was happening in the disturbed regions showing no sign of recovery till year 2011, we summed the area for each specific event according to their onset years (Figure S5). It was found that most of those regions experienced very recent disturbances. For instance, 64.6% of the disturbances with no regrowth happened between 2006 and 2011 (Figure S5). This implies that the long-term recovery rate may be ultimately higher than 24.2% since most disturbed areas were disturbed relatively recently.

For places that experienced a single disturbance, burned areas had the most regrowth, with a total area of 1,428 km². Sites with only stand-replacing clear-cuts had the second largest area in regrowth (338 km²), which was 49.8% of the total clear-cut area. MPB disturbed forests had the least amount of area classified as recovering (662 km²). The regeneration rate, which was defined here as the ratio of post-disturbance regrowth area to the disturbed area, was 57.9% for burned sites and 49.8% for clear-cut sites over the 13-year study. However, only 9% of the forest area was recovering within the MPB-induced mortality zones. In terms of multiple disturbances, their compounded effects on the subsequent recovery were evident. The regeneration rate for Fire+MPB was 14.9%, and MPB+Fire produced the highest regeneration rate (64.9%) amongst all kinds of singular and compounded disturbances.

Figure 5. Disturbance and recovery pathways in Southern Rocky Mountains Ecoregion. Percentages in the arrow indicate the percentage of area in the left box to that in the right. MPB: mountain pine beetle.

4 Discussion

4.1 New classification framework and uncertainties

Mapping over a decade of forest disturbance history at the ecoregion scale using moderate spatial resolution satellite imagery presented many challenges that led to our development of new methodological techniques. The mean overall accuracies of 83% (segment-based assessment) and 86% (design-based assessment) we achieved are satisfying considering the large-scale geographic coverage, the high levels of landscape heterogeneity, and the diversity of disturbance agents. The performance should be encouraging to others considering the use of temporal trajectory analysis (e.g. Kennedy et al., 2010) and the new ideas developed for this study, specifically: 1) Our mapping approach is based on segments instead of pixels, which can reduce illogical annual changes since the labels are consistent over the duration of the segment. 2) The inclusion of spatial-temporal neighborhood features not only improves the robustness of classifier's performance, but also minimizes the "salt-and-pepper" effects from both spatial and temporal perspectives. 3) The new sampling strategy that selects samples in the

overlapping areas between adjacent Landsat scenes reduces the time required for image interpretation or on-site data collection. By further including NOA samples sharing dissimilar spectral features with the SOA samples, the overall spectral variability as well as the sample representativeness is enhanced. This is of particular importance for parametric algorithms (Gong et al., 2013). 4) Additionally, the use of LANDFIRE vegetation type data to confine the extent of MPB host forests was an effective way to avoid the inclusion of non-MPB host conifer sites. Those sites might show similar spectral properties as the host pixels, which can make the classification of regions with beetle mortality more difficult.

Nevertheless, two weaknesses are evident in this mapping approach. As revealed from the ground-based assessment, 38 out of 119 samples were not located within our mapping extent. This type of omission error is propagated from the LANDFIRE Refresh 2001 Existing Vegetation Type data layer. Thus, although the mapping accuracy within the study area was as high as 98.7%, the omission error for MPB disturbance was up to 32%. In addition, because of incomplete temporal segments and loss of previous year's segment information, the classification accuracies for the first few years were relatively low. Incomplete segments could cause miscalculations on certain feature metrics. For instance, the duration is shorter and the absolute magnitude of change is lower than in reality. Meanwhile, because no temporally adjacent segments from previous years can be used, the event labels for the first years of the trajectory would be less accurately predicted by random forest than those in all other years. If data allows, we suggest readers who plan to use this algorithm to extend their study period, but to exclude the first few years of results in their analysis.

Moreover, several other limitations not immediately evident might impact the overall accuracy. First, the temporal segmentation was run on a unified set of parameters that were calibrated with data from Grand County, Colorado (Liang et al., 2014a), where there is a high concentration of lodgepole pine forests and MPB mortality. The differences in the density of MPB populations and host trees might have impacted the segmentation outputs, and resulted in a certain amount of false positive errors in MPB

mortality pixels. Second, segments showing slow and gradual declining might not necessarily correspond to MPB-induced mortality. Forests experiencing drought-induced tree die-off, disturbances from other insects or disease-causing pathogens and mistletoes, or thinning/partial harvest, could produce a similar trajectory and be incorrectly attributed to MPB-mortality. Restricting our analysis within the MPB host tree species can minimize but not entirely avoid this confusion. Thirdly, at the Landsat pixel scale, variability exists in the duration of beetle attack, which could range from 1-2 years to more than 6 years, with the average as 3-4 years (Meddens et al., 2014). The violation of our assumption that insect-induced mortality usually lasts shorter than abrupt disturbances could result in commission errors.

4.2 Interactions between fire and MPB outbreaks

Although the area of post-MPB fires was almost double than that of post-MPB clear-cuts, its coverage was fairly small compared to the vast area affected by MPB. Thus, it remains challenging to draw any solid conclusions about the role of MPB in increasing the risks of fire hazards. There is a growing body of ecological research exploring the relationships between MPB and fire, and some have conflicting conclusions (Hicke et al., 2012; Schoennagel et al., 2012; Simard et al., 2011). Most of them were carried out by measuring the surface and canopy fuels in a time-since-beetle-outbreak chronosequence at a limited number of field sites to parameterize fire simulation models, and to predict future potential fire behavior. However, beetle-fire interactions could be nonlinear under different environments (Simard et al., 2011), and the small quantities of on-site data used in the past studies may not fully capture the underlying heterogeneity and variation. Our satellite measures, although lacking the details of the site-level collections, are capable of providing information about where and when those events happened at the landscape level.

In the Southern Rockies, fire occurrence was largely unrelated to pre-fire MPB outbreak severity. For fires that burned through beetle-impacted forests, their severity did not show any significant difference from fires that burned through forests not affected by beetles. Beetles can affect the forest ecosystem in a variety of ways. Wind penetration through

stands may increase if beetle killed trees started to fall. But we did not observe a large proportion of fallen trees in this region during field visits and on high-resolution images. The surface and canopy fuel would be altered after the death of individual trees, but not in a consistent way. Shortly after beetle mortality, canopy fuels are high due to the declining foliage moisture. Over several years, the fuel load in the canopy layer may be reduced because limbs and trees fall, whereas the surface fuel loads would increase because of the accumulation of fallen needles and branches. Besides antecedent surface fuel conditions, relative humidity, wind speed, topography, burning conditions, proportion of stand infested (mortality level) as well as the temporal duration since infestation all significantly affect the fire occurrence after beetle infestation (Bigler et al., 2005; Collins et al., 2007; Harvey et al., 2013; Romme et al., 2006). Because of the varying and complex environmental conditions, readers should be aware that conclusions drawn here might not be valid for every place across the ecoregion. And it is likely that the contrary characteristics at different places can be counteracting each other and lead to the uncorrelated relationship that we observed. Future research could be conducted to find the reasons for this phenomenon. Our study documented the fire occurrence after beetle infestation at the ecoregion scale. Looking for the broad scale patterns of post beetle mortality fires is a precursor to do detailed field-based studies about the mechanisms that led to them.

Most Fire+MPB interactions occurred in fire-tolerant ponderosa pine forests and mixed woodlands. Most fire-intolerant lodgepole and limber pines in the Southern Rockies did not support subsequent beetle attack after fire disturbance, because beetles can not survive in dead or severely damaged trees. We found that in the fire-tolerant forests, post-fire beetle severity tended to decrease with burn severity, which suggested that MPB usually caused less damage in forests where had a prior fire event. Wildfire can impose immediate stress on trees. A prominent effect of fire injury is reflected in the low substrate quality, as the brood production declined with the degree of fire injury (Powell et al., 2012). Hence, heavily burned stands would harbor fewer beetles and resulted in milder outbreaks. Moreover, fires consume a larger amount of biomass. If fire-affected

stands did not recover back to their healthy status before the beetle attack, the decline in canopy cover will be less evident as compared with beetle attack on healthy trees.

4.3 Impacts of singular and compounded disturbance on forest resilience mechanisms

Our analysis found that more than half of the burned area has subsequent regrowth. The dominating species in our study area is lodgepole pine, which typically has serotinous cones and need high temperatures to trigger and release the seeds. This fire-dependent characteristic allows for post-fire tree regeneration soon after fires (Turner et al., 1997), but the abundance of serotinous cones can vary with fire regimes and pre-fire seed density (Schoennagel et al., 2006). Lodgepole pine start seed production at an early age, approximately 5-10 years (Burns & Honkala, 1990). And thus it is not surprising we found such high regeneration rates.

Another question that many researchers and stakeholders are concerned with is how forests respond to post-MPB disturbances? Our results showed that a small portion of the MPB disturbed forests was on a trajectory to full recovery. The remaining areas were either under disturbance or recovering at a slower rate than areas affected by other types of disturbance. These observations may be related to the warmer and drier climate conditions that have been unfavorable for tree regeneration in recent years. Additionally, if the size of disturbed patches is large enough, regeneration could be limited by the short dispersal distances of tree species (Bentz, 2010; Kurz et al., 2008; Sambaraju, 2012).

Interactions between disturbance processes can amplify or mute the overall effects of changes (Vose et al., 2012). The outcome is largely determined by the magnitude and direction of effects they created, as well as the order of events. One interesting piece of evidence we found is that beetle outbreaks could alter ecosystem response to subsequent fire by increasing the post-fire regeneration rate of serotinous lodgepole pines in this ecoregion. Despite the fact that beetles might reduce seed availability by killing large seed-producing trees (Bjorklund & Lindgren, 2009), they can help in redistributing the remaining viable cones and seeds through wind and animal dispersal (Chambers &

MacMahon, 1994), and thus promote the generation of a larger number of seedlings as aided by the following fire. However, this finding may only be valid in this particular ecosystem and under specific conditions, such as stand age, density, and burn severity, because we other studies have drawn different conclusions. For instance, Harvey et al. (2014) suggested that there was no direct effect of outbreak severity on initial post-fire regeneration of lodgepole pine in Greater Yellowstone.

5 Conclusions

In this study, we documented the disturbance and recovery history in the lodgepole pine forests of the Southern Rocky Mountains Ecoregion using Landsat time series classification maps. The maps were generated from a new SOA/NOA sampling strategy, the integration of Landtrendr with a random forest classifier for disturbance type attribution and the inclusion of spatial-temporal neighbor features. By tracking the spatial- temporal patterns of MPB mortality and burned areas, we found that the linkage, strength, and direction of interactions between bark beetle mortality and fire to be related to their sequence of occurrence. The severity of fire was unlinked to pre-fire outbreak severity, and no evidence was found towards the role of MPB in increasing fire occurrence. However, post-fire beetle severity tended to be lowest in areas with higher burn severity, likely due to the limited availability of host trees. Most recovery events happened immediately after stand-replacing disturbances. Whereas beetle mortality that disturbed the most area showed an extremely low regeneration rate. Although the probability of multiple disturbance events occurring is low in this region, their compound effects on the resilience mechanism of serotinous lodgepole pine forests is ecological relevant. The most evident impact is the increased regeneration rate after the combined effects of pre-fire beetle outbreaks and subsequent fire. As bark beetle outbreaks and wildfire occurrence are both predicted to increase with continued climate warming in North America (Kurz et al., 2008; Westerling et al. 2011), the maps and findings presented in this study could serve as baseline information for assessing landscape-scale effects of disturbance and recovery on forest ecosystems, and inform effective management options.

6 Acknowledgements

We are grateful to the U.S. Geological Survey, Climate and Land Use Mission Area Land Remote Sensing Program provided funding to support this research (grant number G12AC20085). This work is/was supported, at least in part, by the USDA National Institute of Food and Agriculture, McIntire Stennis Capacity project. Two anonymous reviewers and Benjamin Sleeter provided insightful comments on a previous draft of this manuscript and their comments helped greatly to improve the completeness and clarity of the manuscript. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Reference

- Baldocchi, D. (2008) Breathing of the terrestrial biosphere: lessons learned from a global network of carbon dioxide flux measurement systems. *Australian Journal of Botany*, 56(1), 1-26.
- Barbour, M.G. & Billings, W.D. (Eds.). (2000) *North American terrestrial vegetation*. Cambridge University Press.
- Bentz, B.J., Régnière, J., Fettig, C.J., Hansen, E.M., Hayes, J.L., Hicke, J.A., Kelsey, R.G., Negron, J.F. & Seybold, S.J. (2010) Climate change and bark beetles of the western United States and Canada: direct and indirect effects. *BioScience*, 60(8), 602-613.
- Bigler, C., Kulakowski, D. & Veblen, T.T. (2005) Multiple disturbance interactions and drought influence fire severity in Rocky Mountain subalpine forests. *Ecology*, 86(11), 3018-3029.
- Björklund, N. & Lindgren, B.S. (2009) Diameter of lodgepole pine and mortality caused by the mountain pine beetle: factors that influence their relationship and applicability for susceptibility rating. *Canadian Journal of Forest Research*, 39(5), 908-916.

- 645
646 Bonan, G.B. (2008) Forests and climate change: forcings, feedbacks, and the climate
647 benefits of forests. *Science*, 320(5882), 1444-1449.
648
- 649 Bond, M.L., Lee, D.E., Bradley, C.M. & Hanson, C.T. (2009) Influence of pre-fire tree
650 mortality on fire severity in conifer forests of the San Bernardino Mountains,
651 California. *Open Forest Science Journal*, 2, 41-47.
652
- 653 Buma, B. & Wessman, C.A. (2011) Disturbance interactions can impact resilience
654 mechanisms of forests. *Ecosphere*, 2(5), art64.
655
- 656 Burns, R.M. & Honkala, B.H. editors (1990). *Silvics of north America. Volume 1.*
657 *Agriculture Handbook 654.* United States Department of Agriculture Forest Service,
658 Washington, D.C., USA.
659
- 660 Calkin, D.E., Gebert, K.M., Jones, J.G. & Neilson, R.P. (2005) Forest Service large fire
661 area burned and suppression expenditure trends, 1970–2002. *Journal of Forestry*, 103(4),
662 179-183.
663
- 664 Creeden, E.P., Hicke, J.A., & Buotte, P.C. (2014) Climate, weather, and recent mountain
665 pine beetle outbreaks in the western United States. *Forest Ecology and Management*, 312,
666 239-251.
667
- 668 Chambers, J.C. & MacMahon, J.A. (1994) A day in the life of a seed: movements and
669 fates of seeds and their implications for natural and managed systems. *Annual Review of*
670 *Ecology and Systematics*, 263-292.
671
- 672 Collins, B.M., Kelly, M., van Wagtenonk, J.W. & Stephens, S.L. (2007) Spatial patterns
673 of large natural fires in Sierra Nevada wilderness areas. *Landscape Ecology*, 22(4), 545-
674 557.
675

- Eidenshink, J., Schwind, B., Brewer, K., Zhu, Z.L., Quayle, B. & Howard, S. (2007) A project for monitoring trends in burn severity. *Fire Ecology*, 3, 3–20.
- Franklin, J.F., Spies, T.A., Pelt, R.V., Carey, A.B., Thornburgh, D.A., Berg, D.R., Lindenmayer, D.B., Harmon, M.E., Keeton, W.S., Shaw, D.C., Bible, K. & Chen, J.Q. (2002) Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. *Forest Ecology and Management*, 155(1), 399-423.
- Geiszler, D.R., Gara, R.I. & Littke, W.R. (1984) Bark beetle infestations of Lodgepole pine following a fire in South Central Oregon. *Zeitschrift für angewandte Entomologie*, 98(1-5), 389-394.
- Gibbs, H.K., Brown, S., Niles, J.O. & Foley, J.A. (2007) Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environmental Research Letters*, 2(4), 045023.
- Gibson, K.E., Skov, K., Kegley, S., Jorgensen, C., Smith, S. & Witcosky, J. (2008) Mountain pine beetle impacts in high-elevation five-needle pines: current trends and challenges. Washington, DC, USA: US Department of Agriculture, Forest Service, Forest Health Protection.
- Gong, P., Wang, J., Yu, L., Zhao, Y., Zhao, Y., Liang, L., ... & Chen, J. (2013). Finer resolution observation and monitoring of global land cover: first mapping results with Landsat TM and ETM+ data. *International Journal of Remote Sensing*, 34(7), 2607-2654.
- Gough, C.M., Vogel, C.S., Harrold, K.H., George, K. & Curtis, P.S. (2007) The legacy of harvest and fire on ecosystem carbon storage in a north temperate forest. *Global Change Biology*, 13(9), 1935-1949.

Groffman, P.M., Baron, J.S., Blett, T., Gold, A.J., Goodman, I., Gunderson, L. H.,
 Levinson, B.M., Palmer, M.A., Paerl, H.W., Peterson, G.D., Poff, N.L., Rejeski, D.W.,
 Reynolds, J.F., Turner, M.G., Weathers, K.C. & Wiens, J. (2006) Ecological thresholds:
 the key to successful environmental management or an important concept with no
 practical application?. *Ecosystems*, 9(1), 1-13.

Harvey, B.J., Donato, D.C., Romme, W.H. & Turner, M.G. (2013) Influence of recent
 bark beetle outbreak on fire severity and postfire tree regeneration in montane Douglas-
 fir forests. *Ecology*, 94(11), 2475-2486.

Harvey, B.J., Donato, D.C. & Turner, M.G. (2014) Recent mountain pine beetle
 outbreaks, wildfire severity, and postfire tree regeneration in the US Northern
 Rockies. *Proceedings of the National Academy of Sciences*, 111(42), 15120-15125.

Hicke, J.A., Meddens, A.J., Allen, C.D. & Kolden, C.A. (2013) Carbon stocks of trees
 killed by bark beetles and wildfire in the western United States. *Environmental Research*
Letters, 8(3), 035032.

Hicke, J.A., Johnson, M.C., Hayes, J.L. & Preisler, H.K. (2012) Effects of bark beetle-
 caused tree mortality on wildfire. *Forest Ecology and Management*, 271, 81-90.

Hopkins, A.D. (1909) Practical information on the scolytid beetles of North American
 forests. USDA Bureau of Entomology Bulletin 83, U.S. Government Printing Office,
 Washington, D.C., USA.

Johnstone, J.F., Hollingsworth T.N., Chapin F.S. & Mack, M.C. (2010) Changes in fire
 regime break the legacy lock on successional trajectories in Alaskan boreal forest. *Global*
Change Biology, 16:1281–1295.

Justice, C.O., Giglio, L., Korontzi, S., Owens, J., Morisette, J. T., Roy, D., Descloitres, J.,
Alleaume, S., Petitcolin, F. & Kaufman, Y. (2002) The MODIS fire products. *Remote
Sensing of Environment*, 83(1), 244-262.

Kautz, M. (2014) On correcting the time-lag bias in aerial-surveyed bark beetle
infestation data. *Forest Ecology and Management*, 326, 157-162.

Kennedy, R. E., Yang, Z. & Cohen, W. B. (2010) Detecting trends in forest disturbance
and recovery using yearly Landsat time series: 1. LandTrendr—Temporal segmentation
algorithms. *Remote Sensing of Environment*, 114(12), 2897-2910.

Key, C.H. & Benson, N.C. (2005) Landscape assessment. FIREMON: Fire effects
monitoring and inventory system. Gen. Tech. Rep. RMRS-GTR-164-CD, Fort Collins,
CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Kurz, W.A., Dymond, C.C., Stinson, G., Rampley, G.J., Neilson, E.T., Carroll, A. L.,
Ebaba, T. & Safranyik, L. (2008) Mountain pine beetle and forest carbon feedback to
climate change. *Nature*, 452(7190), 987-990.

Liang, L., Chen, Y.L., Hawbaker, T., Zhu, Z.L., & Gong, P. (2014a) Mapping mountain
pine beetle infestation through growth trend analysis of time-series Landsat data. *Remote
Sensing*, 6(6), 5696-5716.

Liang, L., Hawbaker, T., Chen, Y.L., Zhu, Z.L., & Gong, P. (2014b) Characterizing
recent and projecting future potential patterns of mountain pine beetle outbreaks in the
Southern Rocky Mountains. *Applied Geography*. DOI: 10.1016/j.apgeog.2014.09.012

Man. G. (2010) Major forest insect and disease conditions in the United States: 2009
Update. US Department of Agriculture, Forest Service, Forest Health Protection,
Washington.

- Meddens, A.J.H., & Hicke, J.A. (2014) Spatial and temporal patterns of Landsat-based detection of tree mortality caused by a mountain pine beetle outbreak in Colorado, USA *Forest Ecology and Management*, 322, 78-88.
- Meddens, A.J.H., Hicke, J.A., & Ferguson, C.A. (2012) Spatiotemporal patterns of observed bark beetle-caused tree mortality in British Columbia and the western United States. *Ecological Applications*, 22, 1876-1891.
- Meddens, A.J.H., Hicke, J.A., Vierling, L.A., & Hudak, A.T. (2013) Evaluating methods to detect bark beetle-caused tree mortality using single-date and multi-date Landsat imagery. *Remote Sensing of Environment*, 132, 49-58.
- Meigs, G.W., Kennedy, R.E. & Cohen, W.B. (2011) A Landsat time series approach to characterize bark beetle and defoliator impacts on tree mortality and surface fuels in conifer forests. *Remote Sensing of Environment*, 115(12), 3707-3718.
- Meigs, G. W., Kennedy, R. E., Gray, A. N., & Gregory, M. J. (2015) Spatiotemporal dynamics of recent mountain pine beetle and western spruce budworm outbreaks across the Pacific Northwest Region, USA. *Forest Ecology and Management*, 339, 71-86.
- National c2001 Assessment. Available online:
http://www.landfire.gov/dp_quality_assessment.php (accessed on 12 June 2014).
- Nelson, K.J., Connot, J., Peterson, B. & Martin, C. (2013) The LANDFIRE Refresh strategy: updating the national dataset. *Fire Ecology*, 9(2), 80-101.
- Pflugmacher, D., Cohen, W.B., Kennedy, R.E. & Yang, Z. (2013) Using Landsat-derived disturbance and recovery history and lidar to map forest biomass dynamics. *Remote Sensing of Environment*, 151, 124-137.

796 Powell, E.N., Townsend, P.A. & Raffa, K.F. (2012) Wildfire provides refuge from local
 797 extinction but is an unlikely driver of outbreaks by mountain pine beetle. *Ecological*
 798 *Monographs*, 82(1), 69-84.
 799

800 Romme, W.H., Clement, J., Hicke, J., Kulakowski, D., MacDonald, L.H., Schoennagel,
 801 T.L. & Veblen, T.T. (2006) Recent forest insect outbreaks and fire risk in Colorado
 802 forests: a brief synthesis of relevant research. Fort Collins, Colorado, USA: Colorado
 803 Forest Restoration Institute.
 804

805 Running, S.W. (2008) Ecosystem disturbance, carbon, and climate. *Science*, 321(5889),
 806 652-653.
 807

808 Sambaraju, K.R., Carroll, A.L., Zhu, J., Stahl, K., Moore, R.D. & Aukema, B.H. (2012)
 809 Climate change could alter the distribution of mountain pine beetle outbreaks in western
 810 Canada. *Ecography*, 35(3), 211-223.
 811

812 Schoennagel, T., Turner, M.G., Kashian, D.M. & Fall, A. (2006) Influence of fire
 813 regimes on lodgepole pine stand age and density across the Yellowstone National Park
 814 (USA) landscape. *Landscape ecology*, 21(8), 1281-1296.
 815

816 Schoennagel, T., Veblen, T.T., Negron, J.F. & Smith, J.M. (2012) Effects of mountain
 817 pine beetle on fuels and expected fire behavior in lodgepole pine forests, Colorado,
 818 USA. *PloS one*, 7(1), e30002.
 819

820 Simard, M., Romme, W.H., Griffin, J.M. & Turner, M.G. (2011) Do mountain pine
 821 beetle outbreaks change the probability of active crown fire in lodgepole pine
 822 forests?. *Ecological Monographs*, 81(1), 3-24.
 823

824 Simon, M., Plummer, S., Fierens, F., Hoelzemann, J.J. & Arino, O. (2004) Burnt area
 825 detection at global scale using ATSR-2: The GLOBSCAR products and their
 826 qualification. *Journal of Geophysical Research: Atmospheres* (1984–2012), 109(D14).

827

828 Skole, D. & Tucker, C. (1993) Tropical deforestation and habitat fragmentation in the
 829 Amazon. *Science*, 260(5116), 1905-1910.

830

831 Smith, J.M., Paritsis, J., Veblen, T.T., & Chapman, T.B. (2015) Permanent forest plots
 832 show accelerating tree mortality in subalpine forests of the Colorado Front Range from
 833 1982 to 2013. *Forest Ecology and Management*, 341, 8-17.

834

835 Stephens, S.L. (2005) Forest fire causes and extent on United States Forest Service
 836 lands. *International Journal of Wildland Fire*, 14(3), 213-222.

837

838 Turner, M.G., Romme, W.H., Gardner, R.H. & Hargrove, W.W. (1997) Effects of fire
 839 size and pattern on early succession in Yellowstone National Park. *Ecological*
 840 *Monographs*, 67(4), 411-433.

841

842 Turner, M.G. (2010) Disturbance and landscape dynamics in a changing
 843 world. *Ecology*, 91(10), 2833-2849.

844

845 U.S. Environmental Protection Agency, 2013, Level III ecoregions of the continental
 846 United States: Corvallis, Oregon, U.S. EPA – National Health and Environmental Effects
 847 Research Laboratory, map scale 1:7,500,000,
 848 http://www.epa.gov/wed/pages/ecoregions/level_iii_iv.htm.

849

850 Veblen, T.T., Hadley, K.S., Nel, E.M., Kitzberger, T., Reid, M. & Villalba, R. (1994)
 851 Disturbance regime and disturbance interactions in a Rocky Mountain subalpine
 852 forest. *Journal of Ecology*, 125-135.

853

854 Vose, J.M., Peterson, D.L. & Patel-Weynand, T. (2012) Effects of climatic variability
 855 and change on forest ecosystems: a comprehensive science synthesis for the US forest
 856 sector. Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest
 857 Research Station.

858

859 Westerling, A.L., Turner, M.G., Smithwick, E.A., Romme, W.H. & Ryan, M.G. (2011)
860 Continued warming could transform Greater Yellowstone fire regimes by mid-21st
861 century. *Proceedings of the National Academy of Sciences*, 108(32), 13165-13170.

862

863 White, P.S. & Pickett, S.T. (1985) Natural disturbance and patch dynamics: an
864 introduction. In *The ecology of natural disturbance and patch dynamics*. Edited by S.T.A.
865 Pickett and P.S.White. Academic Press, Orlando. Pp.3-16.

866

867 Wulder, M.A., Dymond, C.C., White, J.C., Leckie, D.G. & Carroll, A.L. (2006)
868 Surveying mountain pine beetle damage of forests: A review of remote sensing
869 opportunities. *Forest Ecology and Management*, 221(1), 27-41.

870

871 **List of Figure Captions**

872 Figure 1. The coverage of Southern Rocky Mountain Ecoregion and the frame of 14
873 intersected Landsat scenes with trimmed edges.

874

875 Figure 2. Classification workflow.

876

877 Figure 3. Sampling workflow.

878

879 Figure 4. Boxplots displaying the Normalized Burn Ratio change magnitude of cases and
880 controls.

881

882 Figure 5. Disturbance and recovery pathways in South Rocky Mountain Ecoregion.