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Relative importance of climate and mountain pine beetle outbreaks on the occurrence of large wildfires in the western USA

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Abstract. Extensive outbreaks of bark beetles have killed trees across millions of hectares of forests and woodlands in western North America. These outbreaks have led to spirited scientific, public, and policy debates about consequential increases in fire risk, especially in the wildland–urban interface (WUI), where homes and communities are at particular risk from wildfires. At the same time, large wildfires have become more frequent across this region. Widespread expectations that outbreaks increase extent, severity, and/or frequency of wildfires are based partly on visible and dramatic changes in foliar moisture content and other fuel properties following outbreaks, as well as associated modeling projections. A competing explanation is that increasing wildfires are driven primarily by climatic extremes, which are becoming more common with climate change. However, the relative importance of bark beetle outbreaks vs. climate on fire occurrence has not been empirically examined across very large areas and remains poorly understood. The most extensive outbreaks of tree-killing insects across the western United States have been of mountain pine beetle (MPB; *Dendroctonus ponderosae*), which have killed trees over >650,000 km^2, mostly in forests dominated by lodgepole pine (*Pinus contorta*). We show that outbreaks of MPB in lodgepole pine forests of the western United States have been less important than climatic variability for the occurrence of large fires over the past 29 years. In lodgepole pine forests in general, as well as those in the WUI, occurrence of large fires was determined primarily by current and antecedent high temperatures and low precipitation but was unaffected by preceding outbreaks. Trends of increasing co-occurrence of wildfires and outbreaks are due to a common climatic driver rather than interactions between these disturbances. Reducing wildfire risk hinges on addressing the underlying climatic drivers rather than treating beetle-affected forests.

Key words: climate change; Dendroctonus ponderosae; *disturbance interactions; drought; ecological disturbance; linked disturbances; lodgepole pine;* Pinus contorta; *subalpine forests; Wildland–urban interface.*

Introduction

Wildfires and bark beetle outbreaks have increased in frequency, severity, and extent across forests of the western United States over the past 30 years (Fig. 1a, c; Raffa et al. 2008, Bentz et al. 2010, Dennison et al. 2014, Baker 2015). The combination of these disturbances has had major effects on public safety, tourism, carbon storage, carbon sequestration, water quality, and other natural and economic resources (Negron et al. 2008, Bentz et al. 2009, Price et al. 2010, Powell et al. 2012, Saab et al. 2014, Ghimire et al. 2015). The most extensive outbreaks of tree-killing insects across the western United States have been of mountain pine beetle (MPB; *Dendroctonus ponderosae*), which have killed trees over >650,000 km^2, mostly in forests dominated by lodgepole pine (*Pinus contorta;* USFS 2014). The combination of these extensive outbreaks and prolonged drought (Fig. 1d) has caused widespread concern about increased wildfire risk in remote forests, as well as in the wildland–urban interface (WUI) where homes and communities are at high risk from wildfires (Fig. 1b). While the link between climatic variability and wildfires has been well established (e.g., Dennison et al. 2014), the effects of MPB outbreaks on wildfires are complex, sometimes counterintuitive, and not fully explicated (e.g., Hicke et al. 2012, Jolly et al. 2012, Moran and Cochrane 2012, Kulakowski and Veblen 2015).

The perception that bark beetle outbreaks increase wildfire risk stems from the dramatic and visible changes in foliage of beetle-killed trees. When forest stand and climatic conditions are suitable, MPB initiate a pheromone-mediated mass attack to overwhelm the defenses of healthy, living trees (Raffa and Berryman 1983). Immediately following tree mortality (approximately <1 yr), foliar moisture content decreases and leaf chemistry changes, turning foliage red (approximately 1–3 yr), at the same time or shortly thereafter foliage, twigs, branches (approximately 4–30 yr), and the tree itself fall (approximately >30 yr; Appendix S1: Table S1; Klutsch et al. 2009, Jolly et al. 2012, Nelson et al. 2014). The exact timing of this progression is variable and contingent on tree and site condition, including weather and

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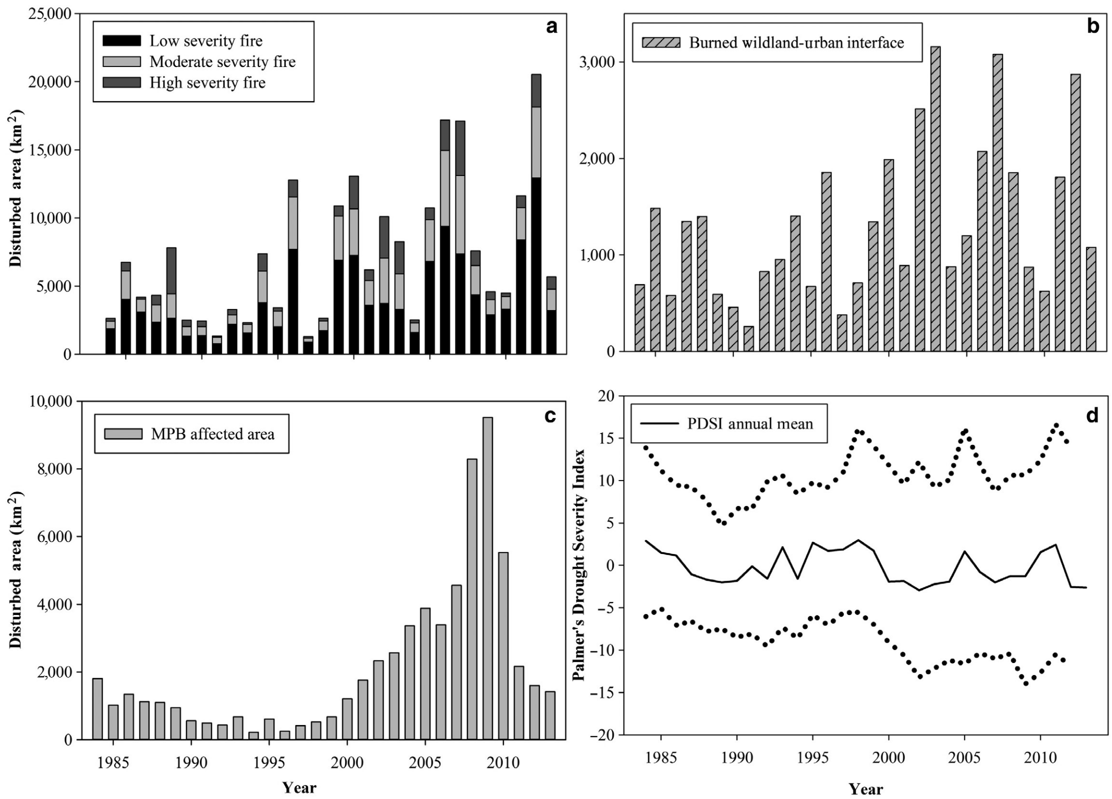

FIG. 1. Total annual area burned by (a) low, medium and high severity wildfires; (b) annual area burned in the wildland–urban interface; (c) annual area affected by new mountain pine beetle outbreaks; (d) mean (solid line), maximum and minimum PDSI values in the western USA from 1984 to 2013.

biophysical setting. Research at scales from individual tree needles to forest stands has consistently described major changes in the characteristic and arrangement of forest fuels (potentially flammable materials) following bark beetle outbreaks (Page and Jenkins 2007*a, b*, Klutsch et al. 2009, Simard et al. 2011, Collins et al. 2012, Jenkins et al. 2012, Schoennagel et al. 2012, Page et al. 2014), which can affect the flammability of beetle-killed trees (Jolly et al. 2012, Page et al. 2012). However, the question of how reduced foliar moisture and altered needle chemistry actually affect fire regimes has been controversial and remains not fully understood.

Although much has been learned about variation in crown and surface fuels following MPB outbreaks, understanding how post-outbreak fuel conditions affect wildfires over large areas has proven elusive and continues to be an important topic of research. Some modeling studies have suggested that the potential for active crown fire may increase following outbreaks, especially during the so-called red phase, when desiccated needles remain on beetle-killed trees (Hicke et al. 2012, Hoffman et al. 2013). But differences in how foliar moisture content of recently killed trees vs. other fine fuels respond to changing environmental conditions can affect the accuracy of fire behavior models (Page et al. 2013, 2014). In contrast, empirical studies have generally found that wildfire hazard, occurrence, and severity do not substantially increase following MPB outbreaks, though results have not been consistent (e.g., reviewed in Hicke et al. 2012, Kulakowski and Veblen 2015). Ongoing outbreaks of MPB in Colorado, USA, had no detectable effect on the extent or severity of a fire during a drought in 2002, even though recently killed trees were in the critical red phase (Kulakowski and Veblen 2007). Similarly, during the initial years of MPB outbreak in Yellowstone, foliar moisture content and canopy bulk density were found to decrease simultaneously, effectively reducing hazard of active crown fires during the red phase (Simard et al. 2011). Stands affected by MPB 13–16 years prior to the 1988 Yellowstone fires were marginally more likely to burn, but stands that were affected by outbreak from five to eight years prior to the fires were no more likely to burn in comparison to unaffected stands (Lynch et al. 2006). The effect of MPB on fire severity has been shown to be contingent on the stage of the outbreak and on burning conditions, such that outbreaks can slightly affect fire severity under moderate burning conditions but have little effect on fire severity under extreme burning conditions (Harvey et al. 2014*a*, *b*). Indeed, during several extremely dry years across the western United States, area burned was unaffected by preceding MPB outbreaks (Hart et al. 2015*a*). There is general agreement that following the red phase, the risk of active high-severity crown fires decreases due to reduced canopy bulk density (Romme et al. 2006, Jenkins et al. 2008, 2012, 2014, Kaufmann et al. 2008, Hicke et al. 2012, Black et al. 2013), but this assertion has not been thoroughly tested using empirical data at landscape and regional scales.

While the link between wildfires and MPB outbreaks across broad spatial scales are just starting to be understood (e.g., Hart et al. 2015*a*, Meigs et al. 2015, Meigs et al. 2016), a largely separate body of literature has made clear that large wildfires in coniferous forests of the Rocky Mountains across western North America are strongly influence by climatic variability (e.g., Dennison et al. 2014). The ecologically most important fires in lodgepole pine forests are high severity and the species is well adapted to regenerate following such fires. While some low-severity fires have been documented in lodgepole pine forests, they are generally thought not to have major effects on stand structure (Barrett et al. 1991, Hessburg et al. 2007). During extreme fire weather that facilitates wildfire activity in lodgepole pine and other subalpine forests, fuels are likely dry enough to promote extensive burning, regardless of changes to fuels due to bark beetle infestation. Indeed, wildfires in lodgepole pine forests in Colorado have been related to drought but not to MPB outbreaks, even in beetle-affected forests (Kulakowski and Jarvis 2011), but this relationship has not yet been tested over very large areas. Fine scale variation in topography and daily fluctuations in wind speed, humidity, wind direction, and temperature also affect fire behavior and fire spread, but these elements are less important across broader (e.g., sub-continental) spatial scales.

Increasing climatically driven disturbances such as wildfires and bark beetle outbreaks are important not only ecologically and biogeochemically, but also socioeconomically. Much scientific focus and government resources have been devoted to mitigating fire risk to structures in and around the wildland–urban interface (WUI), where homes and communities are at particular risk from wildfires. For instance, during the time period from 2000 to 2005 wildfire suppression costs in the USA exceeded US\$1 billion per year (Calkin et al. 2005; USDA and OIG 2006), while from 2001 to 2006 the US Congress appropriated US\$2.7 billion for preventative fuel treatments (Western Governors' Association 2006). The structure of vegetation and the quantity and quality of fuels within the WUI, and especially in the immediate vicinity of homes, are important in determining the risk of fire to those homes (Cohen 2000, Cohen et al. 2001). Over the past decades, some forests in the WUI have been affected by the extensive MPB outbreak and associated increases in dead fuels have contributed to concerns about increased fire risk in the WUI. While progress has been made in understanding how to best protect homes and communities from wildfire, a central question that remains unanswered is how massive bark beetle outbreaks affect fire risk in the WUI, especially in the context of a warming climate.

Despite the extraordinary extent of recent MPB outbreaks, to date no study has examined the relative importance of outbreaks and climatic variability on fire occurrence across the western United States. We address the following questions for lodgepole pine forests across the western United States and within the WUI: (1) What is the relative importance of climatic variability vs. preceding occurrence of MPB outbreaks on the occurrence of large wildfires?, and (2) How has the importance of these two variables changes over the past decades as outbreaks have become larger and more severe?

Materials and Methods

Study area

The study area is composed of 11 states in the contiguous western United States (New Mexico, Colorado, Wyoming, Montana, Idaho, Washington, Oregon, Nevada, California, and Arizona; Fig. 2), constrained to lodgepole pine dominated forests.

Data

Annual burned area and perimeter (Fig. 2a; 1984–2013) were obtained from the Monitoring Trends in Burn Severity group (Eidenshink et al. 2007; information on pre-processing, specific information on index calculation, and derived dataset: *available online*), which use satellite imagery (Landsat TM/ETM+/OLI) to produce spatially explicit extents for all large fires (>400 ha).[4] These data are produced using the differenced Normalized Burn Ratio (dNBR), which uses pre- and post-fire spectral response to define full extents of the disturbance events. In addition to this automated process, and to ensure consistency and precision, manual digitation is performed at on-screen display scales between 1:24,000 and 1:50,000.

Annual MPB outbreak data are spatially represented polygons from the aerial detection surveys (ADS) conducted by the USDA Forest Service for the years 1981–2013 (Fig. 2b; USFS 2014). These data cover the western contiguous USA (USFS regions 1–6). USFS accuracy assessments of ADS and ground references for MPB infestation in lodgepole pine forests indicate increasing agreement as spatial scale increases (i.e., 70% agreement at a 30-m scale and 87% agreement at a 500-m scale), suggesting that ADS data are appropriate for assessing forest disturbances across broad areas (Johnson and Ross 2008, Meddens et al. 2012). Previous studies have successfully used ADS across broad spatial scales (state and sub-continental) to study bark beetle outbreaks (Raffa et al. 2008, Chapman et al. 2012, Hart et al. 2015*a*, *b*). Even though ADS data have been used extensively to study the spatiotemporal development of outbreaks, our analysis was designed to minimize any potential effects of variability in the temporal acquisition of the ADS data

[4] http://mtbs.gov/

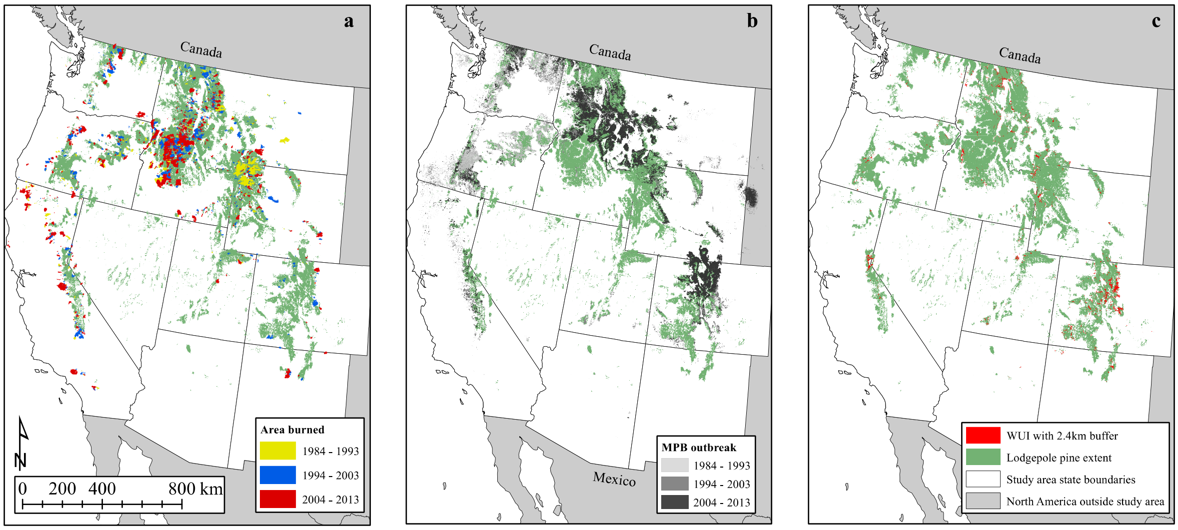

FIG. 2. (a) Area burned, (b) new mountain pine beetle outbreak, and (c) the wildland–urban interface across the study area from 1984 to 2013.

(e.g., less complete mapping during early stages of the outbreak). To do so, we conducted our analysis on the time period 1984–2013 but also examined how outbreaks affect fire occurrence separately for three shorter time periods: 1984–1993, 1994–2003, and 2004–2013.

Our analysis considered only large (>400 ha) fires in lodgepole pine dominated forests. Locations of lodgepole pine forests were determined based on the intersection of three separate datasets to produce a less biased and more conservative cover-type classification (Table S2-1; Rollins 2009), USGS 2011, Mathys et al. 2014). Physiographic variables of slope, aspect, and elevation were derived from a 30-m Digital Elevation Model (Gesch et al. 2002).

Climate data were derived from PRISM (Parameter–Elevation Regressions on Independent Slope Model; PRISM Climate Group 2014), which includes spatial monthly temperature and precipitation grids for the contiguous USA covering 1984–2013 (Daly et al. 2008). Three drought indices from PRISM were used to represent short and long-term drought variability (John Abatzoglou, *unpublished data*). The self-calibrated Palmer's Drought Severity Index (scPDSI) and the Palmer's Drought Severity Index (PDSI) represent the accumulation or deficit of surface water over a nine-month period and were used to describe long-term drought. The scPDSI is a locally calibrated version of the PDSI that makes values more comparable across space (Wells et al. 2004). Additionally, the Palmer's Z index was used to measure short-term drought variability, depicting monthly drought orthogonal to previous drought. Each drought index is represented as a monthly grid with values ranging from −8 (severe drought) to 8 (anomalously wet). All climate variables were standardized and expressed as departures from the long-term mean.

The wildland–urban interface (WUI; Appendix S1: Table S2) is defined by the Federal Register as the area where houses meet or intermingle with undeveloped wildland vegetation (USDA and USDI 2001). A WUI GIS dataset (Fig. 2c; Radeloff et al. 2005) was used to assess the relative importance of MPB vs. climate within the WUI. Each polygon within this dataset represents a 2010 US census housing block, which is defined as an interface, intermixed, or non-WUI category. Intermix WUI census blocks exceed 6.17 housing units km^2 and have >50% wildland vegetation. Interface WUI census blocks exceed 6.17 housing units km^2 and have <50% wildland vegetation but are located within 2.4 km of an area larger than 5 km^2 containing >75% wildland vegetation. We combined the interface and intermixed WUI zones to represent the total potential area of the WUI within the western USA. Despite the wildfire risk posed to WUI structures from ignitions located within the WUI, not all WUI wildfires are ignited within the WUI itself. Brand production is likely to vary between forests affected by bark beetle outbreak and forests unaffected by outbreaks. As reliable models of these differences are not available, we added a standard 2.4 km buffer (an estimate of the distance a firebrand could be transported ahead of a wildfire front and ignite a new wildfire (California Fire Alliance 2001) to each WUI census block.

Spatial resolution

This study evaluates the occurrence of large wildfires as a function of antecedent climate, bark beetle disturbance, and static physiographic variables across the western United States at the resolution of the individual wildfire event. We aggregated MPB data by categorizing each burned area as having been affected by MPB if any portion of that burned area (>0%) was affected. This conservative approach effectively biases the results to overestimate the influence of MPB on fire occurrence. We aggregated climate data by averaging all PRISM pixels within a given burned area. This aggregation assumes antecedent climate did not vary at the scale of individual fires (4–2,286 km^2).

Spatial analysis

All spatial analyses were performed in ArcGIS 10.2 (ESRI 2014) and IDRISI Selva (Eastman 2012). Polygons affected by MPB were spatially joined to MTBS wildfire polygons. Next, the initial presence of MPB and the peak year of presence of MPB were identified within each wildfire. Annual ADS data from 1984 to 2013 data were further refined into two datasets: one representing first year of MPB presence and the other the peak of MPB presence within each wildfire area. For each of these resulting polygons, we also calculated a metric of the ratio of area affected by MPB to the total area of the burned area during both the years of initial onset of outbreaks and the years of peak occurrence of MPB. Thus, we produce four unique explanatory variables to quantify the timing and amount of MPB presence with a polygon prior to wildfire event. We recast the pre-fire MPB outbreaks (based on both years of initiation and peak years) based on the lag between the outbreak and wildfire/non-wildfire event: 0–3 yr (red stage), 4–6 yr (young gray stage), 7–14 yr (old gray stage), and 15–30 yr (old MPB stage; Appendix S1: Table S1).

During spatial re-projection, monthly climate variables were converted to points based on the centroid of each grid cell to preserve the spatial integrity and original pixel value of the raster cell (Christman and Rogan 2012). Points were projected into the Albers equal-area contiguous United States system to preserve the areal integrity of the wildfire polygons and transformed back to raster grid cells. Each monthly climate variable time series was temporally aggregated from a monthly to seasonal and yearly time stacks based on the mean, minimum, maximum, and standard deviation. Monthly, seasonal, and yearly climate data were extracted for each wildfire polygon and subsequently appended to the attribute data for those wildfire polygons. Antecedent climate variables were temporally lagged (e.g., months, seasons, years) from time of wildfire/non-wildfire event in one time-step increments for up to four time steps (e.g., 4 yr, 4 seasons, 4 months) prior to event. This extraction process was also used to acquire all other values for independent variables within each wildfire polygon (Appendix S1: Table S2).

The joined spatiotemporal independent (e.g., climate, MPB presence, elevation, etc.) and dependent (wildfire) variables produced an array of presence (fire occurrence) only data. Non-fire information for each fire location was produced using a randomly selected year prior to the fire. All independent variables were joined to that non-fire event based on the methods described above. Paired sampling points are often used to compare present/absence data. However, using a paired sampling design for this analysis leaves open the possibility that non-fire points were unburned because they did not have the potential to burn during the study period due to fires occurring prior to 1984, timber harvest, or other activities that removed all flammable materials immediately prior to the analysis period. By selecting non-fire events at the same locations as fire events, we assure that, by definition, the forest vegetation had the potential to burn during the analysis period but did not because conditions related to climate or outbreaks were not suitable. Even though selecting non-fire data in this manner is robust, we complemented this analysis with a conventional paired sampling design by generating a restricted random paired point distribution within the subalpine zone (2,750–3,350 m above sea level) across the western USA (non-fire points) and within the fire burned area (fire points) in lodgepole pine stands. An equal number of points were located inside and outside of fire perimeters for each year in the series.

Statistical analysis

Classification trees and random forest were used to test the relationship between wildfire occurrences and climatic, topographic, and beetle-related variables. Classification trees are ideal for complex ecological data analysis due to their ability to robustly deal with nonlinear relationships and high-order interaction and to provide easily interpretable results (De'ath and Fabricius 2000). Random forests are an ensemble of single classification trees describing the non-parametric relationship of environmental and disturbance conditions prior to wildfire events. The random forest algorithm significantly improves the predictive power, can handle collinear data (Cutler et al. 2007, Quach 2012), and reduces the overall variance of a single classification tree (Breiman 2001). To effectively reduce the model variance, the ensemble generates leaf predictors of a weighted nearest neighbor to appropriately combine trees into the forest scheme (Lin and Jeon 2006). This procedure produces a list ranking of the most explanatory variables based on the mean Gini Index (Breiman 2001), where higher index values indicate greater importance. To minimize out-of-bag error and account for overfitting of the random forest model, we tuned our model by optimizing the number of node splits (*mtry*), which effectively minimized the root-mean-square error (RMSE). Data for each run was randomly sub-divided into calibration (60%) and validation (40%) samples. Five runs of a cross-validated random forest model (*k* folds = 10) containing 2,000 trees were independently grown, providing the best-fit rank list of variable importance. Predictive power and accuracy of the model was calculated by evaluating multiple assessments (multi-dimensional scaling plot, receiver operating characteristic, out-of-bag error) to provide a more holistic evaluation of model performance. Mean decrease accuracy was evaluated to describe importance of predictor variables across decades. Independent variables in random forest models of wildfire occurrence included antecedent climate, MPB disturbance, and topographic setting.

Outputs of random forests are limited by overall interpretability of complex relationships among many variables, which can be overcome by combining these outputs with classic classification trees (Breiman 2001). Therefore, we used classification trees to provide visual interpretation of the outputs from the random forest model. To ensure that the random forest contained no bias towards the climate variables, a series of classification trees were created for each decade based on the four most important climatic and the four most important MPB variables. We hypothesized that if MPB infestation increased over our time series, and is important for the occurrence of fire, then in the more recent years MPB should be more important for fire occurrence when compared to the earlier years of the record. The classification tree was pruned to reduce over-fitting and error was minimized by removing splits exceeding the complexity parameter. All statistical analysis was performed in the R statistical software (R Development Core Team 2014). Random forests and classification trees were computed using the package caret, randomForest, and rpart, respectively (Liaw and Wiener 2002).

Chi-square was used to test the association between occurrence of fires and preceding outbreaks based on the complementary paired sampling design. Points were considered affected by MPB if they had been affected at any time prior to the fire (1984–2013).

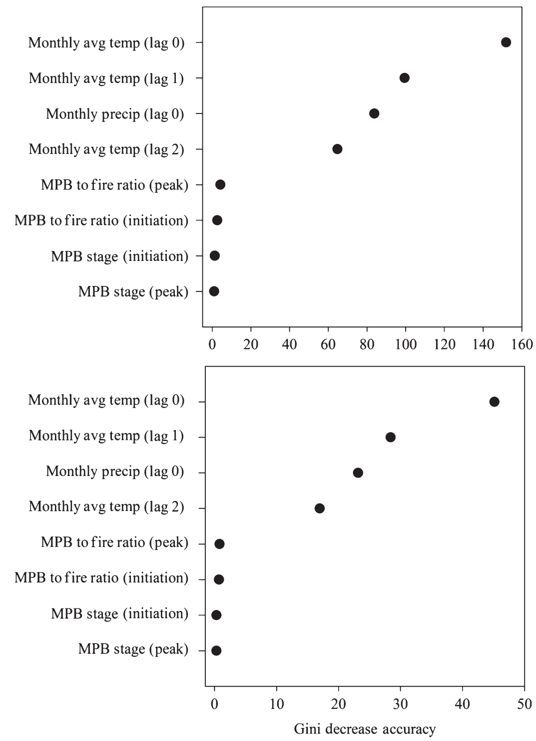

Fig. 3. Most important variables explaining wildfire occurrence (a) across the western USA and (b) within the WUI based on a random forest model from 1984 to 2013.

Results

Wildfire occurrence

Based on random forest classification large wildfire occurrence in lodgepole pine forests across the western USA from 1984 to 2013 was determined primarily by antecedent monthly and seasonal temperature and precipitation (Fig. 3a) and was not affected by preceding outbreaks of mountain pine beetle (Fig. 3a). Temperature and precipitation were better predictors than drought indices and short-term (0–2 months) variability was more important than long-term (>2 seasons). Topographic variables were not important to wildfire occurrence. The least important predictors of wildfire occurrence were the proportion of the area affected by MPB, the time since MPB initiation, and the peak MPB stage at time of wildfire. Overall out-of-bag error of the random forest model of wildfire occurrence was 7% and the area under the curve (AUC) was 98% (Appendix S1: Fig. S1a). MDS indicates separability between wildfire and non-wildfire occurrences (Appendix S1: Fig. S1b). Likewise, based on the complementary paired sampling, there was no significant relationship between occurrence of wildfires and preceding outbreaks (Chi-square: red stage = 0.552; young gray = 0.032; old gray = 1.016; $P > 0.3$).

Across all three decades (1984–1993, 1994–2003, 2004–2013), the occurrence of large wildfire was associated with short-term high temperature (≤1 month prior to the fire), and secondly, low precipitation during the month of the fire. Long-term variation in antecedent climate conditions (e.g., >1 season) was less important to the occurrence of large fires across the western USA. Topographic variables were not important to wildfire occurrence in any decade. The least important predictors of wildfire occurrence were the ratio of the area affected by MPB, the time since MPB initiation, and the peak MPB stage present at time of wildfire (Table 1). MDS plots for each decade exhibit high separation of wildfire vs. non-wildfire events (Fig. 5a–c), while maintaining low out-of-bag error (Appendix S1: Fig. S2a–c).

Classification tree analysis using the top four climate variables, as derived from the random forest model, and all four MPB variables indicate high temperatures explained the greatest variance of large fire occurrence across all decades (Fig. 6). Results from the 2004–2013 decade of peak-MPB-outbreak indicate that given high monthly temperatures and a low MPB-to-area-burned ratio there is a 56% probability that a fire will occur, while a high MPB-to-area-burned ratio resulted in a 4% probability that no fire would occur (Fig. 6c).

Wildfire occurrence within the WUI

Over the 29-year record, the occurrence of large wildfires within the WUI was primarily determined by

TABLE 1. Mean decrease accuracy results from the final random forest model.

	Mean decrease accuracy (%)					
	Fire occurrence			Fire within the WUI		
	1984–1993	1994–2003	2004–2013	1984–1993	1994–2003	2004–2013
Monthly average temperature (lag 0)	35.624	48.423	50.554	25.093	37.558	33.671
Monthly average temperature (lag 1)	29.138	35.881	36.112	24.901	25.091	25.759
Seasonal average temperature (lag 1)	14.239	26.499	30.705	7.128	25.939	25.949
Monthly precipitation (lag 0)	24.338	34.753	29.329	16.464	27.414	23.841
MPB to fire ratio (peak)	−4.491	4.519	12.227	2.237	1.915	13.653
MPB to fire ratio (initiation)	−2.291	3.789	12.926	0.074	1.671	12.666
MPB stage (peak)	−0.948	2.867	27.046	0.147	0.167	10.561
MPB stage (initiation)	−0.436	6.223	25.277	1.001	1.299	9.215

Note: Variables evaluated include the most important overall predictors of wildfire occurrence and all MPB variables for each decade.

climatic variables (monthly high temperature and monthly low precipitation; Fig. 3b) as determined by the random forest analysis. Variability in short-term precipitation and temperature (0–1 month) preceding a fire were more important than long-term predictors (>3 months). Topographic variation and MPB presence, regardless of stage of infestation, were not important to wildfire occurrence. MDS indicates separability among wildfire/non-wildfire occurrence (Appendix S1: Fig. S1c). Overall out-of-bag error of this random forest model was 6.6% and the AUC was 98% (Appendix S1: Fig. S1d).

Across the all three decades, the major determinates of large fire occurrence in the WUI were short-term variability of antecedent high temperature and low precipitation (<2 months; Fig. 4). Fire occurrence from 1984 to 1993 was determined by high temperature during and one month prior to the fire event (Fig. 4d). Between 1994 and 2003, high temperature and low precipitation during the month that the fire occurred and prior seasonal high

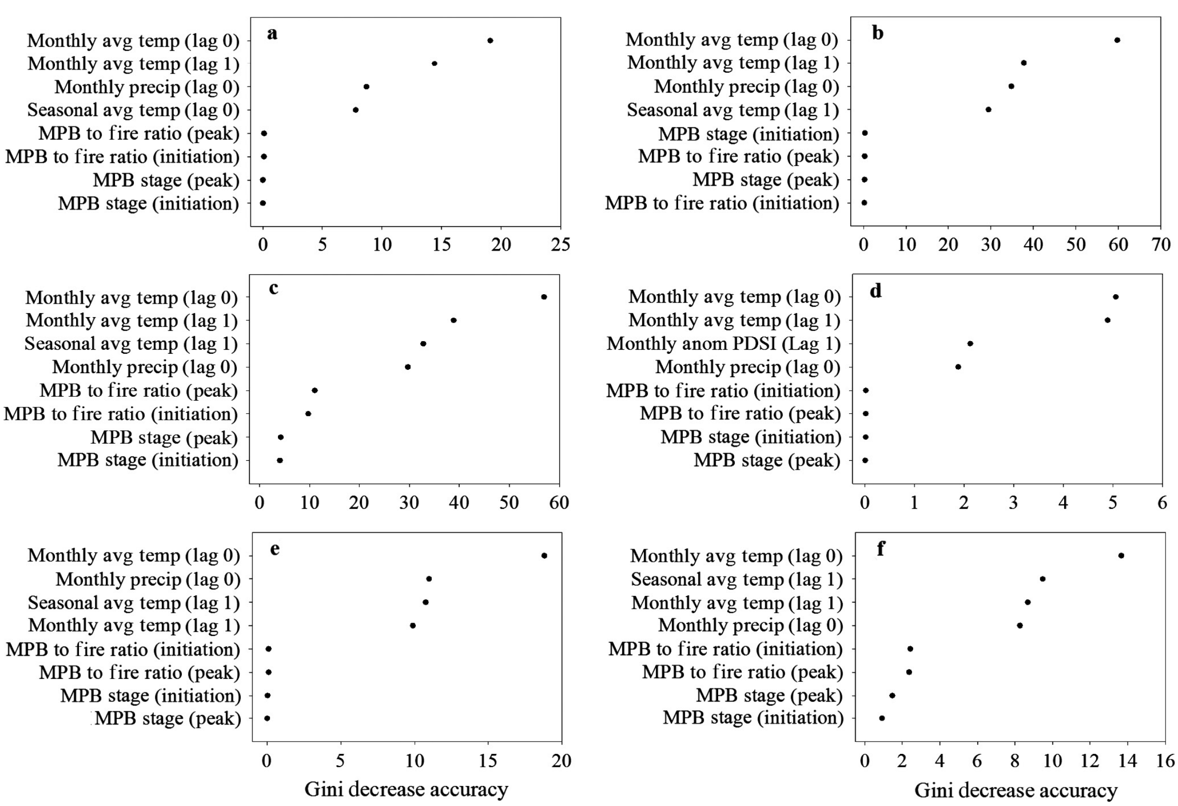

FIG. 4. Most important variables explaining wildfire occurrence across the western USA in (a) 1984–1993, (b) 1994–2003, and (c) 2004–2013, and within the WUI (d) 1984–1993, (e) 1994–2003, and (f) 2004–2013 based on a random forest model.

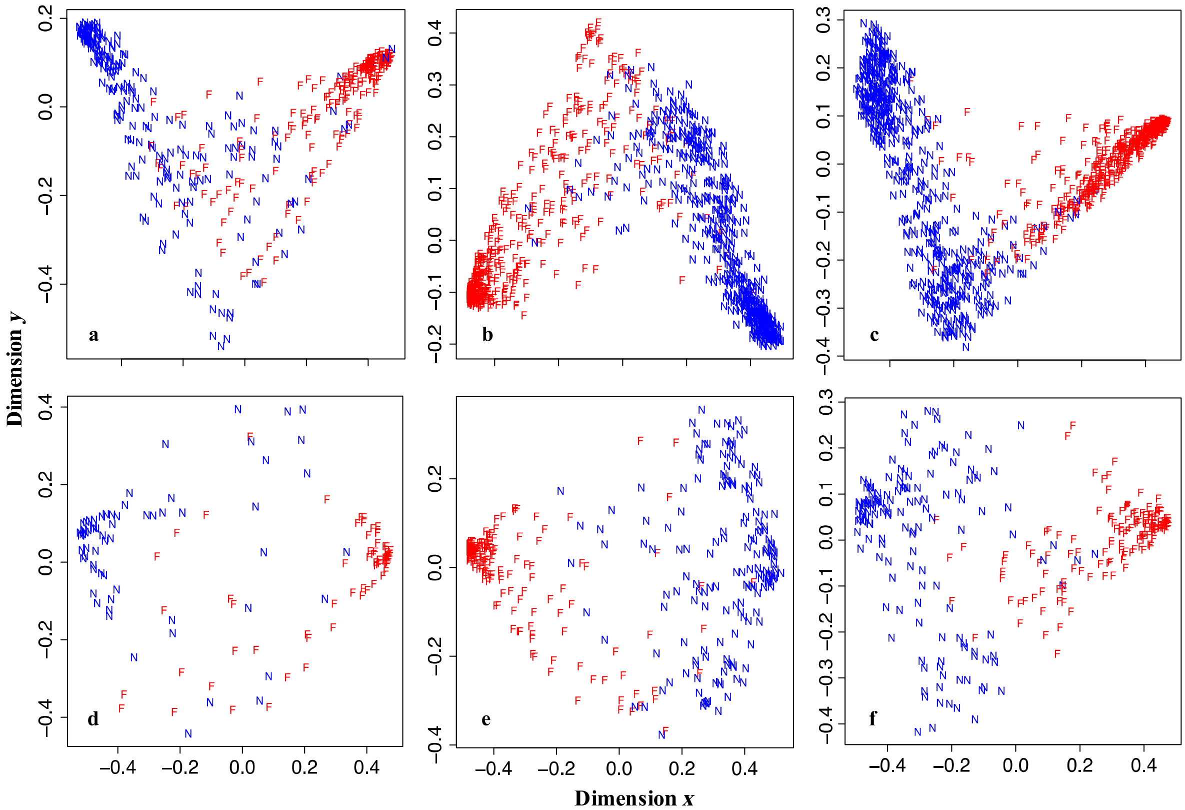

FIG. 5. A metric multi-dimensional scaling (MDS) representation for the proximity matrix separability as predicted by random forests of total wildfire occurrence across the western USA (a) 1984–1993, (b) 1994–2003, and (c) 2004–2013, and within the WUI (d) 1984–1993, (e) 1994–2003, and (f) 2004–2013. Classification separability as fire (F) and no fire (N) are depicted in the colors red and blue, respectively.

temperature (1 season) were the most important variables explaining fire occurrence (Fig. 4e). Fire occurrence from 2004 to 2013 was associated with short-term monthly (<1 month) and seasonal high temperature (<1 season; Fig. 4f). Long-term climate variability (>2 seasons) was less important for the occurrence of wildfire, regardless of decade. Across all three decades, topographic variables were not important to wildfire occurrence. The least important predictors of wildfire occurrence were the proportion of the area affected by MPB, the time since MPB initiation, and the peak MPB stage present at time of wildfire (Table 1). Random forest out-of bag error was low for each decadal model (Appendix S1: Fig. S2d–f). MDS indicated separability between wildfire and non-wildfire occurrences (Fig. 5d–f).

Classification trees results indicate that hot and dry short-term (monthly) conditions were more important to fire occurrence in the WUI than were the presence or timing of MPB (Appendix S1: Fig. S3) across all decades. For the decades of 1984–1993 and 1994–2003, a combination of high monthly temperatures during the fire event and steep slopes explained the most variance. MPB presence across these two decades was not important for predicting fire occurrence. During 2004–2013, high monthly temperatures and long-term drought predicted 56% of the fire occurrences within the WUI (Appendix S1: Fig. S3c).

Using the top four climate variable for each decade and all four MPB variables, each classification tree result indicated climate being the major determinate for fire occurrence (Appendix S1: Fig S6). During the period from 1993 to 2004, there was a 2% probability that fire would occur given high monthly temperatures during the fire and high ratio of MPB to area burned. This contrasts the relationship between high temperatures and no MPB outbreak, which resulted in a 33% probability of fire for that same time period. Similarly, between 2004 and 2013, the probability of fire occurrence given high monthly temperatures and high MPB ratio to area burned during was 10%, while high temperatures and no MPB outbreak present resulted in 36% probability of fire.

DISCUSSION

The present study shows that the extensive and severe outbreaks of mountain pine beetle (MPB) that have killed trees across millions of hectares of forests and woodlands across the western United States over the past three decades have not influenced the occurrence of large wildfires during this period. Instead, the coincident

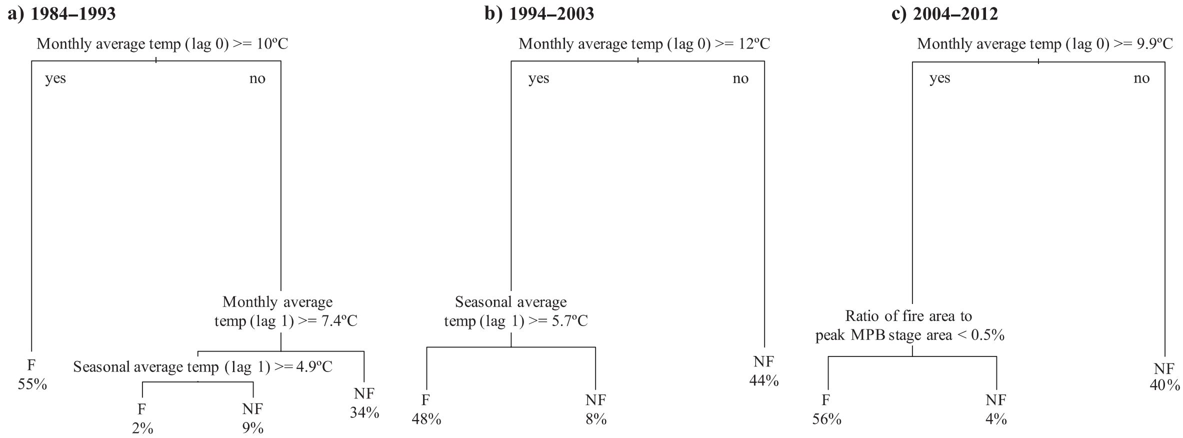

FIG. 6. Pruned classification tree of wildfire occurrence in the western USA (a) 1984–1993, (b) 1994–2003, and (c) 2004–2013, showing significant predictor variables in order of importance based on explained variance. Trees were produced using a 10-fold cross-validation based on top four random forest outputs and all four MPB predictors. Each terminal node for the wildfire occurrence classification tree shows the probability of wildfire or no wildfire for that node. The model cross-validation error rate was (a) 15%, (b) 12.5%, and (c) 17%, respectively. Complete trees were pruned based on the cost-complexity measure and included equal fire and non-fire samples, where total samples were (a) 435, (b) 964, and (c) 1,014, respectively.

increase of large wildfires has been driven by warm and dry conditions. These relationships hold for lodgepole pine forests across the western USA, as well as those in the wildland-urban interface (WUI), where houses and other structures are particularly vulnerable to wildfire. Our findings are consistent with studies that have shown that area burned is not associated with preceding MPB outbreaks across smaller areas (Lynch et al. 2006, Kulakowski and Veblen 2007, Liang et al. 2014, Meigs et al. 2015, Meigs et al. 2016) as well as large areas during dry years (Hart et al. 2015*a*). Our findings are also consistent with studies that have shown that climatic variation is more important than MPB outbreaks across smaller areas (Kulakowski and Jarvis 2011, Harvey et al. 2014*a*, *b*). The current study builds on these findings by directly examining the relative importance of climatic variation and MPB outbreaks on wildfire occurrence at a sub-continental scale.

Across the western United States from 1984 to 2013 fires and MPB both increased concurrently. At a broad spatial scale, this may imply that outbreaks have been contributing to an increase in wildfires, but our analysis shows that that the increase in large wildfires during this period actually has been driven primarily by warm and dry climatic conditions, rather than by outbreaks of MPB (Fig. 3). Even as outbreaks of MPB have become more extensive over the past three decades, fires have been no more likely to occur in beetle-affected stands than in unaffected ones. Even given possible spatiotemporal limitations in the aerial detection survey data during the course of outbreak development (i.e., less complete mapping during earlier phases of outbreaks), we found climate to be predominantly and consistently associated with fire occurrence across all decades. Interestingly, during the most recent decade (2004–2013), when MPB outbreaks were most extensive, wildfires were less likely to have occurred in beetle-affected stands, suggesting that MPB outbreaks may have reduced the occurrence of wildfire across lodgepole pine stands in the western USA. Although dramatic changes in forest fuels following outbreaks do affect the flammability of needles and trees, the current analysis shows that outbreaks have been negligible in determining wildfire occurrence across large forested areas, in contrast to the dominant and overriding effects of climate. Both ignitions and fuel continuity are important in shaping fire regimes, but in relatively mesic and dense forests, such as those dominated by lodgepole pine, fuels and other vegetation conditions are much less important than the effects of climate (Schoennagel et al. 2004). As a caveat, we recognize that the current study has focused on large (>400 ha) wildfires, which are ecologically and socially most important, but in doing so, it has left open the possibility that relationships among climate, outbreaks and fires may be different for smaller (<400 ha) wildfires.

Previous work has shown that the occurrence of large wildfires over long periods and large areas has been strongly influenced by climatic variation (e.g., Dennison et al. 2014). Our findings show that climate continues to be the most important factor driving wildfires, even in the context of unprecedented outbreaks of bark beetles. Consequently, the occurrence of large wildfires should be expected to increase under future climate scenarios (Moritz et al. 2012, Pierce et al. 2013), regardless of future outbreaks (Bentz et al. 2010). However, while temperature and precipitation are the most important factors driving sub-continental occurrence of large wildfires, fuel availability, which will depend not only on disturbances, but also on variability in post-disturbance regeneration (Vyse et al. 2009, Diskin et al. 2011, Kayes and Tinker 2012), will also continue to be important in modulating disturbance interactions under future climate scenarios.

Furthermore, specific relationships among climate, outbreaks, and fires are likely to vary across ecoregions and understanding that variability continues to be an important research goal (Littell et al. 2009).

Some previous studies have suggested that fire risk may increase following outbreaks, especially during the red phase, when dry needles are still present on beetle-killed trees (Hicke et al. 2012). However, we found no evidence that wildfire occurrence has increased across the western USA, even during the red phase. Although the most visually striking consequence of MPB outbreaks is the prevalence of red needles on recently killed trees, the most important consequence for actual wildfire is likely to be reduced canopy bulk density (CBD) as beetle-killed trees lose their needles (Simard et al. 2011). CBD is important to fire regimes and fire behavior because it is an indicator of the amount and continuity of canopy fuels that are available to burn and carry a fire. How quickly CBD decreases following tree mortality and how this varies with biophysical setting remains poorly understood and a subject of debate. Some conceptual frameworks suggest that CBD remains unchanged initially following an outbreak (Hicke et al. 2012, Jenkins et al. 2012). In contrast, the limited empirical data that directly address this issue indicate that CBD decreases shortly after tree mortality (Simard et al. 2011), although it is not known how universal this phenomenon is. How quickly CBD decreases following outbreaks necessarily hinges on the synchrony of beetle attack and tree death within a stand, as well as site conditions (Hicke et al. 2012, Kulakowski and Veblen 2015). For instance, after substantial reduction of stand-scale foliar moisture content (FMC), stand-level CBD would be most likely to remain unchanged following outbreaks if 100% of trees in a stand were killed during the initial year of the outbreak and if site moisture, temperature, and wind conditions promoted retention of dead needles. If an outbreak lasts several years within a given stand (as it normally does; Schmid and Amman 1992), then it becomes increasingly likely that some trees would lose their needles before other trees are killed, effectively reducing CBD synchronously with reductions in stand-scale FMC.

Our analysis included several predictor variables related to pre-fire MPB, including the time since an outbreak was first detected and the time since the peak of the outbreak. But the only beetle-related variable that influenced wildfire occurrence was the proportion of the burned area to have been affected by outbreak, such that wildfire occurrence was lower in stands that had been affected by outbreaks. Our data indicate that by the time >0.5% of a stand is affected by outbreak, the outbreak began an average of 12 years earlier. In such scenarios, it is likely that trees that were first killed by the outbreak had lost their needles and CBD was reduced by the time of the wildfire. The current study has examined disturbance interactions across a very broad spatial scale. However, the effect of outbreaks on canopy and surface fuels can vary within affected stands due to variations in pre-outbreak forest structure, severity of outbreak, and underlying environmental heterogeneity (e.g., Donato et al. 2013). Understanding how this type of heterogeneity affects disturbance interactions remains an important research goal.

The vast majority of areas affected by bark beetle outbreaks are in remote areas rather than in the wildland–urban interface (Aronson and Kulakowski 2012). Nevertheless, outbreaks that do occur in the WUI pose substantial hazards, including increased tree falls. However, we have shown that the fires that have occurred in the WUI have been primarily determined by antecedent climate conditions rather than outbreaks. Future climate scenarios are likely to promote large wildfires and insect outbreaks, unless host depletion creates negative feedbacks (Temperli et al. 2015, Hart et al. 2015*b*). Our findings are consistent with other previous studies that suggest fuel reduction treatments in the immediate vicinity of homes and communities, rather than in beetle-affected forests, are likely to most effectively reduce wildfire risk to those structures.

Conclusion

Over the past 30 years, occurrence of large wildfires across the western United States has been driven primarily by warm and dry antecedent climate conditions and not by the extensive and severe outbreaks of mountain pine beetle. Recent trends of increasing co-occurrence of wildfires and outbreaks are due to a common climatic driver rather than interactions between these disturbances. Therefore, reducing wildfire risk in relatively mesic forests, such as those dominated by lodgepole pine, where fuels do not limit wildfire occurrence, hinges on addressing the underlying climatic drivers, rather than treating beetle-affected forests.

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