The influence of mountain pine beetle outbreaks and drought on severe wildfires in northwestern Colorado and southern Wyoming: A look at the past century

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**A R T I C L E   I N F O**

- Article history:
  - Received 26 March 2011
  - Received in revised form 6 July 2011
  - Accepted 12 July 2011

- Keywords:
  - Disturbance interactions
  - Colorado
  - Pinus contorta
  - Dendroctonus ponderosae

**A B S T R A C T**

Outbreaks of bark beetles and drought both lead to concerns about increased fire risk, but the relative importance of these two factors is the subject of much debate. We examined how mountain pine beetle (MPB) outbreaks and drought have contributed to the fire regime of lodgepole pine forests in northwestern Colorado and adjacent areas of southern Wyoming over the past century. We used dendroecological methods to reconstruct the pre-fire history of MPB outbreaks in twenty lodgepole pine stands that had burned between 1939 and 2006 and in 20 nearby lodgepole pine stands that were otherwise similar but that had not burned. Our data represent c. 80% of all large fires that had occurred in lodgepole pine forests in this study area over the past century. We also compared Palmer Drought Severity Index (PDSI) and actual evaportranspiration (AET) values between fire years and non-fire years. Burned stands were no more likely to have been affected by outbreak prior to fires than were nearby unburned stands. However, PDSI and AET values were both lower during fire years than during non-fire years. This work indicates that climate has been more important than outbreaks to the fire regime of lodgepole pine forests in this region over the past century. Indeed, we found no detectable increase in the occurrence of high-severity fires following MPB outbreaks. Dry conditions, rather than changes in fuels associated with outbreaks, appear to be most limiting to the occurrence of severe fires in these forests.

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1. Introduction

Across western North America ongoing outbreaks of bark beetles (Dendroctonus spp.) have killed trees over hundreds of thousands of square km, especially in forests dominated by lodgepole pine (Pinus contorta Dougl. ex Loud. var. latifolia Engelm) (Raffa et al., 2008). These outbreaks have high economic costs and have led to widespread concern among the public, managers, and policy makers about the increased risk of forest fires. However, there is ongoing debate about how important outbreaks actually are to fire risk in contrast to the potentially overriding influence of climate and fire weather on the fire regime of these forests.

Bark beetles (Dendroctonus spp.) are native to the forests of western North America. Under endemic conditions they normally infest and kill only the largest and most stressed trees that are scattered throughout a forest (Safranyik, 2004). However, when climate warms and forest conditions are appropriate, populations can to epidemic size. Under such outbreak conditions beetles mount pheromone-mediated mass attacks that overwhelm the natural defenses of healthy trees and can kill lodgepole pine over many thousands of hectares (Raffa, 2001; Raffa and Berryman, 1983; Safranyik, 2004).

Despite the long-standing expectation that bark beetle outbreaks increase the risk of wildfires, recent research has shed doubt on this expectation for some forest types. In considering this topic, it is important to distinguish “fuel hazard”, which normally refers to the fuel complex (type, volume, arrangement) that determines the probability of ignition and resistance to control regardless of the fuel type’s weather-influenced moisture content (Hardy, 2005) versus “fire risk”, which refers to the chance that a fire might start based on all causative agents (i.e. fuel hazard, ignition source and weather). Thus, we use “risk” to refer to the probability of a fire event actually occurring. Several studies in Colorado have generally concluded that outbreaks of spruce beetle (Dendroctonus rufipennis Kirby, Coleoptera: Scolytidae) in forests dominated by Engelmann spruce (Picea engelmannii Parry ex Engelm) and subalpine fir (Abies lasiocarpa (Hook.) Nutt.) do little or nothing to increase the risk of wildfires. For example, after a 1940s spruce beetle outbreak that killed trees over thousands of hectares of subalpine forests in Colorado, there was no increase in the numbers of fires compared to unaffected spruce-fir forests (Bebi et al., 2003). Furthermore, beetle-affected stands more susceptible to a low-severity fire that spread through adjacent were not forest several
years after the outbreak subsided (Kulakowski et al., 2003). During the severe drought of 2002, large fires affected extensive areas of Colorado, including some spruce-fir stands that were previously affected by the 1940s outbreak of spruce beetle, but the outbreak had only a minor influence on the behavior of this fire (Bigler et al., 2005). Likewise, outbreaks of spruce beetle that were ongoing at the time had no detectable effect on the extent or severity of fires that occurred in 2002 in Colorado (Kulakowski and Veblen, 2007). These empirical findings are consistent with modeling studies that predict reductions in the probability of active crown fire after spruce beetle outbreaks due to reductions in canopy bulk density (Derose and Long, 2009; Jenkins et al., 2008; Page and Jenkins, 2007b). However, modeling studies should be reviewed with caution since many crown fire models greatly underestimate fire potential (Cruz and Alexander, 2010).

Instead of outbreaks being the most important determinants of fires in spruce-fir forests, climatic conditions appear to have an overriding effect on fire regimes in these ecosystems. It is well established that in spruce-fir forests, extensive fires are highly dependent on infrequent, severe droughts (Buechling and Baker, 2004; Schoennagel et al., 2007; Sibold et al., 2006; Sibold and Veblen, 2006). Under such extreme drought conditions, increased dead fuels from bark beetle outbreaks appear to play only a minor role, if any, in increasing fire risk.

Although the evidence is compelling that outbreaks of spruce beetles do not substantially increase the risk of wildfires in spruce-fir forests, it is less clear what effect outbreaks of mountain pine beetle (MPB; Dendroctonus ponderosae Hopk. Coleoptera: Scolytidae) have on the fire regime of lodgepole pine forest. In contrast to spruce-fir forests, mature lodgepole pine forests often tend to be less dense and located in more xeric settings, and thus the fire risk in these forests may be more sensitive to changes in fuels such as those brought about by outbreaks. Outbreaks of mountain pine beetle have been shown to clearly alter fuel structures (Klutsch et al., 2009; Page and Jenkins, 2007a; Simard et al., 2011), but the actual effects of these changes in fuels on subsequent fire risk and behavior have not yet been fully elucidated.

Lodgepole pine stands affected by MPB outbreaks in 1972–1975 had an 11% higher probability of burning in the 1988 Yellowstone fires but stands that were affected by outbreaks in 1980–1983 were not more likely to have burned in the same fire (Lynch et al., 2006), which is consistent with modeling studies of predicted fire behavior following outbreaks (Page and Jenkins, 2007a,b; Jenkins et al., 2008). Kulakowski and Veblen (2007) found that outbreaks of MPB that were ongoing at the time did not affect the extent or severity of a wildfire in 2002 in Colorado and suggested that changes in fuels brought about by outbreaks may be overridden by climatic conditions. Likewise, following outbreaks in Yellowstone National Park (YNP) canopy moisture content and canopy bulk density were both reduced (Simard et al., 2011). Although the canopy was drier immediately after outbreaks in YNP, simulation modeling found no increase in fire risk because of the more important effect of reductions in canopy bulk density (Simard et al., 2011). In Colorado, West (2010) examined the actual occurrence of fires following outbreak and found that MPB-caused

![Fig. 1. Location of sampling sites.](image-url)
The large severe fires that have shaped lodgepole pine forests of Colorado over the past several centuries have occurred during short-term periods of significant drought (negative PDSI), which have been associated with extreme cool (negative) phases of El Nino Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO) and positive departures from mean Atlantic Multi-decadal Oscillation (AMO) values (Schoennagel et al., 2007; Sibold and Veblen, 2006).

Emerging research does not support the hypothesis that outbreaks of MPB increase the risk of wildfires in lodgepole pine forests but does suggest that fire risk is strongly associated with climatic conditions. However, our current understanding of how outbreaks influence the risk and spread of fire is based primarily on modeling studies and empirical studies that have only examined individual fire events or changes in fuels (but see West, 2010). There is a paucity of studies that have examined the degree to which outbreaks have been associated with actual wildfires in lodgepole pine forest over a long period of time. The central question of the present study was: did stands of lodgepole pine that burned in stand-replacing fires over the past century have a different history of pre-fire MPB outbreaks than nearby stands that did not burn?

### Table 1

Approximate stand origin dates, years in which outbreaks of mountain pine beetle occurred, years in which fires occurred, and PDSI values during fire years.

<table>
<thead>
<tr>
<th>Site</th>
<th>Origin</th>
<th>Pre-fire MPB</th>
<th>Fire year</th>
<th>Fire size (ha)</th>
<th>PDSI during fire year</th>
</tr>
</thead>
<tbody>
<tr>
<td>1U</td>
<td>1720s</td>
<td>–</td>
<td>2002</td>
<td>25</td>
<td>–4.755</td>
</tr>
<tr>
<td>1B</td>
<td>1720s</td>
<td>1990s⁴</td>
<td>2002</td>
<td>25</td>
<td>–4.755</td>
</tr>
<tr>
<td>4U</td>
<td>1880s</td>
<td>1990s¹</td>
<td>2002</td>
<td>8</td>
<td>–1.976</td>
</tr>
<tr>
<td>4B</td>
<td>1750s/1880s</td>
<td>1990s¹</td>
<td>2002</td>
<td>8</td>
<td>–1.976</td>
</tr>
<tr>
<td>6U</td>
<td>1690s</td>
<td>1990s¹</td>
<td>2002</td>
<td>1781</td>
<td>–4.755</td>
</tr>
<tr>
<td>6B</td>
<td>1690s</td>
<td>1990s¹</td>
<td>1999</td>
<td>4094</td>
<td>2.668</td>
</tr>
<tr>
<td>10U</td>
<td>1870s</td>
<td>–</td>
<td>1999</td>
<td>121</td>
<td>2.668</td>
</tr>
<tr>
<td>1U</td>
<td>1900s</td>
<td>–</td>
<td>1999</td>
<td>121</td>
<td>2.668</td>
</tr>
<tr>
<td>11B</td>
<td>1720s/1850s</td>
<td>–</td>
<td>1999</td>
<td>121</td>
<td>2.668</td>
</tr>
<tr>
<td>13U</td>
<td>1880s</td>
<td>–</td>
<td>1999</td>
<td>121</td>
<td>2.668</td>
</tr>
<tr>
<td>13B</td>
<td>1880s</td>
<td>–</td>
<td>2006</td>
<td>422</td>
<td>–0.468</td>
</tr>
<tr>
<td>14U</td>
<td>1710s</td>
<td>1990s²/2000s³</td>
<td>2006</td>
<td>422</td>
<td>–0.468</td>
</tr>
<tr>
<td>14B</td>
<td>1710s</td>
<td>1990s²/2000s³</td>
<td>2006</td>
<td>422</td>
<td>–0.468</td>
</tr>
<tr>
<td>15U</td>
<td>1880s</td>
<td>2000s³</td>
<td>2005</td>
<td>56</td>
<td>1.488</td>
</tr>
<tr>
<td>15B</td>
<td>1880s</td>
<td>2000s³</td>
<td>2005</td>
<td>56</td>
<td>1.488</td>
</tr>
<tr>
<td>16U</td>
<td>1660s</td>
<td>1990s/2000s³</td>
<td>2005</td>
<td>71</td>
<td>1.488</td>
</tr>
<tr>
<td>16B</td>
<td>1660s</td>
<td>1990s/2000s³</td>
<td>2005</td>
<td>71</td>
<td>1.488</td>
</tr>
<tr>
<td>17U</td>
<td>1640s</td>
<td>–</td>
<td>1962</td>
<td>121–405³</td>
<td>0.469</td>
</tr>
<tr>
<td>17B</td>
<td>1640s</td>
<td>–</td>
<td>1988</td>
<td>405–2023³</td>
<td>–0.256</td>
</tr>
<tr>
<td>18U</td>
<td>1660s</td>
<td>–</td>
<td>1939</td>
<td>405–2023³</td>
<td>–2.013</td>
</tr>
<tr>
<td>18B</td>
<td>1660s</td>
<td>–</td>
<td>1939</td>
<td>405–2023³</td>
<td>–2.013</td>
</tr>
<tr>
<td>19U</td>
<td>1650s</td>
<td>–</td>
<td>1952</td>
<td>405–2023³</td>
<td>1.453</td>
</tr>
<tr>
<td>19B</td>
<td>1650s</td>
<td>–</td>
<td>1952</td>
<td>405–2023³</td>
<td>1.453</td>
</tr>
<tr>
<td>20U</td>
<td>1880s</td>
<td>–</td>
<td>1952</td>
<td>405–2023³</td>
<td>1.453</td>
</tr>
<tr>
<td>20B</td>
<td>1880s</td>
<td>–</td>
<td>1952</td>
<td>405–2023³</td>
<td>1.453</td>
</tr>
<tr>
<td>21U</td>
<td>1860s</td>
<td>–</td>
<td>1952</td>
<td>405–2023³</td>
<td>1.453</td>
</tr>
<tr>
<td>21B</td>
<td>1860s</td>
<td>–</td>
<td>1952</td>
<td>405–2023³</td>
<td>1.453</td>
</tr>
<tr>
<td>22U</td>
<td>1670s</td>
<td>1940s⁴</td>
<td>–</td>
<td>–</td>
<td>0.272</td>
</tr>
<tr>
<td>22B</td>
<td>1670s</td>
<td>–</td>
<td>1978</td>
<td>405–2023³</td>
<td>–0.675</td>
</tr>
<tr>
<td>23U</td>
<td>1840s</td>
<td>–</td>
<td>1994</td>
<td>405–2023³</td>
<td>–2.154</td>
</tr>
<tr>
<td>23B</td>
<td>1840s</td>
<td>–</td>
<td>1994</td>
<td>405–2023³</td>
<td>–2.154</td>
</tr>
<tr>
<td>24U</td>
<td>1640s</td>
<td>1980s⁴</td>
<td>–</td>
<td>–</td>
<td>–0.107</td>
</tr>
<tr>
<td>24B</td>
<td>1640s</td>
<td>1980s⁴</td>
<td>–</td>
<td>–</td>
<td>–0.107</td>
</tr>
<tr>
<td>25 U</td>
<td>1740s</td>
<td>–</td>
<td>1972</td>
<td>16</td>
<td>–1.079</td>
</tr>
<tr>
<td>25 B</td>
<td>1740s</td>
<td>–</td>
<td>1972</td>
<td>16</td>
<td>–1.079</td>
</tr>
<tr>
<td>26 U</td>
<td>1850s</td>
<td>–</td>
<td>1980</td>
<td>129</td>
<td>1.463</td>
</tr>
<tr>
<td>26 B</td>
<td>1850s</td>
<td>–</td>
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<td>1956</td>
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<td>–4.011</td>
</tr>
<tr>
<td>27 B</td>
<td>1620s</td>
<td>–</td>
<td>1956</td>
<td>388</td>
<td>–4.011</td>
</tr>
</tbody>
</table>

1. Outbreaks that occurred following fire years were not considered in this analysis which is focused on outbreaks that preceded fire years.
3. Based on dendroecologically reconstructed mortality dates and USDA Forest Service data.
4. Based on dendroecologically reconstructed mortality dates.
5. Estimate based on Size Class of Fire. Exact extent of fire is not known.
6. Based on USDA Forest Service unpublished data.
2. Study area

The twenty study sites are located in a c. 150 km by 150 km area of western Colorado and adjacent areas of southern Wyoming (Fig. 1). Sites are dominated by lodgepole pine and range in elevation from 2420 m to 3017 m. High severity fires occurred at these sites between 1939 and 2006 and varied in size from 8 to 4094 ha (USDA Forest Service unpublished data). Median fire size was 263 ha. Over the past centuries forests dominated by lodgepole pine in this region have been characterized by infrequent high-severity fires (Kulakowski and Veblen, 2002; Kulakowski et al., 2003; Schoennagel et al., 2007; Sibold et al., 2006; Sibold and Veblen, 2006).

Based on data from a centrally located weather station (Steamboat Springs, CO), the study region has a continental climate with a mean annual temperature of 3.9 °C that ranges from a low mean monthly temperature of −9.5 °C in January to a high mean monthly temperature of 13.1 °C in July (Western Regional Climate Center, http://www.wrcc.dri.edu/). Mean annual precipitation is 61.0 cm and ranges from a mean monthly precipitation of 6.4 cm in January to 3.8 cm in July.

3. Methods

3.1. Site selection

USDA Forest Service maps and documents identified a total of 25 fires >8 ha that had occurred in lodgepole pine forests in Colorado and southern Wyoming over the past century (USDA Forest Service unpublished data). We set out to sample all 25 of these burned stands, but during the course of field work we were unable to locate evidence of several of the smaller fires and thus collected data at only 20 sites, each of which included pairs of burned and unburned stands (Fig. 1). Nevertheless, it is important to note that our data represent the vast majority of large (>8 ha) fires that have occurred in this region over the past century. Based on our field observations, most of these fires were stand-replacing, but two (Sites 11 and 23) were mixed severity (i.e. mortality of canopy trees <90%).

3.2. Field sampling

At each site we used well-established dendroecological methods that have been previously used in these forest types to reconstruct stand origin (Kipfmueller and Baker, 1998; Kulakowski and Veblen, 2002; Kulakowski et al., 2003) and the history of mountain pine beetle outbreaks (Alfaro et al., 2010; Axelson et al., 2009) in the burned stand and in an adjacent unburned stand. Stands are characterized by similar structure and composition, which is distinct from the structure and composition of adjacent forest. Unburned stands were located on similar aspect, elevation, and topographic positions as the corresponding burned stands. The stand structure (i.e. density and dbh and height distributions) and composition of unburned stands was also similar to the pre-fire structure and composition of corresponding burned stands.

![Fig. 2.](image-url) Dates of establishment, mortality (last complete year of growth), and releases (abrupt >200% increases in ring width sustained >10 year) in 10-year classes at Site 13 in an unburned stand (U) that established in c. 1880s and that was not affected by MPB outbreak (a–c) and in a burned stand (B) that established in c. 1880s and that was not affected by MPB outbreak (d–f). The vertical grey bar (e) indicates the year of the 2006 fire. The horizontal line (c, f) indicates sample depth (No. of trees).
Stand origin was reconstructed in all 40 stands (20 burned and 20 unburned) in order to verify that burned and unburned stands did not differ in stand age, which has an influence on stand structure and which can affect fire risk. A sampling point was located in a representative area of each stand (i.e. free from meadows, drainages, rock outcroppings, etc.). In the summer of 2007 increment cores were collected from 15 to 36 of the largest live and dead trees within a c. 100 m search area around the sampling point to obtain stand-origin and beetle-caused mortality dates. In unburned stands, dead trees were visually inspected in the field for evidence of MPB galleries and only dead trees with visible MPB galleries were sampled. As fires were high-severity, in burned stands dead trees were sampled regardless of the presence of visible MPB galleries in order to study mortality that may have resulted from either outbreaks or fires. Testing in similar forests has shown that subjectively selecting the largest trees gives a better estimate of stand age than randomly selecting trees (Kipfmueller and Baker, 1998). These trees were cored as close to the ground and to the pith as was possible. Additional cores were collected from neighbors of dead trees, which may have experienced an abrupt and sustained increase in growth resulting from a reallocation of resources following the death of the neighboring tree. Such releases in the tree ring record are sometimes used to reconstruct disturbances such as outbreaks (Alfaro et al., 2010; Axelson et al., 2009). Cores were processed in the laboratory using standard dendroecological procedures (Stokes and Smiley, 1968). All tree cores collected in the field were air-dried then glued to wooden mounts and sanded with successively finer grains of sand paper until ring boundaries are clear enough for microscopic inspection (Stokes and Smiley, 1968). The annual rings were then counted and measured to the nearest 0.01 mm with a stereomicroscope, slide-stage micrometer, and a the T-SAP software. Increment cores from a total of 920 trees were successfully processed (see Appendix A for mean series intercorrelation for each site).

3.3. Reconstruction of stand disturbance history

Stand disturbance history was reconstructed based on stand-origin dates and dates of mortality of dead trees. We also examined dates of releases (abrupt >200% increases in ring width sustained >10 years) in remnant trees. These methods have been widely used to reconstruct disturbance history in these forest types (e.g. Alfaro et al., 2010; Axelson et al., 2009; Kipfmüller and Baker, 1998; Kulakowski and Veblen, 2002; Kulakowski et al., 2003). No evidence of other extensive disturbances (e.g. wind) was found at any sites. Past outbreaks can be distinguished from past wind blowdowns by the coincident death dates of standing trees in the former versus the coincidence of azimuths of the logs in the latter. Furthermore, trees killed in windstorms are more likely to be up-rooted while trees that fall after they are dead are more likely to have snapped boles (Veblen et al., 2001).

Methods of reconstructing stand origin dates are as those that have been previously used in these forests (e.g. Kulakowski and Veblen, 2002; Kulakowski et al., 2003). Because fires that shape
Colorado subalpine forests are primarily large, severe, and infrequent, our primary aim was to group stands into broad age classes that most likely arose following such fires. Post-fire stands typically have identifiable pulses of establishment. Dates of establishment of the oldest sampled trees in each stand were used as an approximation of the stand-origin date. Allowing for a lag time between the fire and seedling establishment, the stand origin date indicates the approximate time of such a disturbance.

Methods of reconstructing history of mountain pine beetle (MPB) outbreak are as those described in Axelson et al. (2009) and Alfaro et al. (2010). Outbreaks of MPB normally result in coincident mortality of large (>10 cm dbh) lodgepole pine during the outbreak and coincident releases (abrupt and sustained increases in growth pattern) of the surviving trees if non-host species or small diameter lodgepole are present in the stand at the time of outbreak (Axelson et al., 2009; Alfaro et al., 2010; Campbell et al., 2007; Taylor et al., 2006). Furthermore, distinct larval galleries and the presence of blue-stain fungus (Multiple Ophiostoma spp.) are corroborating evidence of MPB (Safranyik and Wilson, 2006). The occurrence of an outbreak was determined in each sampled stand by dating the year of mortality of dead trees and checking tree cores for coincident releases in living trees (Veblen et al., 1991a). In burned stands, this mortality was distinguished from mortality due to fire, which occurred in known years. The occurrence of outbreaks that may have occurred since 1994 was verified using Aerial Detection Survey (ADS) data (http://www.fs.fed.us/r2/resources/fhm/aerialsurvey/), where such data were available. As the severity of outbreaks could not be measured due to inherent methodological limitations, our focus was on reconstructing the presence or absence of outbreaks.

3.4. Effect of outbreaks and drought on fire occurrence

We reasoned that if outbreaks of bark beetles increase the risk of wildfire then stands that had burned should have a higher incidence of pre-fire outbreaks than otherwise similar stands that had not burned. Thus, a Fisher Exact test was used to compare the incidence of pre-fire outbreaks in burned stands versus unburned stands. We used two indexes of drought: (1) Regional summer Palmer Drought Severity Index (PDSI) values are scaled to climate of a region and have previously been found to be related to fire occurrence in subalpine forests in Colorado (Schoennagel et al., 2007; Sibold and Veblen, 2006); and (2) Raw water deficits may be interpreted relative to the total amount of precipitation required to ameliorate drought. Regional summer PDSI values for fire years and non-fire years between 1900 and 2006 were derived based on data from Cook (2008), grid point 131. One thousand Monte Carlo simulations and a subsequent t-test were used to compare mean PDSI values during all unique fire years (n = 15) versus values during all non-fire years (n = 92). We also calculated summer (June–August) water deficits using a water balance model based on a modified Thornthwaite method (Willmott et al., 1985) using data from a centrally-located weather station (Steamboat Springs, CO), which has a continuous record of temperature and precipitation from 1909 to the present. Other climate stations are located within our study region but these had records that were either too short...
or too incomplete to use for our analysis. One thousand Monte Carlo simulations and a subsequent Mann–Whitney Rank-Sum test were used to compare median summer actual evapotranspiration (AET) values during all unique fire years \((n = 15)\) versus values during all non-fire years \((n = 83)\).

4. Results

4.1. Reconstruction of disturbance history

Data on all sampled stands are listed in Table 1 and data from select stands are displayed in Figs. 2–7, which illustrate how dendroecological data were used to reconstruct disturbance history. Dendroecological data from all stands are not displayed in figures due to space constraints. Based on coincident establishment dates, stands originated between the 1620s and the 1900s (Table 1, and Figs. 2–7). Although the goal of the present study was not to focus on the long-term fire history of lodgepole pine stands, we note that the pulses of coincident establishment that we observed are consistent with patterns of tree establishment following stand-replacing fires. All but two stands showed the presence of only one distinguishable cohort. Pairs of burned and unburned stands shared the same stand-origin dates in most cases (Table 1). At sites 4 and 11 the burned stands were made up of two distinguishable cohorts, whereas the unburned stands were made up of only one. As our field observations indicated that the younger of these cohorts were more dominant in both cases, these two sites were retained for further analysis. Based on pulses of mortality that preceded dates of stand-replacing fires and ADS data, stands showed evidence of having been affected by MPB outbreaks in the 1940s, 1980s, 1990s, and 2000s.

Stands that were interpreted as not having been affected by outbreak lack pulses of mortality except for mortality during the fire year (e.g. Fig. 2). In contrast, stands that were interpreted as having been affected by outbreak exhibit pulses of mortality in burned and unburned stands that precede the fire year (e.g. Figs. 3–7). In the case of what we interpret to be a 1940s outbreak, a pulse of mortality of lodgepole pine began in the 1940s and continued through the 1960s. Contrary to our expectations that coincident mortality dates should be associated with corresponding release dates, relatively few sites exhibited this relationship. Thus our reconstruction of outbreaks is primarily based on the mortality dates of dead lodgepole pine that exhibited MPB galleries. The most likely explanation as to why releases were not associated with mortality dates is that lodgepole pine stands often lack abundant advanced regeneration, which would be released following the mortality of canopy trees. Alternatively, our criteria of releases (abrupt >200% increases in ring width sustained >10 years) was too strict to detect releases that may be more minor in lodgepole pine forests.

4.2. Effect of outbreaks and drought on fire occurrence

Of the 20 unburned stands, seven showed evidence of MPB outbreak prior to the date of the corresponding wildfires. However, of
the 20 burned stands, only six showed evidence of MPB outbreaks prior to wildfires. There was no statistical difference between the proportion of burned and unburned stands that were affected by outbreak (Fig. 8; Fisher Exact Test $p > 0.05$). For recent (1990s and 2000s) outbreaks, trees killed by MPB in burned stands had clearly visible galleries and many still had intact bark. However, it may be possible that following older outbreaks (e.g. 1940s), some evidence of outbreaks may have been erased by subsequent fires that occurred many decades after the outbreaks, when beetle-killed trees may have been more prone to partial or complete combustion. Thus, we analyzed these data while omitting sites affected by the two oldest outbreaks (1940s and 1980s) from the analysis. When these sites are excluded from the analysis, there was still no difference between the proportion of burned and unburned stands that were affected by outbreak (Fisher Exact Test $p > 0.05$). Between 1900 and 2006, the mean PDSI value during fire years ($\bar{C}_0 = 1.100$) was significantly lower than the mean PDSI value during non-fire years ($\bar{C}_0 = 0.139$) ($t = 123.213; p < 0.001$). Between 1909 and 2006, the median summer AET value during fire years (153.000) was significantly lower than the median summer AET value during non-fire years (193.997) (Mann–Whitney U Statistic = 414.000; $t = 123.213; p = 0.025$).

5. Discussion

5.1. Effect of outbreaks and drought on fire occurrence

This study suggests that over the past century drought has been more important than MPB outbreaks in determining the occurrence of severe fires in lodgepole pine forests of northwestern Colorado and adjacent areas of southern Wyoming. Stands that were burned in high-severity fires were no more likely to have been affected by outbreaks than nearby stands that did not burn. However, years during which fires occurred were significantly drier (based both on PDSI and AET) than years during which fires did not occur.

The present study is consistent with recent research that has de-emphasized the importance of MPB outbreaks in increasing fire risk in lodgepole pine (Kulakowski and Veblen, 2007; Simard et al., 2011). The most likely explanation for why severe fires are not more strongly associated with MPB outbreaks than would be expected by chance is twofold: First of all, the effect of outbreaks in reducing foliar moisture content is outweighed by reductions in canopy bulk density, which has a net effect of reducing fire hazard (Simard et al., 2011). Second of all, given the importance of infrequent but extreme drought to fires in lodgepole pine forests (Schoennagel et al., 2007), it appears that the abundance of live versus dead fuels is not generally limiting to the occurrence of severe fires. Simard et al. (2011) examined fuel conditions for 35 years following outbreaks of MPB in Yellowstone National Park and documented reduced canopy moisture content after outbreaks, which was coupled with reduced canopy bulk density. In simulation models of fire behavior, these authors found that under most weather conditions changes in fuels brought about by MPB outbreaks had little effect on fire behavior and under some weather conditions outbreaks actually reduced the probability of active crown fire. Thus, although the canopy of affected stands is drier immediately after outbreaks due to the abundance of dead trees,
no increase in fire hazard is observed because the killed trees rapidly lose their needles and effectively reduce canopy bulk density. Other independent modeling studies have also predicted a reduced risk of active crown fire five to 60 years after outbreaks due to decreased canopy bulk density (Jenkins et al., 2008; Page and Jenkins, 2007b).

Although our findings are consistent with emerging research, we note that there are inherent methodological difficulties associated with dendroecologically reconstructing pre-fire history of forest stands that had burned in high-severity fires. For example, trees killed by MPB in recent (1990s and 2000s) outbreaks in stands that had subsequently burned, had clearly visible beetle galleries and many still had intact bark. However, it may be possible that following the older outbreaks (e.g. 1940s), some evidence of outbreaks may have been erased by subsequent fires that occurred many decades after the outbreaks, when beetle killed trees may have been more prone to partial or complete combustion. Thus, the findings of the current study are more robust regarding more recent disturbance interactions as well as the effect of outbreaks on fires that occurred many decades prior to fires.

While our research indicates that outbreaks of MPB are not associated with increased risk of wildfire, it does indicate that fire risk is associated with drought conditions. One caveat of the current work is that the past outbreaks reconstructed in this study were not as extensive as are the ongoing outbreaks, which are unprecedented during the past century. Other recent studies have and continue to examine the effects of the ongoing outbreaks on fire risk and hazard (see citations above). In this context, the main contribution of the current study is the finding that over the
long-term, the occurrence of severe fires has not increased following outbreaks of MPB. Rather climatic variability appears to have been the most important determinant of fire activity in these lodgepole pine forests. Over the past centuries, forests dominated by lodgepole pine have been characterized by infrequent high-severity fires that have occurred during drought conditions (Buechling and Baker, 2004; Schoennagel et al., 2007; Sibold and Veblen, 2006; Sibold et al., 2006). Climate appears so important to the occurrence of wildfires in these forests that changes in stand structure brought about by outbreaks appear to do little or nothing to increase the risk of fires. As lodgepole pine forests are characteristically dense, fuels are generally not limiting to the occurrence of fire. Instead, the relatively mesic conditions of these forests make drought the limiting factor to the occurrence of large severe fires.

5.2. Conclusion

The present research indicates that over the past century the occurrence of severe fires in lodgepole pine forests in this region has been primarily influenced by climatic conditions rather than changes in fuels caused by bark beetle outbreaks. We suggest that the concern for an increase in fire risk due to changes in fuels following outbreaks has been misplaced relative to the increased fire risk associated with climatic conditions.

Acknowledgements

This work was supported by the National Science Foundation under Awards DEB 0743351 and 0743498. For research assistance we thank C. Griffin, M. Trivette, C. Kriso, J.M. Smith, and H. Westermann. For helpful comments on an earlier draft of this paper we thank T. Veblen and D. West.

Appendix A. Supplementary material


References


