Review

Bark beetles, fuels, fires and implications for forest management in the Intermountain West

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Abstract

Bark beetle-caused tree mortality in conifer forests affects the quantity and quality of forest fuels and has long been assumed to increase fire hazard and potential fire behavior. In reality, bark beetles, and their effects on fuel accumulation, and subsequent fire hazard, are poorly understood. We extensively sampled fuels in three bark beetle-affected Intermountain conifer forests and compared these data to existing research on bark beetle/fuels/fire interactions within the context of the disturbance regime. Data were collected in endemic, epidemic and post-epidemic stands of Douglas-fir, lodgepole pine and Engelmann spruce. From these data, we evaluated the influence of bark beetle-caused tree mortality on various fuels characteristics over the course of a bark beetle rotation. The data showed that changes in fuels over time create periods where the potential for high intensity and/or severe fires increases or decreases. The net result of bark beetle epidemics was a substantial change in species composition and a highly altered fuels complex. Early in epidemics there is a net increase in the amount of fine surface fuels when compared to endemic stands. In post-epidemic stands large, dead, woody fuels, and live surface fuels dominate. We then discuss potential fire behavior in bark beetle-affected conifer fuels based on actual and simulated fuels data. Results indicated that for surface fires both rates of fire spread and fireline intensities were higher in the current epidemic stands than in the endemic stands. Rates of spread and fireline intensities were higher in epidemic stands due, however, to decreased vegetative sheltering and its effect on mid-flame wind speed, rather than changes in fuels. Passive crown fires were more likely in post-epidemic stands, but active crown fires were less likely due to decreased aerial fuel continuity. We also discuss the ecological effects of extreme fire behavior. Information is presented on managing forests to reduce the impact of bark beetle outbreaks and the interplay between management, bark beetle populations, fuels and fire hazard and behavior.

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1. Introduction

The interactions among disturbances operating over various spatial and temporal scales determine the nature of forested landscapes. In some western forest systems, for example fire-dependent lodgepole pine forests, the influence of a single disturbance agent, such as wildfire, is so pervasive that it dominates forest dynamics. In most low to mid-elevation ponderosa pine forests, the relatively rapid accumulation of fuels and weather conditions required for fire ignition and spread make seasonal fires common. Fire return intervals from as few as 5 up to 100 or more years are capable of driving forest succession (Jenkins et al., 1998). In high elevation, subalpine forests, lightning is relatively common and sufficient amounts of fuels accumulate to sustain fire. However, short snow-free periods, low temperatures, and high relative humidities often preclude fire occurrence for extended periods (Arno, 2000). In these forest systems, *Dendroctonus* bark beetles (Coleoptera: Curculionidae, Scolytinae) coupled with other widespread disturbance events such as harvesting, snow avalanches and wind throw primarily regulate forest dynamics (Veblen et al., 1994; Jenkins et al., 1998). A conceptual model of the interactions of agents of disturbance over large spatial and long temporal scales is presented in Fig. 1.

Bark beetles naturally occur throughout the ranges of their host trees and have a critical role within healthy, functioning ecosystems (Roe and Amman, 1970; Geiszler et al., 1980; Gara et al., 1985; Veblen et al., 1994; Jenkins et al., 1998; Bebi et al., 2003; Kulakowski et al., 2003). During endemic population phases, bark beetles typically infest blowdown and trees weakened by fire, diseases, and drought (Schmid, 1981; Raffa and Berryman, 1983; Mattson and Haack, 1987; Goheen and Hansen, 1993; Cunningham et al., 2005). Periodically, populations of bark beetles reach outbreak levels infesting otherwise healthy trees (Raffa, 2001; Samman and Logan, 2000).

The level of bark beetle activity fluctuates with the availability of suitable host material, stand conditions, environmental factors, and the abundance of parasites and predators. Although many factors may contribute to the initiation of outbreaks, stand conditions largely determine the

potential for bark beetle population growth and spread (McCambridge and Knight, 1972; Schmid and Frye, 1977; Cole and Amman, 1980; Bartos and Amman, 1989; Anhold et al., 1996; Hansen et al., 2001; Hindmarch and Reid, 2001; Fettig et al., 2007; Hebertson and Jenkins, 2007). Historically, forest dynamics and patterns resulting from various fire regimes largely influenced the susceptibility of stands to bark beetle
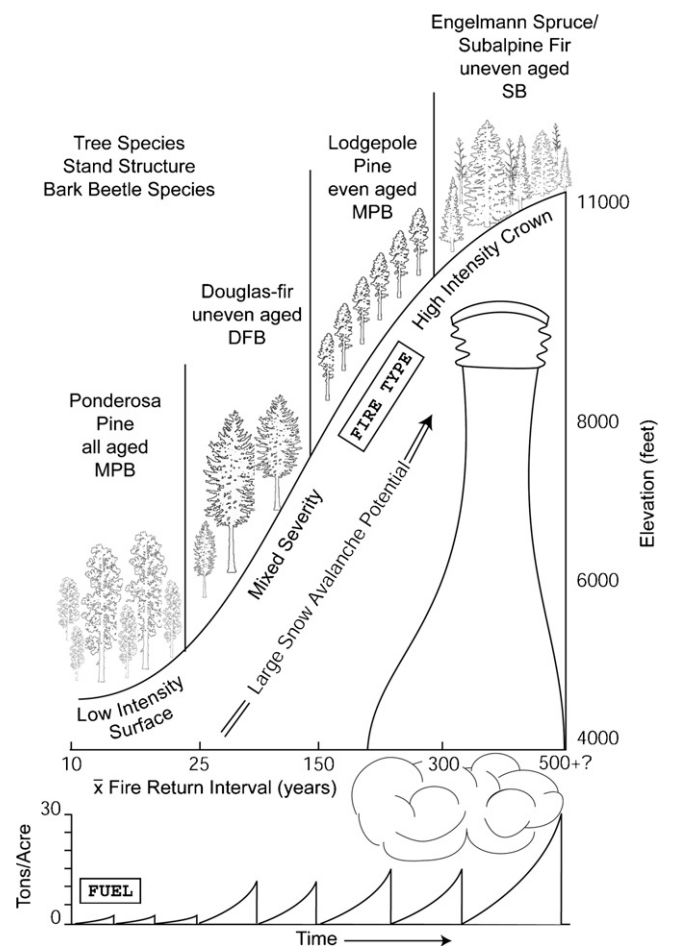

Fig. 1. Conceptual model of interrelationships between selected agents of disturbance in Intermountain Forests over varying temporal and spatial scales.

infestation. More recently, management practices including fire suppression and exclusion, livestock grazing, and timber harvesting have altered forest structure and composition (Arno, 1980; Covington et al., 1997). The specific effect of altered forest composition and structure on bark beetle epidemiology will vary with tree species, the effectiveness of fire suppression, historic fire regime and geographic parameters. One consequence of change in some forest types is that larger and more contiguous landscapes have become simultaneously susceptible to bark beetle outbreaks due to the high percentage of old, large diameter, and less vigorous host trees (Covington et al., 1997; Samman and Logan, 2000; Gallant et al., 2003). In general, factors that increase susceptible tree numbers and decrease heterogeneity increase bark beetle hazard (Amman, 1977; Schmid and Frye, 1976; Rudinsky, 1966). After reviewing extensive literature, Keane et al. (2002) concluded that fire suppression programs have resulted in atypical fuel accumulations, declines in forest health, greater risk to firefighters and increased likelihood of insect and disease epidemics. Sibold and others (2006), however, found little evidence to support the assertion that fire suppression practices have altered the historic range of variability in their study area in Rocky Mountain National Park.

In the Intermountain West, Douglas-fir beetle (*D. pseudotsugae* Hopkins), mountain pine beetle (*D. ponderosae* Hopkins), and spruce beetle (*D. rufipennis* Kirby) are capable of causing extensive mortality of host trees in relatively short periods of time (Mielke, 1950; Massey and Wygant, 1954; Amman, 1977; Schmid and Frye, 1977; Furniss and Orr, 1978; McGregor and Cole, 1985; Amman et al., 1988b; Patterson, 1991; Schmitz and Gibson, 1996; Holsten et al., 1999; Samman and Logan, 2000; Dymerski et al., 2001). This mortality alters the amount, composition and arrangement of living and dead biomass in various fuel complexes (Schmid and Frye, 1977; Cole and Amman, 1980; McGregor and Cole, 1985; Hadley and Veblen, 1993; Hadley, 1994; Jenkins et al., 1998; Hopkins, 1909; Kulakowski et al., 2003). Stands with a significant loss of overstory trees may also experience increased solar input resulting in higher temperatures and decreased relative humidity desiccating fuels and lowering fuel moisture content (Brown, 1975). It has long been assumed that such changes in fuel complexes increase the potential for fire occurrence, and fire intensity and severity (Brown, 1975; Lotan et al., 1985). Policies outlined in the National Fire Plan and the Healthy Forest Restoration Act of 2003 assume that bark beetle epidemics increase fire hazard for extended periods. It is true that any factor that alters the quantity and/or quality of the fuels complex potentially affects fire hazard. In reality, little research provides specific, quantitative descriptions of bark beetle-caused modifications to forest fuels over time. Romme et al. (2006) suggested that few scientific studies support the assertion that insect outbreaks set the stage for severe forest fires.

The majority of research to date has focused on the influences of the mountain pine beetle on fuels accumulation and subsequent wildfires in lodgepole pine (*Pinus contorta* Dougl. ex Loud) forests (Brown, 1975; Geiszler et al., 1980;

Armour, 1982; Lotan et al., 1985; Romme et al., 1986; Lynch et al., 2006). Some research has investigated the spatial and temporal relationships of historic spruce beetle outbreaks and fires in Colorado (Bebi et al., 2003; Kulakowski et al., 2003; Bigler et al., 2005; Kulakowski and Veblen, 2007). Research to examine relationships between Douglas-fir beetle-caused tree mortality and mixed severity wildfires is very limited.

Literature reviewed by Parker et al. (2006) presents an uncritical acceptance of the hypothesis that fire exclusion is the main cause of bark beetle outbreaks. The literature they cite supports the contention that outbreaks contribute to a buildup of standing and down fuels increasing fire occurrence and severity. More empirical research has been done to compare the extent of historic bark beetle outbreaks mapped using aerial detection surveys. Kulakowski and Veblen (2007), for example, used aerial detection survey maps and other spatially explicit forest disturbance data to model the effect of prior disturbances, including spruce beetle outbreaks, on severity of wildfire in Colorado subalpine forests. They found no difference in fire severity among stands with current outbreaks and endemic stands. Using similar data layers, Lynch et al. (2006) found that the odds of burning increased in mountain pine beetle-killed stands in one of two outbreaks studied following the Yellowstone fires of 1988. No significant association was detected in the second mountain pine beetle outbreak. Their data set was limited, however, because it did not consider the historic stand conditions, and the temporal and spatial variability of the beetle-influenced fuels stratum. Lynch et al. (2006) indicated that the accuracy of aerial detection survey data was more than sufficient for their analysis. Kulakowski and Veblen (2007) suggested, however, that errors inherent in aerial detection surveys of bark beetle activity limit conclusions regarding the influence of bark beetle outbreaks on fire extent and severity. They indicated that future studies could benefit from methods that incorporate ground surveys to better measure the extent of current and past outbreak events. These studies are further confounded by consideration of factors such as geography, climate and disturbance history which make isolation of the cause and effect relationship of fire/fuels/beetle factors difficult. Additionally, they do not consider other important factors that affect the spatial distribution of burned and unburned areas such as real time fire weather.

Few studies for any species consist of fuels data collected in actual outbreak areas or fires in bark beetle-affected fuels. In this paper, we describe fire regimes in Intermountain Douglas-fir (*Pseudotsuga menziesii* var. *glauca* (Beissn.) Franco), lodgepole pine, and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.)–subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) forests and their associated bark beetle species. We use actual fuels and stand data to describe changes in the fuels complex that occur in bark beetle-affected stands in all three species. We then use the fuels data to develop and describe custom fuel models that predict fire behavior in endemic, epidemic and post-epidemic stands. We compare and contrast our data with existing research relating bark beetle-caused tree mortality to fuels and potential changes in fire behavior. Finally,

we discuss the management implications for bark beetles, fuels and fire in each of the three forest types.

2. Disturbance regimes

The interplay between fuels, weather, and topography primarily determines the pattern of fire in vegetation over time, commonly called the fire regime (Pyne et al., 1996). Topography is constant while fuels and weather can vary substantially over time and space. The term 'fuels' refers to the living and dead biomass that is burnable under certain moisture conditions, which varies by forest type, time of year, and disturbance history (Brown and Davis, 1973; Albini, 1976; Pyne et al., 1996). Fuels are grouped by size classes which affect surface area to volume ratios, and the fuels' ability to respond to changing moisture conditions. Smaller fuels are able to respond to changes in atmospheric moisture more quickly than larger fuels (Anderson, 1985). Weather, in terms of its effects on fires, refers to both short-term phenomena such as wind speed and relative humidity and long-term trends such as drought and flooding (Schroeder and Buck, 1970). Throughout the paper, we follow the definitions proposed by Hardy (2005) for fire hazard and fire risk and fire severity. He considers fire risk as the probability of ignition, fire hazard as the state of the fuel, and fire severity as fire effects on wildland systems.

2.1. Douglas-fir fire regime

Intermountain Douglas-fir forests typically occupy steep, moist, northern aspects at mid elevations (1850–2500 m) between the lower elevation ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) zone and higher elevation, mixed conifer forests (Lanner, 1984; Harlow et al., 1979). This mid-elevational, vegetative gradient where Douglas-fir occurs has a mixed severity fire regime characterized by both short interval surface fires and long interval crown fires (Kilgore, 1981; Wright and Bailey, 1982). Arno (1976) found that on the Bitterroot National Forest surface fire return intervals ranged from 15 to 30 years at lower elevation sites to 50 years on moist, higher elevation sites. Crown fires occurred at intervals ranging between 35 and 60 years (Arno, 1976). This variation in fire frequency and intensity primarily depends upon specific site conditions such as stand characteristics, habitat type, climate, and fuel loads (Arno, 1980). Consequently, Douglas-fir can form pure even-aged or uneven-aged stands across a variety of habitat types (Mauk and Henderson, 1984). In uneven-aged stands, common associates include subalpine and white fir (*Abies concolor* (Gord. and Glend.) Lindl. ex Hildebr.), quaking aspen (*Populus tremuloides* Michx.), ponderosa pine, limber pine (*Pinus flexilis* James), and spruce (*Picea* sp.). Where fires have occurred less frequently, individual Douglas-firs might be up to 150 cm at diameter at breast height (dbh) (Lanner, 1984).

During the interval between stand-replacing fires, Douglas-fir stands are commonly subjected to wind, snow avalanches, and Douglas-fir beetle infestations among other disturbances. The interplay between disturbance agents often creates a mosaic of even-aged patches across the landscape. As the age of

an individual patch increases, it becomes increasingly susceptible to Douglas-fir beetle activity especially where recently fallen trees provide suitable habitat for colonization and brood production (Rudinsky, 1966; Hebertson and Jenkins, 2007). Coincident with the lengthening fire-free interval is an increase in understory conifer establishment including Douglas-fir and true fir. In many Intermountain Douglas-fir stands, western spruce budworm, *Choristorneura occidentalis* Freeman (Lepidoptera: Tortricidae), activity ranges from periodic to chronic (Wulf and Cates, 1987). Budworm outbreaks are exacerbated in multistoried stands and can decrease Douglas-fir resistance to bark beetles and contribute to fuel changes (Fredricks and Jenkins, 1988). Douglas-fir beetle populations can also increase following low-intensity surface fires where the insects are more likely to colonize fire-damaged trees (Cunningham et al., 2005; Parker et al., 2006). As beetle-caused tree mortality increases, the quantity and quality of the fuels complex changes, increasing the potential fireline intensity of subsequent fires. The increased intensity in older patches will perpetuate the age class mosaic into future fire rotations.

2.2. Lodgepole pine fire regime

Lodgepole pine forests in western North America typify fire dependent ecosystems (Agee, 1993). These forests become increasingly flammable with time and traits including cone serotiny, volatile foliage, thin bark, high reproductive capacity, and seedling survival evolved in response to fire (Lotan, 1976; Schoennagel et al., 2003). With lodgepole pine forests occurring at elevations ranging between 2500 and 3000 m, however, local fire regimes can vary widely. For example, fire return intervals in the northern Rockies may range from 200 years or more, to as few as 20 years with combinations of low-intensity surface fires and high-intensity crown fires (Arno, 1980; Lotan et al., 1985; Barrett et al., 1997). Frequent and less intense surface fires are more common in lower elevation forests characterized by dry summers (Arno, 1980). At higher elevations, fire return intervals lengthen and the potential for high-intensity crown fires increases due to the cooler and moister climate conditions (Arno, 1980; Romme, 1982). In northern Utah, fire-free intervals ranged from 12 to 122 years during pre-settlement times (Wadleigh and Jenkins, 1996). Crown fire development in lodgepole pine stands depends upon sufficient heat release from surface fuels, the height of the tree crowns above the ground, and current weather conditions (Bradley et al., 1992). In turn, these factors are dependent upon specific stand conditions, locations, and fuel loadings. Many even-aged lodgepole pine stands owe their origin to stand-replacing fires (Lotan et al., 1985).

Following stand-replacing fires, lodgepole pine quickly colonizes burned sites (Lotan, 1976). After approximately 20 years, subalpine fir begins to establish beneath lodgepole pine. During this period of stand development, the fuels complex in some lodgepole pine stands is conducive to low-intensity surface fire (Jenkins et al., 1998). The occurrence and influence of surface fire on the lodgepole pine fire regime varies. Wadleigh and Jenkins (1996) reported that surface fire was

common during the stand-replacing fire interval in the central Rocky Mountains. Jenkins et al. (1998) evaluated the influence of surface fire on successional patterns in subalpine spruce-fir forests in northern Utah. Sibold and others (2006), however, reported that surface fire affected only 1–3% of the forested area in their study in Colorado. Where it occurs, low-intensity surface fire kills late-successional species, such as subalpine fir, with little damage to lodgepole pine. In the absence of fire, several more years are required for lodgepole pine stands to reach peak fire susceptibility (Brown, 1975). Stand-replacing fires become possible due to the lack of self-pruning, standing snags, and the presence of shade tolerant species enhancing vertical continuity. After about 60 years, lodgepole pine begins to self-prune and snags fall creating stands that are moderately fire resistant (Brown, 1975). Lodgepole pine stands of this age, however, become increasingly susceptible to mountain pine beetle infestation, particularly at mid-elevations (Amman, 1977). Geiszler et al. (1980) found that mountain pine beetles tend to initially colonize trees damaged by low-intensity surface fires. Fire-damaged tissue also provides entry courts for decay fungi that gradually weaken the tree resulting in downfall and canopy openings (Gara, 1988). Lodgepole pine seedlings, understory shrubs and herbaceous plants develop in openings (Cole and Amman, 1980). With susceptible stand conditions, populations of mountain pine beetles can reach outbreak levels causing extensive tree mortality (Gara et al., 1985). Consequently, outbreaks in association with fire serve to maintain host stands in a younger, more vigorous seral state (Geiszler et al., 1980; Gara et al., 1985; Baker and Veblen, 1990; Veblen et al., 1994; Jenkins et al., 1998). Where lodgepole pine is persistent, or in the absence of fire, mountain pine beetle outbreaks create uneven-aged stand conditions by removing pockets of larger trees. Shrubs and herbaceous species invade the openings, but over time, diminish as lodgepole pine saplings assume dominance. New gaps seeded in by lodgepole pine result in different age classes of trees and multistoried stands (Cole and Amman, 1980). Without fire, and depending on site conditions, major reductions in lodgepole pine allow for the release of understory Engelmann spruce and subalpine fir at higher elevations and convert the stand to Douglas-fir at lower elevations (Roe and Amman, 1970; Bradley et al., 1992; Jenkins et al., 1998).

2.3. Engelmann spruce/subalpine fir fire regime

A high-severity fire regime typically characterizes Intermountain spruce-fir forests (Agee, 1993; Edmonds et al., 2000). Elevations exceed 2400 m in the spruce-fir zone and climates are generally cool and moist in summer (Alexander, 1987). These conditions primarily influence the flammability of fine fuels reducing ignition risk and initial fire spread. Fire weather conditions conducive to extreme fire behavior are also limited to a few weeks in late summer during most years. As a consequence, the frequency of wildfires in spruce-fir forests is low with return intervals often greater than 100 years (Schmid and Frye, 1977; Wright and Bailey, 1982; Alexander, 1987; Baker and Veblen, 1990; Veblen et al., 1994; Jenkins et al.,

1998). For example, Jenkins et al. (1998) determined that 200–400 years are needed for a spruce-fir climax forest to develop in northern Utah, which generally agrees with other fire history studies in the northern Rockies (Arno, 1980). Environments for tree establishment within the spruce-fir zone are marginal (Daniel and Schmidt, 1972; Paden et al., 1978; Knapp and Smith, 1982; Alexander, 1987). Survival is especially low on burned sites where extreme heat, frost, and browsing kill unprotected Engelmann spruce seedlings. Consequently, grasses and shrubs may dominate post-fire communities for more than a century (Edmonds et al., 2000).

The fuels complex that develops in spruce-fir forests between major wildfires creates the potential for crown fires rather than surface fires (Taylor and Fonda, 1990). Trees with tightly arranged crowns and low growing branches that are often covered with lichens facilitate crown fire spread. Under dense, closed canopy conditions, spruce-fir stands have, on average sparse understories and a compact litter layer comprised of short, fine needles that limit the potential for low-intensity surface fires (Bradley et al., 1992; Swetnam and Baisan, 1996). Therefore, many even-aged spruce-fir forests likely resulted from stand-replacing fires (Loope and Gruell, 1973).

During the relatively long period between major fire events, spruce beetle-caused tree mortality has an important influence on the vegetative dynamics and patterns of spruce-fir forests. Following severe outbreaks, the return to climax spruce-dominated communities may require up to several hundred years (Schmid and Frye, 1977; Baker and Veblen, 1990; Veblen et al., 1991a; Veblen et al., 1994; Jenkins et al., 1998). With the death of overstory spruce, advanced regeneration releases. The majority of advanced regeneration, particularly on dry sites at lower elevations is subalpine fir. The relatively small proportion of advanced spruce regeneration in the understory of mature stands can be attributed to the inability of spruce seedlings to establish in heavy duff layers and several species of fungi that specifically infect spruce seeds (Daniel and Schmidt, 1972; Paden et al., 1978; Knapp and Smith, 1982; Alexander, 1987; Aplet et al., 1988). Consequently, subalpine-fir typically dominates stands for about 200 years until trees succumb to the effects of disease (Veblen, 1986a,b; Aplet et al., 1988). Dead and windthrown subalpine fir create canopy gaps and upturned mineral soil that provide suitable habitat for spruce seedling establishment (Shea, 1985; Aplet et al., 1988). Several more decades may elapse before mature spruce dominates the main canopy once again. Veblen et al. (1994) reported that spruce-fir stands affected by the 1940s outbreak on the White River Plateau, Colorado had a scarcity of spruce older than 140 years.

3. Bark beetle ecology

3.1. Douglas-fir beetle

The Douglas-fir beetle occurs throughout the range of its primary host, Douglas-fir (Schmitz and Gibson, 1996). Endemic populations of beetles typically infest downed host material and large diameter trees weakened by drought, fire,

defoliation, and root diseases (Furniss, 1965; Coulson and Witter, 1984; Christiansen et al., 1987; Fredricks and Jenkins, 1988; Jenkins, 1990; Stark, 1993; Cunningham et al., 2005). Populations have been known to reach outbreak levels following disturbances that produce an abundance of green host material (Lejeune et al., 1961; Hadley and Veblen, 1993). Outbreaks also appear to coincide with prolonged periods of drought.

Hadley (1994) found that the vulnerability of post-fire stands to bark beetle attack was related to both aspect and topography. Specifically, it appears that stands of Douglas-fir on north-facing slopes are more susceptible to beetle attack. This may be due to the effects of fire suppression resulting in higher tree densities and a more suitable distribution of large diameter trees (Hadley and Veblen, 1993; Hadley, 1994). In the Intermountain region, Douglas-fir beetle outbreaks are site specific depending on the availability of susceptible hosts and mortality centers may be confined to small groups of trees. Within highly susceptible stands or landscapes, beetles may kill large groups of trees (exceeding 100 trees) and result in 60–80% mortality of the Douglas-fir over 30 cm in diameter (Furniss and Orr, 1978; Amman and Ryan, 1991; Patterson, 1991; Schmitz and Gibson, 1996). Larger diameter trees have thicker phloem necessary for greater reproductive success (Lessard and Schmid, 1990; Amman and Ryan, 1991). Bark beetles generally only attack smaller diameter trees during epidemics.

Stands considered "high hazard" for a Douglas-fir beetle outbreak have average Douglas-fir diameters greater than 23 cm dbh, average age of Douglas-fir in the stand greater than 120 years, stocking densities exceeding 28 m^2/ha, and more than a 50% Douglas-fir component. Outbreaks of Douglas-fir beetles typically last 2–4 years, but periods of unusually dry weather (Schmitz and Gibson, 1996) or budworm defoliation (Fredericks and Jenkins, 1988) may prolong activity in certain areas.

3.2. Mountain pine beetle

The mountain pine beetle is the most aggressive bark beetle in western North America. In the Intermountain West, this insect primarily infests and kills lodgepole pine, ponderosa pine, white bark pine (*Pinus albicaulis* Engelm.), limber pine, and other western white pines throughout their range. During severe outbreaks, mountain pine beetles may also attack Douglas-fir, true firs, and spruce, although successful brood production in these species is rare (Furniss and Carolin, 1980). When populations are low, beetles infest weakened and injured trees. Drought stress often contributes to the initiation of mountain pine beetle outbreaks (Mattson and Haack, 1987; Goheen and Hansen, 1993). Mountain pine beetle spread occurs most rapidly in old, dense stands comprised of a majority of large lodgepole pines. The most susceptible stands have trees over 80 years old, average tree diameters greater than 20 cm dbh, an average phloem thickness greater than 0.254 cm, over 18 m^2/ha mean basal area, and stand density index between 140 and 245 (Cole and Amman, 1980; Berryman, 1982; Amman et al., 1985; McGregor and Cole, 1985; Safranyik, 1989; Anhold et al., 1996). During outbreaks, beetles infest the older,

large diameter trees first, and eventually kill smaller trees as populations build (McGregor and Cole, 1985). Between 1998 and 2005, mountain pine beetles killed approximately 27 million lodgepole over 149,734 ha in the Sawtooth National Recreation Area, Idaho (USDA FS, 2006). Epidemics usually collapse with the loss of all or most large trees.

3.3. Spruce beetle

The spruce beetle is the most important biotic disturbance agent affecting post-fire, spruce-fir forests in the Intermountain West (Veblen et al., 1994; Jenkins et al., 1998). Engelmann spruce serves as the principle host for spruce beetles, although they may attack Colorado blue spruce (*Picea pungens* Engelm.) and lodgepole pine during epidemics (Miller, 1970; Schmid and Frye, 1977). Endemic populations of spruce beetles typically reproduce within newly fallen trees (Schmid, 1981). Occasionally they will also attack large, overmature spruce, killing trees individually or in small groups creating canopy gaps (Veblen et al., 1991b; Goyer et al., 1998). Local populations can build to epidemic levels following stand disturbances that produce an abundance of suitable host material (Schmid and Frye, 1977; Jenkins et al., 1998; Hebertson and Jenkins, 2007). As competition for food intensifies, beetles may then attack standing live trees. Stand conditions, host vigor, climate, and existing population levels also influence the initiation of outbreaks and the spread of spruce beetles through living stands (Schmid and Frye, 1976; Schmid, 1981; Hard, 1985; Werner and Holsten, 1985; Paine et al., 1997; Jenkins et al., 1998; Holsten et al., 1999; Hansen et al., 2001; Hebertson and Jenkins, 2007). Dense stands comprised of over 65% large diameter spruce (greater than 40 cm dbh) are most susceptible to spruce beetle attack. Recent spruce beetle outbreaks in Utah have resulted in tree mortality exceeding 1.5 million spruce trees and the loss of over 90% of the large spruce component (Dymerski et al., 2001; Keyes et al., 2003; USDA FS, 2006).

4. Bark beetle impacts on forest fuels

Fig. 2a–c shows the influence of bark beetle mortality on various fuels characteristics during the period between bark beetle outbreaks. We refer to this period of time as the bark beetle rotation for Engelmann spruce, Douglas-fir and lodgepole pine. Each of the seven graphs were derived from fuels data collected in stands with endemic, epidemic and post-epidemic populations of bark beetles in each of the three conifer-dominated forests (Page and Jenkins, 2007a). The data were collected at discrete points in time and have been extrapolated to conceptualize continuous fuel changes over the beetle rotation. Fuels are grouped into live (shrubs and forbs), dead and down woody (0–7.62 cm in diameter and greater than 7.62 cm in diameter), litter (recently cast needles), and duff (partially decomposed litter and twigs).

Bark beetle populations and associated tree mortality vary widely with tree and stand conditions and short-term influences on tree vigor. For purposes of describing the influence of bark beetles on forest fuels, the focus is tree mortality rather than
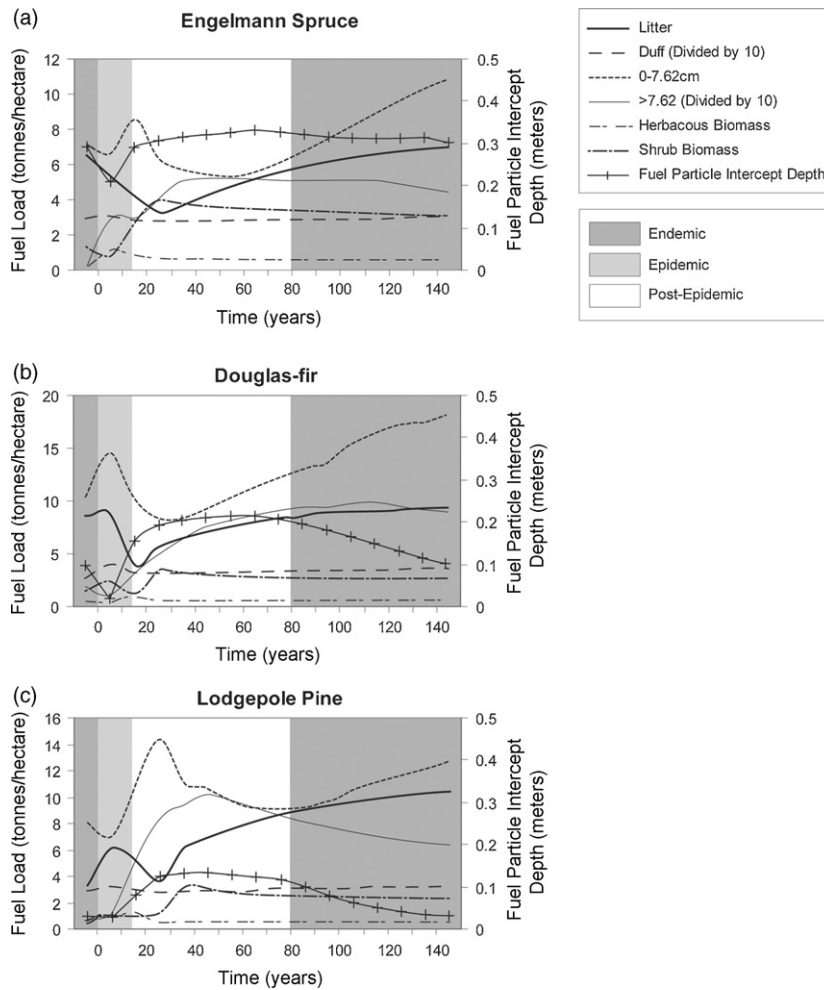
Fig. 2. The influence of bark beetle mortality on various fuels characteristics during the period between bark beetle outbreaks in (a) Engelmann spruce, (b) Douglas-fir and (c) lodgepole pine.

beetle population level. We define endemic stands as those where tree mortality is limited to weakened trees and is relatively constant from 1 year to the next in a given stand. In these stands, the mortality of susceptible trees is generally less than 5 trees/ha. Epidemic stands have increasing beetle populations and a consequent increase in tree mortality that may exceed 80% of susceptible host trees. Post-epidemic stands are those at least 5 years beyond the epidemic phase. Time on the horizontal axis represents one bark beetle rotation for a stand, and is arbitrarily set at 100 years. In reality, the time required for beetles to kill the majority of susceptible host trees in stands at the landscape scale may be decades to centuries. The duration of individual epidemics varies with the availability of susceptible host type across the landscape, climate, and the abundance predators and parasites, but generally lasts 3–10 years. The post-epidemic phase will last until stand conditions again become susceptible to bark beetle attack which may be decades to centuries depending on bark beetle species.

These data show that fine woody fuels (0–7.62 cm in diameter) increase in epidemic stands of all three conifer types reaching their highest levels immediately following the epidemic. Litter increases in epidemic stands as needles are cast from dead and dying trees. Fine woody fuel and litter decrease dramatically in stands following the epidemic and gradually begin to increase again in post-epidemic stands as regeneration occupies the site and contributes to reestablishing background levels of fine woody and litter fuel loads. Duff loads remain relatively constant in epidemic stands and gradually increase with time following snag failure and subsequent decomposition. Large down woody fuel loads (greater than 7.62 cm) are relatively low in epidemic stands and increase with time as beetle-killed trees begin to fall. Few studies provide reliable estimates of fall rates for dead trees in Intermountain forests. Fall rates likely vary widely due to site characteristics, incidence of disease, and tree species. Consequently, these graphs assume constant, uniform fall and decomposition rates which result in the greatest accumulation of large woody fuels in post-epidemic stands over time. Fuel heights increase as dead trees fall. Once the majority of dead trees are on the ground, fuel heights begin to decrease as a result of decomposition and compaction. Over time, these processes result in relatively uniform fuel bed depths for each species through the remainder of the post-epidemic phase.

The death of overstory trees during bark beetle outbreaks increases the availability of water and sunlight for understory plant species. Herbaceous fuel loads are greatest during epidemic phase. Shrub fuels begin to increase during epidemics and peak in early post-epidemic stands. Shrub species gradually out-compete herbaceous plants and comprise the majority of live fuels in early to midpost-epidemic stands. Post-epidemic shrub response is most dramatic in mesic spruce and Douglas-fir sites compared to dry pine sites where shrubs are less common and more sparse in general (Brown and Bevins, 1986). In all forest types, conifer regeneration gradually replaces shrub species in post-epidemic stands over time.

In less comprehensive studies, others have described the changes in fuels in bark beetle-affected conifer stands. Gara et al. (1985), Amman (1991), and Schmid and Amman (1992) largely built upon the relationships previously established by Brown (1975) and Lotan et al. (1985). Studies by Armour (1982) and Romme et al. (1986) evaluated fuels accumulations due to the mountain pine beetle in lodgepole pine stands in Montana and Wyoming. Romme et al. (1986) studied changes to fuels and fire hazard from stands attacked 1–20 years prior. They found that there were slight increases in needle litter for six years following the outbreak and that there were steady and substantial increases in large woody fuels for at least 20 years following the outbreak. Armour (1982) presented models to predict fuel accumulation by habitat type.

Spruce beetle mortality has also been reported to alter dead fuel loads in affected stands over time (Baker and Veblen, 1990; Jenkins et al., 1998; Schulz, 2003). In a study conducted in Alaska, Schulz (2003) found significant increases in fuel heights, fine fuels, and sound large fuel between 1987 and 2000 across several different forest types affected by a spruce beetle outbreak that peaked in the 1990s. Slow rates of decomposition at high elevations allow heavy fuel loads to persist decades after spruce beetle outbreak (Cahill, 1977; Bradley et al., 1992). For example, 30 years following the White River outbreak in Colorado, Cahill (1977) calculated an average of 55 tons/acre of dead, down, woody fuels in more than 40% of beetle-killed stands.

5. Fire behavior in bark beetle-affected conifer fuels

5.1. Fire behavior

Section 4 describes the general effects of bark beetles on conifer fuels complexes over the course of a beetle rotation. The changes in fuels over time create periods where the potential for high intensity and/or severe fires increases or decreases. While fire in fuels will largely determine ecological impact, the likelihood of any fire is controlled by real time fire weather. In general, the potential for suitable fire weather decreases with increasing elevation and becomes less likely at the highest elevations in the subalpine zone (Bessie and Johnson, 1995).

Fire behavior is described in terms of its rate of spread, flame length, and intensity and can be predicted if appropriate inputs of fuels, moisture contents, slope, and wind speed are available (Rothermel, 1983). Although fire behavior inputs are readily

measured and outputs can be accurately derived, the relative importance of inputs on ecosystem effects is poorly understood (Omi and Martinson, 2004).

Fig. 3a–c shows predictions of important fire behavior outputs based on fuels data collected and discussed in Fig. 2a–c for Engelmann spruce, Douglas-fir and lodgepole pine. Each fire behavior prediction was based on custom fuel models for bark beetle-affected fuels and a range of mid-flame wind speeds, slope steepness, and fuel moistures using methods described in Page and Jenkins (2007b). To better understand the differences in fire behavior caused by the fuels characteristics the predictions were made using a 0% slope. To capture a range of fuel moisture conditions, three sets of fuel moistures were used; normal summer, drought summer, and late summer severe drought (Rothermel, 1991). Fire predictions also incorporated a range of mid-flame wind speeds. The data presented in Fig. 3a–c can be considered to represent fire behavior outputs under average late summer fire weather conditions.

5.2. Fire behavior in epidemic stands

These predictions show that potential fire behavior outputs of spread rate, fireline intensity and flame length differ in endemic, epidemic, and post-epidemic conditions in the three conifer types considered. During bark beetle epidemics, rates of surface fire spread and fireline intensity increased but the
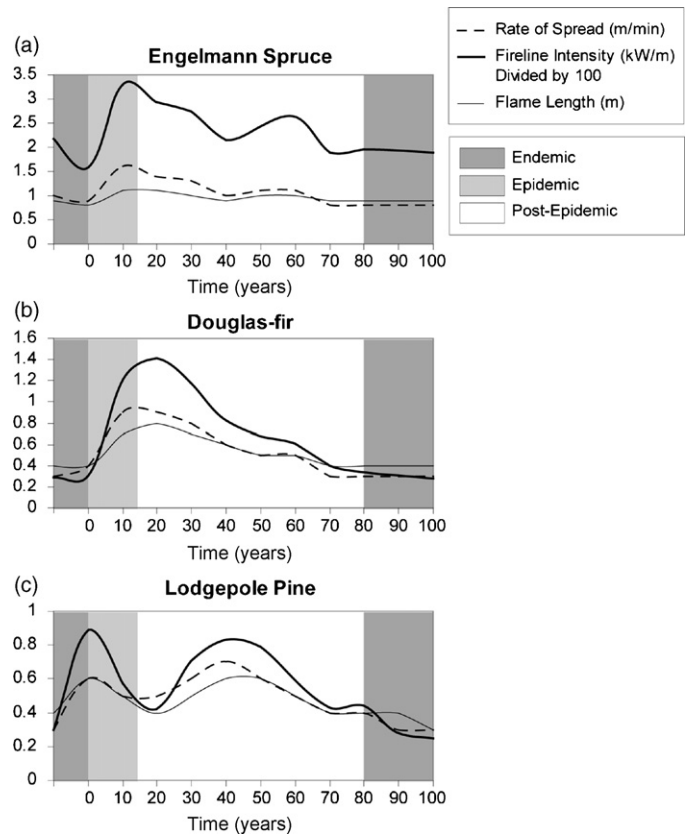

Fig. 3. Predictions of important fire behavior outputs based on fuels data collected and discussed in Fig. 2a–c for (a) Engelmann spruce, (b) Douglas-fir and (c) lodgepole pine.

magnitude of the increase varied substantially. For example, in selected lodgepole pine stands increases in average potential rate of spread ranged from approximately 3 to 6 times when compared with endemic stands (Page and Jenkins, 2007b). This variation in rate of spread and fireline intensity was most dependent upon the net gain in litter and fine fuels with higher loadings corresponding to higher rates of spread. These increases in rates of spread contribute to greater chances for large fire development.

These results differ from those of Schmid and Amman (1992) who suggested that fire hazard in Douglas-fir beetle infested stands is highest during the two years after infestation while dead needles remain on the trees and that fire hazard begins to decrease as needles fall to the forest floor. Romme et al. (1986) found that fire hazard or ''destructive fire'' risk may have actually decreased due to the mountain pine beetle epidemic they studied.

5.3. Fire behavior in early post-epidemic stands

The data in Fig. 3a–c suggest that the initial increase in spread rate and fireline intensity decreases shortly after the epidemic phase and increases decades later as snags fall and regeneration occupies the site. This result is in contrast to work by Brown (1975), Lotan et al. (1985), and Schmid and Amman (1992) who determined that fire risk was always growing beyond pre-outbreak levels after the initial drop of needles in the crowns of infested trees. Armour (1982) obtained similar results where fuel loading increased rapidly, then gradually leveled off or decreased in mountain pine beetle infested stands in northwestern Montana. His attempts to model fuel succession, however, were largely limited due to the large variation in fuel loading that he encountered.

5.4. Fire behavior in post-epidemic stands

Fig. 3a–c also shows that rates of fire spread and fireline intensities in the post-epidemic stands were greater than the endemic stands. These increases, however, were not due to differences in fuel loadings, but to greater mid-flame wind speeds. The lack of dominant overstory trees in the post-epidemic stands decreased the sheltering effect of vegetation, which allowed more wind to reach the forest floor. The greater surface wind speeds increased the fire's ability to transfer heat through convection and radiation, thereby increasing rates of fire spread and the rate of heat release at the flaming front (Rothermel, 1991). However, the increase caused by the drier fuels was negligible compared to the effect of the wind, with increases ranging from 0 to a maximum of 0.4 m/min, under extreme drought fuel moistures. These results are consistent with other studies that found beetle-caused tree mortality opens up the main canopy and intensifies the seasonal desiccation of understory fuels that in turn cause associated increases in fire behavior (Brown, 1975; Furyaev et al., 1983; Knight, 1987).

These effects were most evident in Engelmann spruce (Fig. 3a) and lodgepole pine forests (Fig. 3c) 40–60 years after beetle epidemics when large fuels resulting from fallen beetle-

killed trees begin to accumulate. Ten or more years may lapse before standing dead trees begin to fall over (Ohmann, 2002), and no previous research has reliably demonstrated how fallen snags contribute to fire hazard. However, the highest fire hazard is presumed to exist in post-epidemic stands due to heavy fuel loading by large diameter materials coupled with the presence of regeneration and extreme fire weather conditions (Page and Jenkins, 2007a). Topography and wind patterns also contribute to more erratic fire behavior within the spruce-fir zone (Edmonds et al., 2000) and standing snags can be potential lightning rods that may increase the frequency of lightning ignitions (Barrows, 1951; Schmid and Hinds, 1974). Romme et al. (2006) correctly stated that the influence of bark beetle mortality likely depends on the time since outbreak. They presented a theoretical interpretation of the influence of bark beetle mortality on fire behavior that is supported, in part, by the data we present in this paper.

5.5. Crown fire relationships

The potential transition from a surface fire to a passive crown is determined by a critical intensity of the surface fire and the crown base height that provides a vertically continuous fuel stratum of sufficiently low fuel moisture. This phenomenon is commonly called torching in ladder fuels. The potential transition from passive to active crown fire is determined by canopy closure or canopy bulk density, foliar moisture content, and effective wind speed. Rothermel (1991) developed models that defined the above variables and used foliar moisture inputs from Van Wagner (1977) to predict crown fire initiation and spread. We applied the crown fire spread model to the fuels data to predict crown fire development and spread.

We found dead aerial fuels in the current epidemic stands will increase the likelihood of surface fires transitioning into canopy fuels for a given surface fire intensity more than in endemic stands. Current fire prediction models, however, are based only on live fuels and do not incorporate the effect of dead aerial fuels in the overstory, which can comprise as much as 30% of the total foliage in current epidemic stands (Page and Jenkins, 2007a). Differences in chemical composition of live versus dead foliage, such as volatile oils, and flammability may also influence crown fire spread (Van Wagner, 1974). The crown fire model utilizes live foliar moisture content, greater than 70% of oven dry weight, and crown base height to model crown fire initiation. However, fuel moistures for dead fuels are generally less than 30% of oven dry weight and may be much lower depending upon specific environmental conditions. Thus, the model we used may be inadequate to predict crown fire potential in the current epidemic stands where a range of live, dead and dying foliage contribute to crown fire initiation and spread. Nevertheless, several studies have demonstrated that dead needles with moisture contents as low as 7.7% of oven dry weight will ignite with significantly lower temperatures compared to live fuels, increasing the potential for passive crown fire development (Stockstad, 1975; Xanthopoulos and Wakimoto, 1993).

In epidemic stands, a range of foliar moisture will be present, represented by trees attacked in previous years with red needles, currently attacked trees retaining green needles, and unattacked green-needled trees. The relative proportion of trees in each category will change during the course of the epidemic and the actual stand based foliar condition of moisture and volatile oil content will be unknown. We assume, however, that the range of foliar condition will increase the crown fire potential through decreased ignition temperature while retaining volatile flammability. Brown (1975) and Knight (1987) determined that dead needles retained on the trees dramatically increase the total amount and vertical continuity of fine, dry fuels. It is reasonable that stands experiencing current epidemics will be the most likely to exhibit crown fires during periods of extreme fire weather.

The heavy amounts of large diameter fuels in the post-epidemic stands also produced increased durations of flaming and smoldering combustion along with higher total heat release per unit area. Rothermel (1991) recognized that the combustion of large sound fuels can have significant impacts on crown fire development. Predicted increases in fireline intensity, in conjunction with low crown base heights of regeneration, also contributed to greater potential for crown fire initiation in the post-epidemic stands with surface fires transitioning to involve crown fuels under drought and extreme drought fuel moistures. However, the potential for active crown fire spread was less in the post-epidemic stands due to a lack of aerial fuel continuity.

There has been considerable debate regarding the relative influence of fuels and fire weather on fire development and spread. Fires begin and end with fuels. As state previously, any factor that affects the quality and/or quantity of fuels in time and space has the potential to alter fire behavior. It is fuel moisture content in fine fuels that determines fire ignition and spread and fuel moisture in large fuels that contributes to fire severity. Hence, weather, climate and seasonality mediate their influence on fire behavior via effects on fuel moisture content. Under appropriate fire weather conditions, extreme fire behavior is possible across a wide range of fuel complexes.

Bessie and Johnson (1995) and Johnson et al. (2001) indicated that under extreme fire weather conditions intense crown fires will develop with little regard to surface fuel loading. This is supported by observations during the Yellowstone fires of 1988 in which many of the fires burned through all age classes lodgepole pine stands and fuel conditions with the total daily area burned closely linked to 1000 h fuel moistures. The extremely low 1000 h fuel moistures recorded during the Yellowstone fires suggested that long-term drought was important for the extreme fire behavior observed (Renkin and Despain, 1992; Turner et al., 1994).

In spruce-fir forests in Colorado, Bebi et al. (2003) and Kulakowski et al. (2003) found that when compared to unaffected stands, fuel accumulations due to past spruce beetle outbreaks did not increase fire density. Their observations suggested that heavy fuel accumulations caused by the spruce beetle may be of little relative importance when fire weather conditions are suitable for the development of crown fires.

Regarding fuels, Arno (1980) suggested that site-specific differences such as fuel loading can mean the difference between a surface fire or crown fire developing in a Douglas-fir forest. Similarly, Stocks (1987) found that in spruce budworm-killed forests, after an initial peak in surface fuel loads and fire potential, fire potential gradually decreased as the surface fuels decomposed and understory vegetation flourished.

5.6. Ecological effects of extreme fire behavior

The timing of high-intensity crown fires may influence post-fire succession and be advantageous to species such as lodgepole pine which possesses specific adaptations to high fire intensity. Such fires create good seedbed conditions and cause serotinous cones on lodgepole pine to open resulting in dense stands when seed is abundant. Where lodgepole pine is seral, fire serves to perpetuate lodgepole pine as the dominant species. Thus, in order for lodgepole pine to maintain dominance it is believed that lodgepole pine and the mountain pine beetle have co-evolved so that the fuel buildup and resultant fires favor lodgepole regeneration (Brown, 1975; Lotan et al., 1985). When available fuels are limited, fires have a tendency to creep and with increasing fuel loads increase in rates of spread and intensity (Agee, 1981). It is plausible that mountain pine beetle outbreaks contribute to pine forest flammability and enhance the opportunity for serotinous populations of lodgepole pine to remain on sites indefinitely.

Despain (1990), however, put forth the hypothesis that mountain pine beetle outbreaks actually decrease the risk of crown fires because they decrease overstory fuel continuity and cause changes to the structure of the forest such that fires are inhibited. He noted examples in Yellowstone National Park where fires went around previous mountain pine beetle outbreaks, although admitted that his observations were merely anecdotal in nature and not specifically studied. Despain (1990) concluded that further work should be done to better understand the nature of the beetle–fuel–fire interaction. Romme et al. (1986) also concluded that due to decreases in continuity of canopy fuels and accumulation of fine forest fuels was small that the risk of a high-intensity fire between years 2 and 20 (after outbreak) may be lower.

While the accumulation of fuels and the presence of advanced regeneration in the understory of spruce-fir forest may increase a stand's vulnerability to fire, particularly under extreme fire weather conditions (Heinselman, 1973; Despain and Sellers, 1977; Arno, 1980; Veblen et al., 1991b, 1994; Jenkins et al., 1998; McCullough et al., 1998; Bessie and Johnson, 1995; Schulz, 1995), the severity of these fires is based on their physical characters (rate of spread, flame length, intensity) and the fire adaptations of the vegetation. Engelmann spruce and subalpine fir both have thin, resinous bark and shallow root systems making them very sensitive to fire (Alexander, 1987). With the potential for high-intensity fires initiated at lower elevations to burn up through higher elevation stands, hazardous fuels accumulations pose more of a concern for unnatural levels of damage. These increased durations of flaming and smoldering combustion in post-epidemic stands

should increase fire severity and have important implications to post-fire effects, causing severe damage to living vegetation and higher mortality rates (Whelan, 1985).

6. Bark beetle management strategies and implications for fuels and fire behavior

Land managers often employ various prevention, suppression and restoration strategies to mitigate potentially adverse impacts resulting from bark beetle outbreaks. Prevention strategies often utilize silvicultural practices including thinning to modify stand conditions favorable to insect and disease agents and should occur before populations reach unmanageable levels. After the initiation of an outbreak, management options become increasingly limited and delays in treatment greatly reduce opportunities for success (Fettig et al., 2007). By enhancing stand diversity and resiliency, prevention strategies can help avoid unacceptable losses of valuable resources, maintain or enhance resource objectives, and maximize revenue in the long term (Schmid and Frye, 1977; Cahill, 1978; Cole et al., 1983; McGregor et al., 1987; Amman et al., 1988a,b; Fettig et al., 2007; Schowalter et al., 1992; Anhold et al., 1996; Schmitz and Gibson, 1996; Holsten et al., 1999). Suppression strategies are usually implemented with the detection of building bark beetle populations. These strategies include sanitation and salvage treatments, insecticide applications and the use of pheromones to reduce local beetle populations and protect high value trees from bark beetle attack in recreation sites. When implemented during the initial stages of an outbreak, suppression activities can reduce population levels and the distribution and extent of bark beetle-caused tree mortality. Treatments associated with suppression alternatives, however, are usually limited, often occurring at small scales. Because treatment alternatives do not modify stand conditions at large scales, the benefits are often short term. Restoration activities attempt to reestablish vegetation and promote the long-term resiliency of forests to insect and disease outbreaks or attacks.

Reducing the accumulation of hazardous fuels and/or the potential for extreme fire behavior resulting from bark beetle mortality is often cited as a rationale for implementing bark beetle management treatments (Schott, 1994; Gorte, 1996; USDA Forest Service, 1998). Romme et al. (2006) suggested that there is little ecological justification in removing insect-killed trees. Additionally, slash produced during bark beetle treatments and post-harvest blow down will result in fuel complexes different from those in either bark beetle-killed or uninfested stands. Conventional slash fuel models (Rothermel, 1983) used to estimate fire behavior in stands treated for bark beetles suggest increased potential for ignition and alteration of fire behavior depending on the nature of the treatment (Turner et al., 1994; DellaSala et al., 1995).

To determine how fuels might differ in spruce-fir forests managed for spruce beetle versus unmanaged forests, we measured fuels in thinned and control stands on the Fishlake National Forest in south central Utah (Jenkins et al., 2005). Moderately thinned stands had residual basal areas ranging

between 23 and 28.4 m^2/ha with average spruce diameters of 30.5 cm dbh. Heavily thinned stands had residual basal areas of 13.8–18.3 m^2/ha with average spruce diameters of 25.4 cm dbh. Fuels measurements were then used to create custom fuel models to compare potential fire behavior among the treatments. Fine fuels (1, 10, and 100 h size classes) increased in treated stands with the greatest amount of total fuel accumulating in the heavily thinned plots. Moderately thinned plots generally had the lowest fuel depths. Heavily thinned plots had a greater average fuel depth compared to the control plot in some stands. The 1000 h fuels were quite variable from treatment to treatment in the areas sampled. As expected, the amount of available crown fuel significantly dropped in each of the thinning treatments. Crown bulk densities were also significantly lower for the thinning treatments.

Predicted fire behavior was greatest in heavily thinned stands where the increase in fuel depth, the amount of fuel, along with a higher mid-flame wind speed increased potential fire behavior. In other areas, the greatest fire behavior was in the control plots associated with the highest wind speed. Although more fuel was available in the heavily thinned stands and the mid-flame wind speed was greater, the fuel depth was less than that of the control. This was due to the mechanically compacted fuel bed which does not support combustion as well as the more loosely compacted fuels in the control stands. Andrews (1986) noted that fire behavior prediction system, BEHAVE plus, is very sensitive to fuel depth, which thus helps to explain this result.

Fire behavior in the heavily thinned plots, however, was probably under-predicted because the fuel moistures used did not take into account the effect of increased solar radiation reaching the exposed fuel bed. When compared to the heavily thinned and the control for both areas, the moderately thinned plots had intermediate fire behavior characteristics. In terms of crown fires, the control plots had the greatest potential to sustain a crown fire of some kind (i.e. lowest crown fire index).

These results demonstrate that fuels and consequent fire behavior resulting from spruce beetle management strategies can vary depending on pre-existing stand conditions and the type of treatment implemented. Other research investigating thinning treatments and influences on fire behavior in western forests obtained similar results (Graham et al., 1999; Agee and Skinner, 2005). Moderate thinnings may result in less potential for extreme behavior compared to unmanaged stands. Greater fuel depths, mid-flame wind speeds and lower fuel moistures in heavily treated stands (>60% basal area reduction) might increase potential fire behavior compared to unmanaged stands. This necessitates spruce beetle management strategies more thoroughly consider post-harvest fuel treatments to reduce surface fuels. Thinning followed by sufficient treatment of surface fuels usually outweighs changes in fire weather factors (wind speed and fuel moisture) resulting in an overall reduction in expected fire behavior (Weatherspoon, 1996). Due to similarities in bark beetle-created fuels, this consideration would apply to management strategies implemented for other bark beetle species as well.

7. Fuels and fire management strategies: implications for bark beetles

7.1. Fuels management strategies

With growing concern over the occurrence of uncharacteristic wildfires in the western United States, federal and state land management agencies have initiated aggressive fuels reduction programs to reduce the risk, extent and severity of wildfires, particularly in wildland/urban interfaces (Agee and Skinner, 2005; Glitzenstein et al., 2006). Fuels management practices often include prescribed fire or mechanical treatments that act as fire substitutes or are used to compliment prescribed burn programs (Glitzenstein et al., 2006). Implementation of these treatments, however, can injure residual trees predisposing them to attack by bark beetles (Ryan and Amman, 1994; Rippy et al., 2005; Arno, 2000; McCullough et al., 1998). Increased amounts of host material (slash and stumps) and host volatiles in treated stands can also attract bark beetles (Schmid, 1977; Amman et al., 1988b; Six et al., 2002; Fettig et al., 2006). Much recent research has consequently investigated the influence of fuels treatments on the behavior of certain bark beetle species (Six et al., 2002; Wallin et al., 2003; Fettig et al., 2006).

General recommendations to minimize bark beetle activity associated with fuels treatments resulting from these studies include (1) avoid implementing treatments concomitant with adult beetle flight periods, (2) remove slash and all susceptible non-merchantable host material from the site, or treat slash on site using pile-and-burn, lop-and-scatter, or chipping methods, (3) remove or burn any infested host material, (4) limit the diameter and length of host material, and (5) avoid thinning in stands highly susceptible to root diseases (Six et al., 2002; Steed and Wagner, 2004; Rippy et al., 2005; Fettig et al., 2006). Chipping versus lop and scatter fuel treatments may exacerbate bark beetle attacks in some bark beetle–host systems (Fettig et al., 2006). Where chipping is the preferable treatment method, Fettig et al. (2006) suggested allowing slash to dry before treatment and limiting the amount of chips around the bases of residual trees. With the inability to treat slash using one of the above methods, funnel traps baited with aggregation pheromones and placed around piles can capture a significant number of emerging adult beetles (Ross and Daterman, 1997). Regardless of treatment, it is important for land managers to assess general beetle population levels and beetle-caused tree mortality prior to treatment and monitor treated areas regularly.

7.2. Delayed tree mortality following fire

Less obvious is the fact that fire can wound trees and render them susceptible to bark beetles that in turn can kill trees altering the quantity and quality of the fuels complex. Bark beetle outbreaks following fires are not unprecedented, but neither are they certain. Beetle outbreaks following fires have been documented in the Intermountain West. Douglas-fir beetle, spruce beetle, and pine engraver beetle (*Ips pini* Say (Coleoptera: Curculionidae) outbreaks following wildfires in 1988 and 1994 became extensive, damaging, and long-lasting in parts of Montana, Idaho and Yellowstone National Park (Ryan and Amman, 1996).

Prior to implementing post-fire salvage and rehabilitation treatments land managers often need to know which trees will likely die from fire damage, which might survive fire effects but subsequently be killed by bark beetles (Parker et al., 2006), and which others may be more susceptible to fungal infections and degradation (DeNitto et al., 2000). Bark beetles frequently attack fire-injured trees although their direct effect on tree mortality remains unclear (Ryan and Reinhardt, 1988). Following the Greater Yellowstone fires in 1988 for example, Rasmussen et al. (1996) found that fire damage accounted for the majority of delayed conifer mortality, although bark beetles contributed to some mortality and ultimately changes in the mosaic of fire-killed, green trees. In other studies of those same fires, spruce beetle attacks were associated with fire-damaged Engelmann spruce, however mountain pine beetle showed little attraction to fire-damaged lodgepole pine (Ryan and Amman, 1994, 1996). Weatherby et al. (1994) observed that Douglas-fir beetles killed a significant number of large diameter Douglas-firs that had significantly less damage than fire-killed trees following the Lowman (Idaho) fire of 1989. Hood and Bentz (2007) also found that Douglas-fir beetles had a significant influence on post-fire, delayed Douglas-fir mortality, killing trees that would have otherwise survived wildfires occurring in Montana and Wyoming during 2000 and 2001.

The complex interactions between the physiological condition of host trees pre- and post-fire, the season of burn, the amount and severity of tree damage, the nature of bark beetle populations, weather and climate conditions, other biotic and abiotic factors make accurate predictions of post-fire tree mortality difficult (DeNitto et al., 2000). For individual lodgepole pine, Douglas-fir and Engelmann spruce injured during the Greater Yellowstone fires, bark beetle attacks were strongly and positively correlated with the percentage of basal area circumference killed (Rasmussen et al., 1996). Douglas-fir beetles preferentially attacked large diameter Douglas-fir trees with 60–80% bole char and crown volume scorch, and a 50–70% likelihood of mortality due to fire 1 year after a ground fire on Beaver Mountain (Utah) in 1994 (Cunningham et al., 2005). Similar levels of crown volume scorch and cambium death were associated with 95% of Douglas-fir beetle attacked trees following the Montana and Wyoming fires of 2000 and 2001, although tree size and stand density were slightly more important for attack than fire injury (Hood and Bentz, 2007).

Depending upon the time of burn, bark beetles may attack fire-injured trees until they recover, or their phloem becomes unsuitable for habitation (as long as four years post-burn) (Gibson et al., 1999; Hood and Bentz, 2007). The successful production of broods in fire-injured trees may increase the risk or likelihood of an outbreak population occurring in a stand (Bentz et al., 1993). Whether bark beetle populations build in fire-injured trees and contribute to outbreaks remains uncertain (Rasmussen et al., 1996). Outbreak populations of both spruce beetle and Douglas-fir beetle were documented following the Little Wolf Fire (Flathead National Forest, Montana) in 1994

(Gibson et al., 1999). During the 3 years after the Beaver Mountain fire, Cunningham et al. (2005) found that Douglas-fir beetle attacks shifted from larger diameter, moderately injured trees, to smaller diameter trees with less fire injury, and finally to large diameter green trees around the fire perimeter. Douglas-fir beetle populations however, did not reach outbreak levels outside of the fire boundary (Cunningham et al., 2005). While these observations suggest bark beetle populations may increase following fire, the development of bark beetle outbreaks in fire-affected and surrounding stands largely depends upon (1) the susceptibility of stands in both burned and unburned areas before and after the fire, (2) the extent and severity of fire damage, (3) bark beetle population levels in the vicinity prior to the fire (Bentz et al., 1993), and (4) weather conditions prior to and up to several years following a fire (Bentz et al., 1991; DeNitto et al., 2000).

Based on Ryan and Noste (1985), Gibson (1994), Gibson et al. (1999), DeNitto et al. (2000), Gibson (unpublished) developed a rating system to help land managers determine the potential for increased bark beetle risk and the relative susceptibility of stands to bark beetle infestation following fires of varying intensities experienced in the Northern Rocky Mountain and Intermountain regions. Land managers may use these systems to determine the relative likelihood of higher-than-normal bark beetle-caused mortality in fire-affected and nearby unburned stands.

8. Bark beetle, fuels, and fire management strategies

Policies including the National Fire Plan and the Healthy Forests Restoration Act of 2003 demonstrate that the development of resilient, fire-dependent forests is a national emphasis (Jain and Graham, 2004). This emphasis may soon encompass high-elevation subalpine forests as global climate change increases the potential for bark beetle outbreaks and intense wildfires, wildland/urban interfaces expand, and demands on important forest resources rise (Logan and Powell, 2001; Williams and Liebold, 2002; Carroll et al., 2003; Gan, 2003; Breshears et al., 2005; Brooks et al., 2006).

As this paper indicates, the interaction of western bark beetles, fuels and fire in forest systems is inherently complex and much remains unknown. This presents unique challenges for land managers, particularly in light of competing social values, multiple stakeholder interests, and uncertain management outcomes (Graham et al., 2004; Brooks et al., 2006; Snider et al., 2006). Additionally, differences in the physical environment, stand conditions, the amount and distribution of available fuels, and weather make 'one size fits all' management approaches ineffective (Graham et al., 2004). The development of management plans that give greater consideration to these interactions, however, can provide decision-makers with better guidance for meeting important resource objectives, reducing treatment costs, minimizing adverse ecological impacts, and avoiding potential controversy and litigation.

Identifying and explicitly defining the most important short and long-term resource objectives of the landscape is vital to the success of any management plan. Based on these definitions, land managers can determine the extent of landscapes requiring treatment, develop ecologically appropriate and feasible treatment alternatives, decide where and when treatments would be most efficacious and over what amount of time (Agee and Skinner, 2005). Multiple resource objectives often necessitate that management strategies prioritize landscapes for treatment. Land managers should not consider every landscape with hazardous bark beetle or fuels conditions a high priority candidate for treatment (Schoennagel et al., 2004; Agee and Skinner, 2005). For example, lower elevation drier forest types might receive higher priority for large-scale treatments than higher elevation forest types (Agee and Skinner, 2005). The immediate implementation of large-scale treatments in wildland/urban interfaces, however, may be appropriate in all forest types (Glitzenstein et al., 2006). Other treatment priorities may include landscapes with high economic investment, commercial or recreation values, or critical wildlife habitat. Prioritization should finally consider landscapes where treatments can best enhance long-term sustainability and resiliency to change (Agee and Skinner, 2005).

The mitigation of potentially adverse bark beetles and fire effects is maximized when treatments occur at landscape scales and integrate the spatial arrangement of forest types, stand conditions, treatment units and prescriptions (Graham et al., 2004; Agee and Skinner, 2005; Fettig et al., 2007). Small-scale, scattered treatments do not effectively reduce the long-term susceptibility of forests to bark beetle infestation or fragment the landscape fuels (Samman and Logan, 2000; Graham et al., 2004; Agee and Skinner, 2005; Fettig et al., 2007). Failure to reduce susceptibility across the landscape and/or address the entire bark beetle populations often results in insect spread from untreated stands into treated areas with the depletion of host resources (Fettig et al., 2007). With extreme fire weather conditions, intense fires can also overwhelm small-scale treatments (Agee and Skinner, 2005). Even with a reduction in fire behavior, treatments with substantial edge adjacent to untreated units are likely to suffer substantial tree mortality (Agee and Skinner, 2005).

9. Future research direction

Recent and on-going research has provided greater insight into the nature of bark beetle mortality, associated fuels accumulation, and fire interactions in Intermountain Douglas-fir, lodgepole pine and Engelmann spruce forest types. Similar research is needed to elucidate these relationships for these species in other regions, as well as in western ponderosa pine forest types and pinyon-juniper woodlands.

Many effects of bark beetle-caused injury and mortality on fuels and fire behavior remain uncertain. Adult bark beetles and larval feeding in conjunction with the introduction and colonization of blue-staining fungi (*Leptographium* spp.) disrupt the transport of water to the foliage of host trees (Cates and Alexander, 1982). Foliar discoloration indicative of needle death typically becomes apparent 1 year after bark

beetle attack. Dead needles have low moisture content and strongly influence the potential for crown fire initiation and rates of crown fire spread. Little is known about the moisture content of green needles on current mass attacked trees and how initial changes in moisture content might contribute to crown fire potential and intensity.

Custom fuel models used to predict fire behavior in bark beetle-affected landscapes indicate that fire severity increases as snags begin to fall. Limited research suggests that fall rates of bark beetle-killed trees are highly variable or largely unknown for many Intermountain forest types. Reliable estimates of fall rates also require further study to determine the influence of soil and site characteristics, wind, and root diseases.

Current models of bark beetle-affected fuels assume homogeneous tree mortality across the landscape. In reality, patterns (spatial arrangement and distribution) of bark beetle tree mortality and fuels accumulations exhibit considerable variability across the landscape depending on existing stand conditions, local bark beetle population levels, site characteristics, and the distribution of decay, and root disease centers (Cobb, 1989; Lundquist, 2007). The development of fire spread models in various bark beetle-affected landscapes may help land managers determine how patterns of bark beetle mortality may decrease or exacerbate (prolong) hazardous fuels (fire hazards) and consequently facilitate the identification of stands to prioritize for treatment.

There are currently several new modeling tools including Landscape Fire and Resource Management Planning Tools Project (LANDFIRE) and the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) available to fire and fuels managers to aid in fuel treatment placement and verification. Due to the nature of the data acquisition techniques in these programs, there is often inconsistency between the actual fuel conditions and conventional default fuel conditions provided by the programs (Anderson, 1982; Johnson and Peterson, 2005; Scott and Burgan, 2005). The use of site-specific fuels data inputs improves the reliability of modeled (fire behavior) projections, however the collection of these data is often costly and labor intensive. The development of tools including photo appraisal guides for bark beetle-affected fuels would facilitate collecting large amounts of actual surface fuels data for modeling purposes.

10. Conclusions

Bark beetles and their effects on fuel accumulations and subsequent fire hazard are poorly understood. Clearly, bark beetles cause tree damage and/or mortality that can alter fuels complexes of lodgepole pine, Douglas-fir and Engelmann spruce forest types, among others, and consequently associated fire behavior. Mortality caused by Douglas-fir beetle, mountain pine beetle and spruce beetle species can contribute significant amounts of large woody debris to the fuels complex (Lotan et al., 1985; Romme et al., 1986; McCullough et al., 1998). However, a period of 20 or more years after the outbreak may be necessary before the majority of beetle-killed trees are on the forest floor (Mielke, 1950). Initially after the outbreak, there are substantial increases in the amount of fine dead fuels in the canopy. The majority of these fuels remain in the canopy for 2–3 years after the outbreak (Knight, 1987; Schmid and Amman, 1992; Page and Jenkins, 2007a). Immediately following the outbreak, there may be a large increase in the amount of understory shrub and herbaceous plants (Reid, 1989; Stone and Wolfe, 1996; Page and Jenkins, 2007a). With an abundance of green understory shrub and herbaceous plants there may be a decrease in potential surface fire behavior (Stocks, 1987). The large amount of fuels present due to beetle mortality may not represent abnormal fuel conditions in high elevation systems (Brown, 1983; Johnson et al., 2001; Schoennagel et al., 2004). In the case of spruce beetle, some research indicates that fuel accumulations caused by the spruce beetle do not increase the density of fires in a given area when compared to unaffected stands (Bebi et al., 2003; Kulakowski et al., 2003). Over decades, when there is a significant component of large down wood, there will be an increase in fire severity during high-intensity fire events.

Although bark beetles have significant effects on fuels, weather conditions play an important role in fires in subalpine forests (Bessie and Johnson, 1995). Conifer forests experiencing extreme fire weather are subject to intense fires under many fuel conditions, while stands with high fuel loadings will only have an affect on crown fire initiation under moderate fire weather.

Depending on pre-existing stand conditions bark beetle treatments such as moderate thinnings may result in less potential for extreme behavior compared to unmanaged stands. Greater fuel depths, mid-flame wind speeds and lower fuel moistures in heavily treated stands might increase potential fire behavior compared to unmanaged stands. This necessitates bark beetle management strategies more thoroughly consider post-harvest fuel treatments to reduce surface fuels to lower the risk of severe wildfire.

Fuels reduction treatments may result in residual tree damage and/or the production of host material (green slash and stumps) and volatiles that can attract and concentrate certain bark beetle species, particularly if treatments coincide with beetle flight periods. Land managers can mitigate for bark beetle population increases by avoiding treatments during bark beetle flight, minimizing activities in those portions of stands where decay and root diseases are prevalent, and managing green slash.

Bark beetles frequently attack fire-injured trees and contribute to delayed tree mortality. Several models have been developed that provide land managers with the best predictors for the post-fire Douglas-fir mortality. Whether bark beetle populations build in fire-damaged trees and contribute to outbreaks remains uncertain. Land managers may determine the likelihood of bark beetle depredations using assessments of stand conditions integrated with burn intensities for bark beetle species of concern.

Management plans that consider bark beetle/fire interactions can provide decision-makers with better guidance for meeting important resource objectives, reducing treatment costs,

minimizing adverse ecological impacts, and avoiding potential controversy. Management plans that explicitly define both short and long-term resource objectives enable land managers to devise the most appropriate treatment strategies. The mitigation of potentially adverse bark beetle and fire effects is maximized when treatments occur at landscape scales and integrate the spatial arrangement of forest types and stand conditions.

Acknowledgements

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