The Impact of Western Spruce Budworm on Fire Behavior in High Elevation Spruce-fir Ecosystems

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A Professional Paper

Submitted in Partial Fulfillment

Of the Requirements for the Degree of

Masters of Forestry

Northern Arizona University

May, 2014

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Abstract

Western spruce budworm (WSBW) (*Choristoneura freemani* Razowski) is the most significant defoliator of coniferous trees in the western United States. Although this species is native to western North America current outbreaks may be exceeding historical norms. As demonstrated by bark beetles (*Dendroctonus spp.*), native species which have altered life cycles and survival rates due to climate change can cause significant mortality in western forests and alter fire regimes. Using the Fire and Fuels Effects- Forestry Vegetation Simulator (FFE-FVS) this study examined the impact of a WSBW outbreak on high elevation spruce-fir ecosystems in northern New Mexico to assess current potential fire behavior in stands experiencing ongoing WSBW outbreak to non-outbreak conditions, and to compare future fire behavior between stands with ongoing WSBW outbreak and stands without WSBW. We found that fire behavior decreased overall in stands experiencing WSBW outbreak, but fire in these stands still tended to be high severity, crown fires which resulted in significant mortality. Our work contributes to the body of information on the complex interactions between fire and insect outbreaks. As climate change continues to alter disturbance regimes, it is important to understand how these disturbances currently interact and how they may change in the future.

Introduction

Disturbance is a vital component of ecosystems which shape and maintain the natural landscape and its processes. Once considered an uncommon or irregular event which moved communities away from equilibrium (Cowles 1911, Clements 1916), disturbance is now recognized as an integral part of ecosystems which impacts local communities and physical conditions, and is in turn impacted by local conditions (White 1979, Sousa 1984, Picket and White 1985). Under this interpretation disturbance creates opportunities for new individuals to become established, promotes heterogeneity across the landscape and allows for a diversity of species to persist. While disturbance is a natural part of a functioning ecosystem, introduction of new disturbances (Whisenant 1990, Mayer and Swetnam 2000), changes to existing regimes (Covington et al. 1997, Veblen et al. 2000), and alterations to weather and precipitation due to climate change, can lead to drastic and potentially detrimental alterations of natural disturbance regimes (Williams and Liebhold 2002, Westerling et al. 2006, Waring et al. 2009, Boyd et al. 2013). Insect outbreaks and wildfire are two of the most significant disturbance regimes in North America. In Canada, insects resulted in the defoliation or mortality of 6.2 % of the forested landscape in

2011 while fire burned 2.6 million hectares (the ten year average is 1.8 million hectares) (Canadian Forest Service 2013). In the United States, western bark beetles (*Dendroctonus* spp., *Scolytus* spp., *Pseudohylesinus* spp., *Pityogenes* spp. and *Ips* spp.) resulted in the defoliation or mortality of nearly 5.3 million forested acres in 2011 (Potter and Conkling 2012) while fire burned 8.7 million acres (NICC 2012).

Over the next century the climate is predicted to dramatically change in the Southwest resulting in higher temperatures, and more variable precipitation (IPCC 2007, Garfin 2013). In addition, La Niña weather patterns which result in drought conditions across Arizona and New Mexico are predicted to increase (Woodhouse et al. 2010). There is evidence that these changing weather patterns are increasing fire size, extending the fire season, and increasing fire duration across the West (Westerling et al. 2006). Impacts of milder winters and longer summers on insects include reduced overwinter mortality and increased lifecycles per year resulting in larger outbreaks and greater tree mortality across the landscape (Swetnam and Lynch 1989, Jenkins et al. 2008, Waring et al. 2009). In addition to climate change, anthropogenic impacts over the last century across the West including fire exclusion and logging practices have dramatically altered forest structures resulting in denser stands with more shade tolerant species. The resulting forests are more vulnerable to catastrophic fire and severe insect outbreaks because of the homogenous, dense stand structure (Covington et al. 1997, Veblen et al. 2000). Due to the extent and severity of fire and insect outbreaks on the landscape, understanding the interactions between these two disturbances is critical, especially as climate change leads to altered and less predictable disturbance patterns. Many researchers suggest that insect outbreaks would result in more frequent, larger, and higher severity fire because of increases in desiccated fine fuels in the canopy, and increased surface fuel loading. Additionally, open conditions due to insect mortality would result in additional solar radiation and increases in wind speed, reducing fuel and foliar moisture (Hopkins 1909, Brown 1975, Furyaev et al. 1983, Knight 1987). However, this interaction hasn't been extensively studied until recently when the vast outbreaks of bark beetles (*Dendroctonus spp*.) sparked interest, and the advent of fire models allowed researchers to more easily examine the interactions between insect outbreaks and fire (Hummel and Agee 2003, Simard et al. 2011, Hicke et al. 2012, Schoennagel et al. 2012, Sturtevant et al. 2012).

While the impacts of mountain pine beetle (*Dendroctonus ponderosae* Hopkins) outbreaks in Colorado and British Columbia have been highly publicized (Hawkes et al. 2003, Aukema et al. 2006, Klutsch et al. 2009), the western spruce budworm (WSBW) (*Choristoneura freemani* Razowski) also has a significant impact on western forests. Western spruce budworm is the most significant native defoliator of coniferous trees in the western United States (Fellin and Dewey 1982, Brookes et al. 1987). In 2012 WSBW outbreaks resulted in the mortality or defoliation of 478,000 acres in the southwestern United States in 2012 (USDA Forest Service 2012). While this species is endemic to the U.S and Canada there is some evidence that current outbreaks in New Mexico and Oregon are lasting longer and are more severe than historic outbreak patterns due to logging practices, fire exclusion, and climate change (Anderson 1987, Swetnam and Lynch 1989, Swetnam and Lynch 1993, Swetnam and Betancourt 1995). Due to the possible increase in severity and duration of WSBW outbreaks across the West, it is important to examine the interaction between fire and WSBW outbreaks.

Similar to the results from studies on bark beetle outbreaks, recent research into the impact of WSBW outbreaks on fire behavior have been inconclusive and differ between ecosystems. Impacts of WSBW outbreaks on trees include overstory mortality, top kill, understory mortality, branch die-back, and needle mortality (Fellin and Dewey 1982, Brookes et al. 1987). Impacts on fuels include increased surface fuel loading, reduced canopy bulk density, discontinuous canopy structure (Hummel and Agee 2003), increased desiccated fine fuels in the canopy and ladder fuels (Fellin and Dewey 1982, Brookes et al. 1987) and, decreased fuel moisture content of surface, foliar, herbage, and woody fuels (Hopkins 1909, Brown 1975, Fureyaev et al. 1983, Knight 1987). In the eastern Cascades, Hummel and Agee (2003) found that following a WSBW outbreak fuel loads were significantly increased, canopy closure was significantly reduced, and the majority of mortality was sustained in smaller diameter classes. This resulted in an increase in surface flame length from pre-outbreak to late outbreak conditions, however, crown fire potential did not change, and there was not a significant difference in basal area (BA) loss from fires (Hummel and Agee 2003). In contrast, Lynch and Moorcroft (2008) found that the likelihood of fire was significantly reduced for seven years following WSBW outbreaks in interior Douglas-fir (*Pseudotsuga menziesii*) forests in British Columbia.

Due to the uncertainty of WSBW impacts on fire behavior and the impact of climate change on outbreak cycles and wildfire, it is important for managers to know how WSBW impacts fire behavior. Focusing on the spruce-fir forest type in northern New Mexico, we investigated the impact of WSBW outbreaks on potential fire behavior in this ecosystem. Our specific research objectives were to assess current potential fire behavior in stands experiencing ongoing WSBW outbreak, and pre-outbreak stands, and compare future modeled fire behavior between stands which are currently experiencing WSBW outbreak and stands without WSBW.

Methods

Four spruce-fir stands which were experiencing defoliation from WSBW were sampled in northern New Mexico. Two stands were located in northwestern New Mexico on the Cibola National Forest, and two stands were located in north-central New Mexico on the Carson National Forest. Stands were selected based on U.S Department of Agriculture (USDA) disease and insect aerial survey maps (Forest Health Protection 2012). Sampled stands had >50% host species composition: Engelmann spruce (*Picea engelmannii*), white fir (*Abies concolor*), corkbark fir (*Abies bifolia*), and Douglas-fir, had not received any anthropogenic treatment in the previous twenty years, and were of similar slope, aspect, and elevation (Polinko 2014). Records of fire history in these stands could not be found, but due to the lack of fire scars, old age of aspen trees, and dense conditions, it appears that these stands have not experienced fire recently.

In order to compare outbreak stands conditions to non-outbreak conditions we attempted to find stands which were not experiencing WSBW outbreak to sample, but none could be found in 2012 or 2013. To obtain non-outbreak conditions we used stand inventory data from six stands collected by the USDA Forest Service from pre-outbreak time periods in spruce-fir stands on the Carson and Cibola National Forests. All stand inventory data from pre-outbreak time periods were taken from stands within two kilometers of our sampled stands, and had similar aspects, elevations, and species compositions as sampled stands.

Soils in stands on the Carson National Forest are predominately Marosa cobbly sands, and rock outcrops, while the Cibola National Forest stands are Parkay complexes (USDA, [internet] http://websoilsurvey.sc.egov.usda.gov, January 2014). Stands were predominately north facing and had an average slope of about 17°. Elevations on the Cibola National Forest range from 3,118 to 3,173 m, and 3,018 to 3,296 m on the Carson National Forest. Mean annual precipitation on the Cibola National Forest is 55.6 cm and 61.9 cm on the Carson National Forest. Mean annual temperatures range from -6° to 21.2° C on the Cibola National Forest stands, and -6.8° to 21.8° C on the Carson National Forest (USDA, [internet] http://forest.moscowfsl.wsu.edu/ January 2014).

Outbreaks of WSBW have been documented in northern New Mexico throughout the past century using tree ring analysis (Swetnam and Lynch 1989), and more recently by Forest Health Protection (FHP) aerial surveys. The ongoing outbreak in northern New Mexico began in the early 1990's. On the Cibola National Forest, stands sampled for this study had been defoliated for the past 12 to 17 years, and for 14 years on the Carson National Forest (USDA, [internet] http://www.fs.usda.gov/detail/r3/forest-grasslandhealth/insects-diseases/ January 2014).

Study Design and Field Measurements

Stands experiencing outbreak were sampled in 2013 using a randomized, systematic grid of ten, 1/50th hectare overstory plots and are part of a larger study on the impacts of WSBW on stand dynamics and defoliation (Polinko 2014). Five, 1/1000th hectare regeneration plots were nested inside each of the overstory plots in order to quantify regeneration. One regeneration plot was located at plot center and the other four were located 3.95 m from plot center in each cardinal direction. Measurements taken on all overstory trees (trees with diameter at breast height (DBH, at 1.37 m height) >12.7 cm) included DBH, species, condition (live or dead), height, height to live crown base, and percent defoliation. Saplings (>1.37 m in height and < 12.7 cm at DBH) and seedlings (<1.37 m in height) were recorded within the five regeneration plots. Sapling measurements included DBH and height, and seedling heights were recorded in 0.15 m increments. Defoliation was assessed visually in 10% increments on host species for both seedlings and saplings (Polinko 2014).

Pre-outbreak stand metrics including Individual tree data (species, DBH, height, live crown ratio, and regeneration) and stand level data (aspect, slope, and elevation) was taken from the stand inventory data which was provided by the USDA Forest Service.

Fuels data from stands experiencing outbreak in 2013 were recorded using modified Brown's transects (Brown 1974). Three 16.7 m transects were established from each overstory plot center at 0°, 120°, and 240°. Fuels were tallied by size class: 1-hour fuels (0-0.63 cm) from 1.52-3.35 m, 10-hour fuels (0.63-2.54 cm) from 1.52-5.18 m, 100-hour fuels (2.54-7.62 cm) from 1.52-16.7 m and 1000-hour fuels (>7.62 cm) from 1.52-16.7 m. For 1000-hour fuels, diameter was recorded along with decay class on a scale from 1-5 (1 showing no sign of decay and 5 being extremely decayed). Duff and litter depths were taken at 1.52, 6.09, 10.66 and 15.24 m along each transect. Fuel loading was calculated on a megagrams per hectare scale, and 1000-hour fuels were split into decayed (decay class 4 and 5), and sound (decay class 1, 2, and 3) classes. Because species for 1000-hour fuels were not recorded on site, standing species composition was used as a proxy for 1000-hour fuel species composition. Published values of species' specific gravities were used in the fuels calculations (Sackett 1980, Van Wagtendonk et al. 1996, Van Wagdendonk et al. 1998, Green et al. 1999).

Because stand inventory data provided by the USDA Forest Service does not include fuel loading, published fuel loads from spruce-fir forests in the Manti-LaSal and Fishlake National Forests in southern and central Utah (Jorgensen and Jenkins 2011; Table 1) were used. These stands were not as dense as those sampled for our study, but were the only published fuels loads we could find that included 1, 10, 100, and 1000-hour fuels as well as litter and duff measurements from spruce-fir forests not experiencing WSBW outbreaks.

Fire Behavior Modelling

For this study the Forestry Vegetation Simulator (FVS) was used which is a distant-independent, individual-tree forest growth and yield model that is widely used throughout the United States (Crookston and Dixon 2005). To model fire behavior in these stands the Central Rockies (CR) variant of FVS utilizing the Fires and Fuels Extension (FFE-FVS) was used (Rebain et al. 2013). Prior to initiating model runs, pre-outbreak stands were grown out to 2013 in FVS so the impact of WSBW on fire behavior could be examined over the same time period. In order to obtain the full breadth of fire metrics from FVS, both potential fire as well as fire reports from prescribed burns were used in pre-outbreak and outbreak stands to determine the impact of western spruce budworm on fire. Because FVS does not have a wildfire function, prescribed fire was used as a proxy for wildfire.

For potential fire as well as prescribed fire reports, Individual tree (DBH, height, live crown ratio, and regeneration) and stand level data (aspect, slope, and average fuel loading) from both sampled outbreak stands, and pre-outbreak stands were entered into FVS to initiate model runs. Reports were generated for both severe (97th percentile) and moderate (80th percentile) fire weather conditions in the spruce-fir forest type in New Mexico (Tessa Nicholet personal communication, January 2014) and (Rebain et al 2013) (Table 2). Fuel models were set to the FVS defaults.

Current fire conditions

In order to examine current fire behavior in 2013, prescribed fire was modeled in sampled outbreak stands, and pre-outbreak stands grown out to 2013 in FVS. In addition, potential fire reports were generated in FVS for pre-outbreak and outbreak stands in 2013. Modeled fire as well as potential fire reports were generated for all stands under both moderate and severe fire weather conditions.

Future fire behavior scenarios

Modeled prescribed fire and potential fire reports were run under the assumption that defoliation of WSBW ceased after 2013 in outbreak stands. Potential fire reports were generated for all pre-outbreak and outbreak stands under moderate and severe fire weather conditions in ten year cycles from 2013-2063 in order to determine potential fire behavior, stand conditions, and fuel loading. Prescribed fire was also modeled in all outbreak and pre-outbreaks stands in ten year cycles from 2013-2063 to examine the long-term impact of WSBW on fire. A total of twelve models runs were conducted per stand, six under severe fire weather conditions and six under moderate fire weather conditions. Each model run consisted of one fire so that fire was modeled in each decade from 2013-2063. For example, fire modeled in 2043 would have been proceeded by three decades of growth. Using this method we were able to examine the impact of WSBW on fire over a fifty year time period.

Results

Current conditions

In 2013, live and dead trees were equally abundant in outbreak stands while live BA was greater than dead BA (Table 3). Engelmann spruce accounted for the greatest amount of live BA followed by Douglas-fir, aspen, corkbark fir, limber pine (*Pinus flexilis*), and white fir (Table 4). Pre-outbreak stands grown to 2013 had substantially greater live TPH, and BA than dead TPH and BA (Table 3). Corkbark fir accounted for the greatest amount of live BA followed by Engelmann spruce, Douglas-fir, aspen, and Rocky Mountain bristlecone pine (*Pinus aristata*) (Table 4).

Total surface fuel loading in outbreak stands was high with substantial variation between sites. The majority of the fuel loading was in sound and rotten 1000-hour fuels which also had substantial variation between sites. Both duff and litter fuel loads were higher on average than 1-hour, 10, hour, and 100, hour fuels. One-hour fuels accounted for the least amount of fuel loading across all stands, followed by 10-hour, and then 100- hour fuels (Table 2)

Current potential fire behavior

Under current stand conditions, pre-outbreak and outbreak stands had similar total flame lengths under moderate fire weather conditions, but pre-outbreak stands had longer total flame lengths under severe weather condtions (Table 5). Surface flame lengths were not substantially different under moderate or severe fire weather conditions (Table 5). The probability of torching was nearly 100 % under severe conditions for pre-outbreak and outbreak stands, and was highly likely (≥ 85%) under moderate conditions (Table 5). Torching indices were similar between pre-outbreak and outbreak stands, and crowning indices for pre-outbreak stands was nearly 1/2 that of outbreak stands (Table 5). Canopy base height was similar between stands while canopy bulk density was higher in pre-outbreak stands (Table 5). Potential basal area mortality was equal for both pre-outbreak and outbreak stands under both moderate and severe weather conditions, with all overstory BA predicted to die under severe fire weather conditions (Table 5).

Future fire behavior and consumption

Under moderate weather conditions, total flame lengths (total flame lengths including surface and canopy flame lengths) in pre-outbreak stands were slightly higher on average in all decades except 2063 (Figure 1a). Over time pre-outbreak stands showed decreases in total flame lengths while total flame lengths in outbreak stands increased (Figure 1a). Total flame lengths in pre-outbreak stands were greater than outbreak stands under severe fire weather conditions across all time period (Figure 1b). Both pre-outbreak and outbreak stands under severe weather conditions showed increases in total flame lengths over time.

Surface flame lengths were similar between pre-outbreak and outbreak stands under moderate conditions and increased slightly over time (Figure 1c). Under severe weather conditions outbreak stands experienced longer surface flame lengths until 2033. Pre-outbreak stands showed substantial increases in surface flame lengths from 2013 to 2063 while surface flame lengths in outbreak stands decreased until 2043 and then increased (Figure 1d).

Scorch height (which is the average height to which foliage has been browned by fire) was initially higher in pre outbreak stands under moderate weather conditions, but over time scorch height steadily increased in outbreak stands and decreased in pre-outbreak stands resulting in similar conditions by 2063 (Figure 1e). Under severe weather conditions, scorch height was substantially higher in pre-outbreak stands across all time periods (Figure 1f). Scorch height in pre-outbreak stands remained constant through time, while outbreak stand scorch height increased from 2013-2063.

Crowning indices (which is the measure of wind speed required for an active crown fire) were higher in outbreak stands than pre-outbreak stands (Figure 1g). Over time the wind speed needed for an active crown fire remained constant in pre-outbreak stands while wind speeds needed for an active crown fire decreased over time in outbreak stands (Figure 1g). Torching indices (which is the windspeed required for trees to torch during a fire) were around 0 km/ hr in all time frames for pre-outbreak stands (Figure 1h). Outbreak stands also had low wind speeds needed for torching but were slightly higher on average than pre-outbreak stands (Figure 1h).

Under moderate fire weather conditions outbreak stands had less mortality than pre-outbreak stands in 2023 and 2033, but were similar in all other years (Figure 2a). Nearly all live BA died due to fire in both outbreak and pre-outbreak stands under severe fire weather conditions (Figure 2b).

Under moderate fire weather conditions, pre-outbreak stand crown consumption (which is the amount of fuels consumed (Mg ha^{-1}) by fire in the canopy) was higher than outbreak stands but decreased while outbreak stand crown consumption increased over time until (Figure 2c). Crown consumption was higher in pre-outbreak stands than outbreak stands under severe weather conditions across the entire time frame (Figure 2d). Crown consumption increased in outbreak stands from 2013 to 2063 while pre-outbreak stands remained constant (Figure 2d).

Total surface fuel consumption (which is the amount of fuels consumed (Mg ha^{-1}) by fire on the forest floor) was higher on average in outbreak stands than pre-outbreak stands under both moderate and severe weather conditions (Figure 2e,f). Both pre-outbreak and outbreak surface fuel consumption increased over time (Figure 2e, f).

The percentage of trees crowning under moderate conditions was similar in pre-outbreak and outbreak stands in all decades under both severe and moderate fire weather conditions (Figure 2g, h). Pre-outbreak stands experienced around 25% of trees crowning under moderate conditions and remained constant throughout time. About 20% of trees in outbreak stands experienced crowning under moderate conditions, but increased over time. Under severe conditions, all trees experienced crowning in pre-outbreak stands across the entire time frame. Outbreak stands experienced a high percentage of trees crowning which increased over time to 100% of overstory trees by 2053 (Figure 2g, h).

Canopy base height (the height from the forest floor to the base of the canopy) of pre-outbreak was lower on average than outbreak stands (Figure 2i). Canopy bulk density (amount of fuels per cubic meter) for pre-outbreak stands were higher than outbreak stands across all time periods (Figure 2j). Over time, canopy bulk density in pre-outbreak stands remained constant but increased in outbreak stands.

Discussion

Over the past two decades WSBW in northern New Mexico has altered conditions of spruce-fir stands. Western spruce budworm in spruce-fir stands has increased the amount of standing dead TPH and BA while decreasing the live BA and TPH. However, the total standing TPH and BA has not substantially change. In addition, species composition shifted from predominantly corkbark fir and Engelmann spruce, to Engelmann spruce, Douglas-fir, and aspen.

Pre-outbreak stands showed more severe fire behavior across nearly all metrics in FFE-FVS with the exception of surface flame lengths. Longer surface flame lengths in outbreak stands are most likely due to the fuel model which FVS selected. For outbreak stand, FVS predominantly selected fuel model 12 due to the high surface fuel loads resulting in higher flame lengths. FVS predominantly selected fuels model 10 for pre-outbreak stands which resulted in lower flame lengths but a faster rate of spread. Besides surface flame lengths and surface consumption, the overall predicted decrease in fire behavior in outbreak stands probably resulted from the decrease in live BA and TPH, which resulted in lower canopy bulk density and slightly higher canopy base heights. However, even though outbreak stands experienced less severe fire behavior due to the reduction in canopy bulk density, live BA, and TPH, outbreak stands still experienced passive crown fire in nearly all decades, and a mixture of active and passive crown fire under severe conditions. Pre-outbreak stands experienced passive crown fire under moderate fire weather conditions, and active crown fire under severe conditions.

Over time, both pre-outbreak and outbreak stands tended to show an increase in fire behavior between 2013 and 2063. Outbreak stands tended to show a more dramatic increase in fire behavior (flame lengths, BA lost, fuels consumption) than pre-outbreak stands over time, indicating a return to pre-outbreak fire behavior levels in the absence of WSBW. However, by 2063 many of the fire metrics had not returned to pre-outbreak fire behavior levels suggesting that WSBW has a long lasting (>50 year) impact on fire behavior in the spruce- fir ecosystem. Fire behavior may be impacted more dramatically and for a longer period of time if defoliation continues in these stands at current rates. These results are not surprising as high elevation spruce-fir forests inhabit harsh climates with short growing seasons, and slow growth rates which rebound slowly from severe insect outbreaks (Alexander 1984).

While the sample size is small (n= 4), these trends indicate that WSBW outbreak leads to a long lasting decrease in most fire behavior metrics. Even with a decrease in fire behavior, mortality due to fire is still substantial in stands defoliated by WSBW. Basal area loss due to fire in outbreak stands was nearly 100 % under severe weather conditions in all years and remained above 50% in moderate fire weather conditions. This indicates that while outbreak stands show some decreases in fire behavior, the driving factor for fire in this ecosystem may not be the availability surface and ladder fuels but optimal weather conditions (Kulakowski et al. 2003).

Our results indicate that both outbreak and pre-outbreak stands are highly vulnerable to stand-replacing fire. This is a common occurrence in spruce-fir ecosystems (Arno 1980, Agee 1996). Spruce and fir species tend to have shallow roots which are easily damaged by high severity fire, thin bark, low canopy base heights, and high canopy bulk density, resulting in high mortality from fire (Alexander 1987, Edmonds et al. 2000). In addition, fire return intervals for these ecosystems are long, often well over 100 years (Arno 1980, Agee 1996), which results in abundant ladder fuels, dense stands, and high surface fuel loading (Alexander 1987). Therefore, high elevation spruce-fir stands which have not been recently logged, or experienced fire tend to support fuel structures which are highly susceptible to stand-replacing, high-severity fire (Arno 1980, Alexander 1987, Agee 1996). Due to the persistent cool wet conditions, severe fires in this ecosystem typically occur only in the late summer and early fall during drought years when the snowpack is low and warm dry conditions reduce fuel moisture content (Agee and Smith 1984, Agee 1996, Kulakowski et al. 2003).

Previous research in the spruce-fir ecosystem in the western United States on the interaction between insects and fire have focused on the role of spruce bark beetle (*Dendroctonus rufipennis*) in these stands. Unlike WSBW, spruce bark beetle primarily causes mortality in large diameter spruce and shifts species composition to fir and pine (Veblen 2000, Bebi et al. 2003, Kulakowski et al. 2003, Kulakowski and Veblen 2007). Outbreaks of spruce bark beetles in this ecosystem lead to decreases in live BA, canopy base height, and bulk density, and increases in surface fuel loading (Jorgensen and Jenkins 2010). These results are similar to trends found in our study which also show decreases in live BA and canopy bulk density. Despite these alterations to fuels due to spruce bark beetle, it has been found that these outbreaks do not alter the number of fires, fire severity, or area burned (Bebi et al. 2003, Kulakowski et al. 2003, Kulakowski and Veblen 2007). This is because the fire regime in this ecosystem is dictated by weather and conditions which produce large, high-severity, stand-replacing fire occur infrequently (Arno 1980, Alexander 1987). So, even though Insect outbreaks can alter fuel conditions and impact potential fire behavior, the likelihood of fire occurring is low.

The interaction between WSBW and fire has not been extensively studied. However, research into this interaction has shown that WSBW outbreaks have a significant impact on certain fire metrics and fuel loading (Hummel and Agee 2003). Similar to our study, Hummel and Agee (2003) found that surface fuel loading and surface flame lengths were significantly greater in late outbreak stands when compared to early outbreak conditions. However, this did not significantly alter BA or stand density loss due to fire from early to late outbreak conditions. Similar to our study, WSBW outbreak had an impact on individual fire variables, but did not result in substantial differences in terms of the fire's overall impact on these stands. Moorcroft and Lynch (2008) examined the occurrence of fire in interior Douglas-fir forests in British Columbia over a 26 year period to determine if WSBW impacted the likelihood of fire. They found that fire hazard decreased for 5 to 10 years after WSBW outbreaks. While it is useful to examine our findings in the context of these studies, neither was conducted in spruce-fir so care should be taken when extrapolating their results.

A close relative of WSBW is the spruce budworm (SBW) (*Choristoneura fumiferana*) which is a defoliator that exhibits similar outbreak patterns in the northeastern United States and eastern Canada. Their main host species is balsam fir (*Abies balsamea*), and outbreaks of this species typically result in defoliation and mortality of host species (Fleming et al. 2002). Mortality due to SBW tends to occur on a shorter time scale and results in higher mortality than WSBW (Fleming et al. 2002, Sturtevant et al. 2012). Fire intensity of ignitions started in stands following SBW outbreaks increased for 5 to 8 years following outbreaks due to higher surface fuel loading and increases in ladder fuels (Stocks 1987). In addition, the number of fires on the landscape and the area burned following outbreaks was found to increases for three to nine years following outbreaks (Flemming et al. 2002). These results are contrary to our findings of decreased fire behavior following outbreaks of WSBW. However, SBW outbreaks occur on a different time scale and with more significant mortality than WSBW outbreaks. Also, balsam fir is a much different ecosystem than high elevation spruce-fir, and tends to experience a low-intensity, low-severity regime with long return intervals (Bergeron et al. 2001, Bergeron et al. 2004, Ali et al. 2008). The differences in these two ecosystems as well as the difference between the two insect outbreak patterns are probably the cause of predicted and observed differences in fire behavior following outbreaks.

Extensive research into mountain pine beetle (*Dendroctonus ponderosae*) and Douglas-fir beetle (*Dendroctonus pseudotsugae*) outbreaks have shown similar impacts on fire behavior as our study. The increase of coarse woody debris, reduction in canopy bulk density, and disruption of crown continuity after dead needles have fallen in bark beetle outbreaks result in decreases in active crown fires and torching indices, and increases in surface flame lengths (Simard et al. 2011, Hicke et al. 2012, Schoennagel et al. 2012). This seems to reinforce our results which show a decrease in canopy bulk density, decrease in total flame lengths, and scorch heights in stands experiencing outbreaks with increased surface flame lengths.

While historic fire regimes in high elevation spruce-fir ecosystems follow a high intensity, stand-replacing, low-frequency pattern, this may shift with changing climate patterns. Climate change is predicted to alter weather patterns across the United States resulting in higher temperatures, and more variable moisture conditions in the Southwest (IPCC 2007, Garfin 2013). In addition, La Niña weather patterns are predicted to increase resulting in more frequent and severe drought periods across the Southwest (Woodhouse et al. 2010). Though moisture is highly difficult to predict, the timing of precipitation is predicted to change so less precipitation will fall as snow reducing snowpacks (IPCC 2007). This could alter the fire regime in spruce-fir forests which typically only experiences fires infrequently in drought years with low snowpacks (Agee 1996). Increases in high-severity, stand-replacing fires could have significant implications to human settlements and ecosystem functions. High severity, stand replacing fire can dramatically impact watersheds and hydrology resulting in the sedimentation of water supplies, and intense flooding. A recent example of negative impacts due to stand replacing fire was the Shultz fire which occurred northeast of Flagstaff, Arizona which resulted in significant flooding (Neary et al. 2012). The total cost estimate from this 6,100 ha fire is between 133 and 143 million dollars (Combrink et al. 2013).

Besides the financial and anthropogenic implications of stand replacing fires on water supplies and flooding, sedimentation can cause significant harm to ecosystems. Studies following the Yellowstone fire in the 1980's found significant numbers of dead trout in creeks downstream from the fire (Minshall and Brock 1991). Flood events can also significantly damage riparian vegetation allowing for the colonization of invasive species such as tamarisk (*Tamarix spp*.) (Stevens and Waring 1985). Thus the impact of stand replacing fire on high elevation spruce-fir forests can have wide ranging ecological and anthropogenic impacts.

Conclusion

The results of this study show that contrary to conventional logic, WSBW outbreaks result in decreased fire behavior. Overall decreases in nearly all fire metrics are most likely due to decreases in live BA and canopy bulk density. In addition, the disruption of crown continuity due to mortality may create a structure that is less likely to result in an active crown fire. It is important to note that even with reductions in overall fire behavior, the effects of the modeled fire still resulted in high BA mortality with low wind speeds needed for torching and active crown fires under both moderate and severe fire weather conditions. This indicates that the impacts of WSBW outbreaks on spruce-fir ecosystems, while significant, is not enough to prevent stand-replacing, high-severity fires in the spruce-fir ecosystem.

While a few other studies have shown some increases in fire behavior due to insect outbreaks (Stocks 1987, Flemming et al. 2002, Hummel and Agee 2003, Simard et al 2011, Hicke et al. 2012, Schoennagel et al. 2012, Sturtevant et al. 2012) these were in different ecosystems which have a different fire regime, stand structure, and species composition. The natural fire regime in spruce-fir forests is stand-replacing high-severity, low-frequency fire due to the dense stand conditions, build-up of ladder fuels, and lack of fire adapted traits in spruce and fir species (Alexander 1987). As Kulakowski et al. (2003), and Kulakowski and Veblen (2007) showed in their study of high alpine ecosystems and spruce bark beetle outbreaks, the driving force behind fire in spruce-fir forests is advantageous fire weather conditions and not the addition and restructuring of desiccated fuels from insect outbreak.

This study adds to the body of research which shows that the interaction between insect outbreaks and fire are complex and dependent on the ecosystem. While interior balsam fir forests show increased fire behavior following outbreaks, high alpine spruce-fir forests fire patterns are weather driven and showed decreases in fire behavior. However, climate change will continue to alter the natural fire regimes in high elevation spruce-fir forests. Under climate change conditions there may be an increase of fire in these ecosystems due to increased temperature, drought conditions, and reduced snowpacks. In concert with decreases in fuel and foliar moisture due the increased temperatures, and open conditions because of WSBW outbreaks, stands could become susceptible to fire sooner in the season due to drier conditions. This could lead to an increase in the fire season length, and favorable conditions for fire ignitions and spread. It is foreseeable that fire could start in outbreak stands and then spread outside of the outbreak area, potentially impacting large areas of high elevation spruce-fir ecosystems. Mitigating the risk of stand-replacing fire in these ecosystems would be costly, and intensive, and may not be desirable because if would alter the natural fire regime. Because of the uncertainty surrounding the interaction between climate change, fire, and WSBW in the spruce-fir forests, it is vital to continue to monitor this ecosystem and these disturbance regimes.

Acknowledgements:

We thank the McIntire-Stennis appropriations to the Northern Arizona University School of Forestry and the State of Arizona for their financial support, and Adam Polinko, Matt Blanford, Alec Latusek, and David Tyler for their assistance with fieldwork.

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Tables and Figures:

Table 1: Fuel loads used to initiate fire behavior in the Fuels and Fire Extension- Forestry vegetation Simulator. Fuel loads calculated from stands in northern New Mexico sampled in 2013 under western spruce budworm outbreak (outbreak) and non-outbreak (pre-outbreak) conditions. Standard errors shown in parenthesis.

Fuel Loading (Mg ha^{-1})								
	1-hour	**10-hour**	**100-hour**	**1000-hour (sound)**	**1000-hour (rotten)**	**Litter**	**Duff**	**Total**
Pre-Outbreak[1]	1.05	2.69	3.51	25.46	17.01	4.02	30.44	84.18
	(*0.06*)	(*0.17*)	(*0.16*)	(*3.24*)	(*1.34*)	(*2.01*)	(*1.91*)	(NA)
Percentage of Total	1.2%	3.2%	4.2%	30.2%	20.2%	4.8%	36.2%	-
Outbreak[2] (N=4)	0.62	2.31	6.56	124.11	67.5	18.9	46.58	266.57
	(*0.15*)	(*0.42*)	(*1.5*)	(*49.86*)	(*30.3*)	(*2.2*)	(*5.03*)	(*86.16*)
Percentage of Total	0.2%	0.9%	2.5%	46.6%	25.3%	7.1%	17.5%	-

1. Jorgensen and Jenkins (2010)
2. Stands sampled for this study in 2013

Table 2: Moderate and severe fire weather conditions used in the Fires and Fuels Extension- Forestry Vegetation Simulator to model fire behavior.

Moisture %			
Fuel Type	**Moderate (80th percentile)**	**Severe (97th percentile)**	**Source**
Duff	50	15	Rebain et al. 2013
1-hour fuels	3	1	Tessa Nicholet 2014*
10-hour fuels	4	2	Tessa Nicholet 2014
100-hour fuels	6	3	Tessa Nicholet 2014
1000-hour fuels sound	9	4	Tessa Nicholet 2014
herbage	30	30	Rebain et al. 2013
Woody	60	60	Rebain et al. 2013
Fire Weather			
	Moderate	**Severe**	
Wind Speed km/hr	8.04	28.96	Tessa Nicholet 2014
Temperature C°	18.88	26.66	Tessa Nicholet 2014

* USFS region 3 fuels specialist personal communication January 2014. Information derived from Coyote, Jarita Mesa, and Brushy Mountain Remote Automated Weather Stations in New Mexico (http://raws.wrh.noaa.gov/roman/index.html).

Table 3: Mean basal area and trees per hectare from stands in northern New Mexico under western spruce budworm outbreak (outbreak) and non-outbreak (pre-outbreak) conditions. Standard errors shown in parenthesis.

Mean Basal Area ($m^2 ha^{-1}$)					
Time Period	**Dead**	**Live**	**Total**	**% Dead**	**% Live**
Pre-Outbreak[1]	3.49	52.83	56.32	6.20	93.80
(n=6)	*(0.76)*	*(2.86)*	*(3.58)*	*(1.10)*	*(1.10)*
Outbreak[2]	20.78	30.94	50.8	40.90	60.91
(n=4)	*(3.56)*	*(3.01)*	*(6.47)*	*(2.60)*	*(2.33)*
Trees Per Hectare					
	Dead	Live	Total	% Dead	% Live
Pre-outbreak[1]	82	965	1047	7.80	92.20
(n=6)	*(17.22)*	*(76.52)*	*(90.29)*	*(1.31)*	*(1.31)*
Outbreak[2]	478	500	978	48.90	51.10
(n=4)	*(45.78)*	*(33.26)*	*(46.12)*	*(3.25)*	*(3.36)*

1. 2013 FVS output from stand exam data under non-outbreak conditions
2. Polinko (2014)

Table 4: Species composition by basal area (BA), and trees per hectare (TPH) from stands in northern New Mexico under western spruce budworm outbreak (outbreak) and non-outbreak (pre-outbreak) condition. Pre-outbreak stand values are shown using FVS outputs from stands grown to 2013. Species composition calculated by live BA, and proportion of standing dead trees by species calculated by TPH. Standard errors shown in parenthesis. ABBI= Abies bifolia, ABCO= Abies concolor, JU SP= Juniperus spp., PIEN= Picea engelmannii, PIFL= Pinus flexilis, POTR= Populous tremuloides, PSME= Pseudotsuga menziesii

Species Composition (%)								
Time Period	**ABBI**	**ABCO**	**JU SP**	**PIEN**	**PIAR**	**PIFL**	**POTR**	**PSME**
Pre-outbreak[1] (n=6)	47.71 *(6.24)*	0 *(0)*	0 *(0)*	39.48 *(3.5)*	0.11 *(0.1)*	0 *(0)*	3.97 *(3.29)*	8.75 *(4.34)*
Outbreak[2] (n=4)	10.22 *(3.6)*	0.47 *(0.41)*	0 *(0)*	48.29 *(14.02)*	0 *(0)*	1.85 *(1.05)*	14.07 *(3.72)*	25.09 *(17.55)*
Proportion of Standing Dead Trees by Species (%)								
	ABBI	**ABCO**	**JU SP**	**PIAR**	**PIEN**	**PIFL**	**POTR**	**PSME**
Pre-outbreak[1] (n=6)	4.38 *(1.18)*	0 *(0)*	0 *(0)*	7.35 *(6.71)*	9.51 *(1.9)*	0 *(0)*	13.57 *(8.71)*	4.64 *(1.68)*
Outbreak[2] (n=4)	42.02 *(16.61)*	37.5 *(20.73)*	50 *(25)*	25 *(21.65)*	48.02 *(15.11)*	0 *(0)*	33.53 *(6.98)*	7.56 *(3.96)*

1. FVS output from stand exam data under non-outbreak conditions
2. Polinko (2014)

Table 5: Potential fire behavior for current (2013) conditions in northern New Mexico under western spruce budworm outbreak (outbreak) and non-outbreak (preoutbreak) conditions, as modeled in Fire and Fuels Effects- Forestry Vegetation Simulator. Standard errors shown in parenthesis.

Potential Fire Under Current Conditions				
	Pre-outbreak[1] (n=6)		Outbreak[2] (n=4)	
Surface flame length (meters)				
Moderate	1.12	*(0.01)*	1.55	*(0.19)*[3]
Severe	2.06	*(0.03)*	2.64	*(0.28)*
Total flame length (meters)				
Moderate	2.68	*(0.43)*	2.57	*(0.4)*
Severe	30.62	*(0.43)*	23.99	*(0.57)*
Probability of torching (%)				
Moderate	91	*(2)*	85	*(9)*
Severe	100	*(0)*	97	*(2)*
Mortality (% of basal area)				
Moderate	79	*(2)*	79	*(3)*
Severe	100	*(0)*	100	*(0)*
Torching indices (km/hr)	0.38	*(0)*	0	*(0)*
Crowning indices (km/hr)	15.37	*(1.08)*	28.23	*(1.99)*
Canopy base height (meters)	1.17	*(0.09)*	1.45	*(0.23)*
Canopy bulk density (kg/m³)	0.25	*(0.02)*	0.11	*(0.01)*

1. FVS output from stand exam data under non-outbreak conditions
2. FVS output from stands under western spruce budworm outbreak conditions

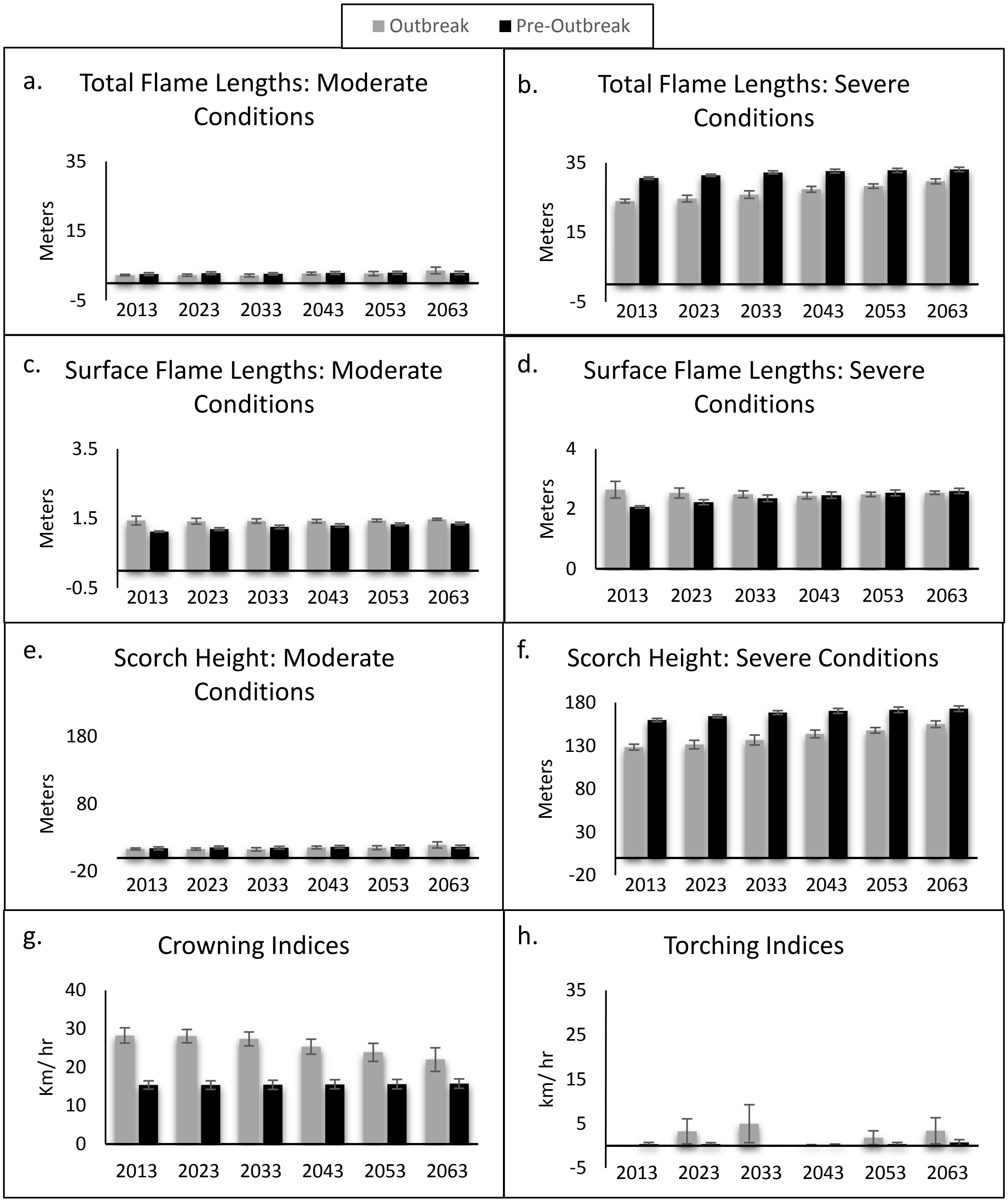

Figure 1: Potential fire behavior predicted from 2013-2063 using Fire and Fuels Effects- Forestry Vegetation Simulator (FFE-FVS). Pre-outbreak stands (n= 6) are stands which were sampled prior to western spruce budworm (WSBW) outbreaks. Outbreaks stands (n=4) were sampled in 2013 during a current outbreak. Points and lines represent means and standard errors at each ten year time step. 1a. Total flame lengths under moderate fire weather conditions (Table 1). 1b. Total flame lengths under severe fire weather conditions (Table 1). 1c. Surface flame lengths under moderate fire weather conditions. 1d. Surface flame lengths under moderate fire weather conditions. 1e. Scorch height under moderate fire weather conditions. 1f. Scorch height under severe fire weather conditions. 1g. Crowning indices. 1h. Torching indices

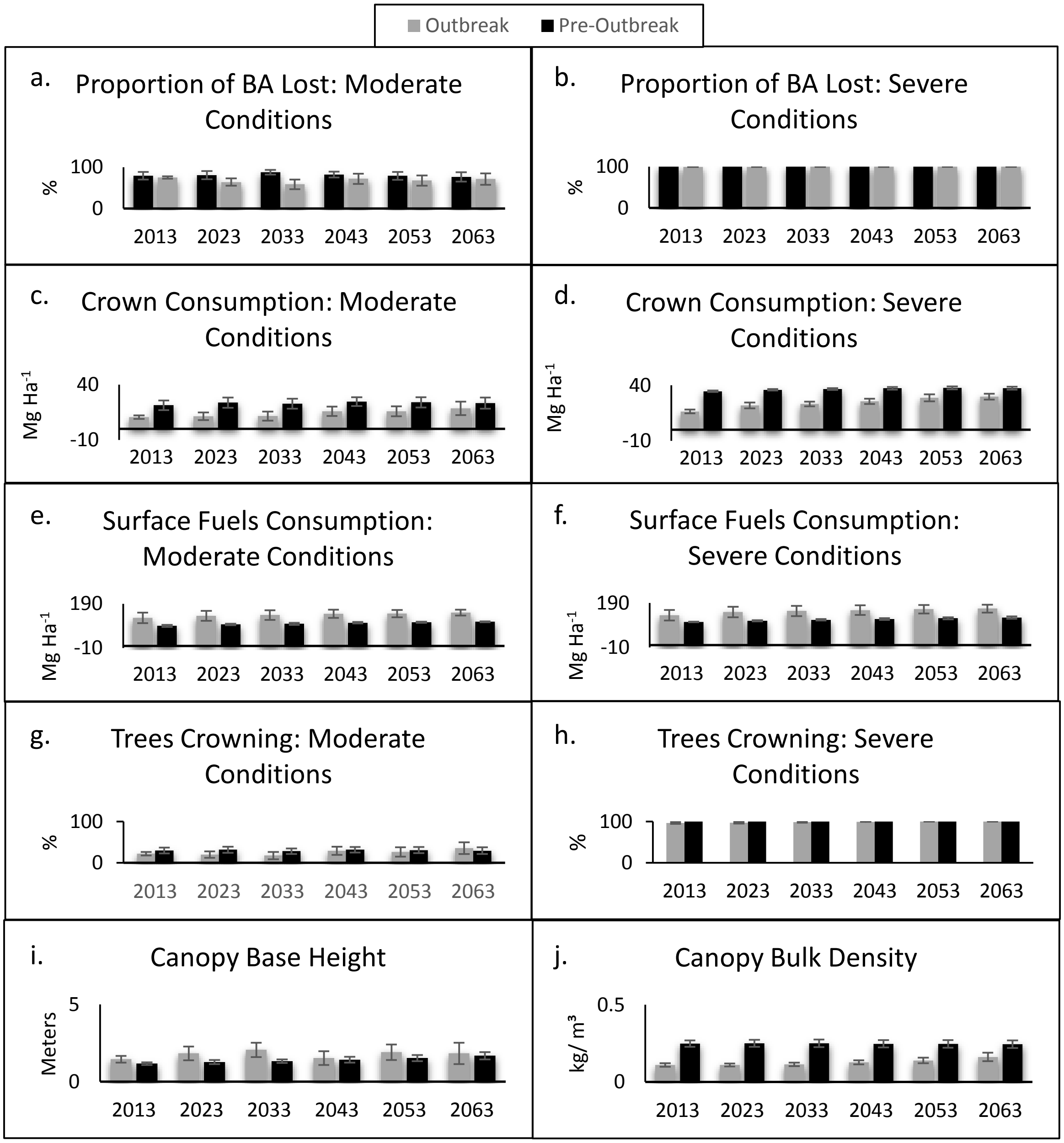

Figure 2: Potential fire behavior predicted from 2013-2063 using FFE-FVS. Pre-outbreak stands (n= 6) are stands which were sampled prior to WSBW outbreaks. Outbreaks stands (n=4) were sampled in 2013 during a current outbreak. Points and lines represent means and standard errors at each ten year time step. 1a. Total flame lengths under moderate fire weather conditions (Table 1) 2a. Proportion of BA lost under moderate fire weather conditions (Table 1). 2b. Proportion of BA lost under severe fire weather conditions. 2c. Total consumption of crown fuels (Mg ha^{-1}) under moderate fire weather conditions. 2d. Total consumption of crown fuels (Mg ha^{-1}) under severe fire weather conditions. 2e. Total consumption of surface fuels (Mg ha^{-1}) under moderate fire weather conditions. 2f. Total consumption of surface fuels (Mg ha^{-1}) under severe fire weather conditions. 2g. Height to the base of live canopy. 2h. Canopy bulk density as calculated by FFE-FVS.