Research Article

# The Influence of Western Spruce Budworm on Fire in Spruce-Fir Forests

**Eric Vane*, Kristen Waring, and Adam Polinko**

Northern Arizona University, School of Forestry,
200 East Pine Knoll Drive, Flagstaff, Arizona 86011, USA

*Corresponding author: Tel.: +1-734-223-3394; e-mail: Vane.Eric@gmail.com

## ABSTRACT

Western spruce budworm (*Choristoneura freemani* Razowski; WSBW) is the most significant defoliator of coniferous trees in the western United States. Despite its important influence on Western forests, there are still gaps in our knowledge of WSBW's impact on fire, and little research has been done on this relationship in high-elevation spruce-fir forests. Although this species is native to western North America, current outbreaks have persisted for many years in some areas, leading to high mortality rates among host tree species. Using the Fire and Fuels Extension of the Forest Vegetation Simulator (FFE-FVS), we examined the current and future impact of an ongoing WSBW outbreak on potential fire behavior in spruce-fir forests in northern New Mexico, USA. We found that this ongoing WSBW outbreak resulted in little to no difference in fire behavior under moderate fire weather conditions. Under high-severity fire weather conditions, stands enduring WSBW outbreaks experienced less severe fire behavior. We also found that WSBW outbreaks resulted in a decrease in fire behavior under severe fire weather conditions for 40 years following the outbreak.

## RESUMEN

La oruga de los pinos (*Choristoneura freemani* Razowski; WSBW) es la defoliadora más importante de las coníferas del oeste de los EEUU. A pesar de su importante influencia en los bosques del oeste, hay todavía vacíos en nuestro conocimiento sobre el impacto de la WSBW en los incendios forestales, y muy pocas investigaciones se han realizado sobre esta relación en bosques de altura dominados por picea y abies. Aunque esta especie de oruga es nativa del oeste de Norteamérica, los ataques corrientes de esta plaga han persistido por muchos años en algunas áreas, llevando consigo altas tasas de mortalidad a las especies arbóreas hospedantes. Usando la Extensión para Fuegos y Combustibles del Simulador de Vegetación Arbórea (FFE-FVS), examinamos el impacto actual y futuro de un ataque de WSBW sobre el comportamiento potencial del fuego en bosques de picea y abies en el norte de Nuevo México, EEUU. Encontramos que este ataque de la WSBW resultó en diferencias pequeñas o imperceptibles en el comportamiento del fuego bajo condiciones ambientales moderadas. Bajo condiciones ambientales de alta severidad el fuego, los rodales que soportaron los ataques de WSBW experimentaron un comportamiento del fuego menos severo. También encontramos que los ataques de WSBW resultaron en un decrecimiento en el comportamiento del fuego bajo condiciones ambientales severas por 40 años después del ataque.

*Keywords:* *Choristoneura freemani* Razowski, defoliator, fire behavior, fire severity, insect, New Mexico, outbreak

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## INTRODUCTION

Disturbances are a vital component of ecosystems that shape and maintain the natural landscape by creating opportunities for new individuals to become established, promoting structural heterogeneity, and allowing for a diversity of species to persist. However, alterations of disturbance characteristics, including introduction of novel disturbance agents (Whisenant 1990, Mayer and Swetnam 2000), changes to existing regimes (Covington *et al*. 1997, Veblen *et al.* 2000), and shifts in weather and precipitation patterns due to climate change may lead to undesirable ecological effects of disturbance (Williams and Liebhold 2002, Westerling *et al.* 2006, Waring *et al.* 2009, Boyd *et al.* 2013). Insect outbreaks and wildfire are considered two of the most important drivers of disturbance in North America, impacting millions of hectares annually (Potter and Conkling 2012, Canadian Forest Service 2013, NICC 2013).

The western spruce budworm (WSBW; *Choristoneura freemani* Razowski) is the most influential native defoliator of coniferous trees in the western United States (Fellin and Dewey 1982, Brookes *et al.* 1987). In 2013, WSBW outbreaks resulted in the mortality or defoliation of 121 001 hectares in the southwestern United States (USDA Forest Service 2015). Previous research indicates that current WSBW outbreaks throughout the western United States are lasting longer and are more severe than historic outbreaks (Anderson 1987; Swetnam and Lynch 1989, 1993; Swetnam *et al.* 1995). Such shifts are due to previous logging practices and fire exclusion, which resulted in dense stands with a high percentage of host tree species (e.g., true fir) (Anderson 1987; Swetnam and Lynch 1989, 1993; Swetnam *et al.* 1995). In addition to influences of fire exclusion and logging practices, it has been hypothesized that climate change will lead to increased WSBW activity (Flowers *et al.* 2014), and defoliated area will increase with elevated temperature and precipitation (Williams and Liebhold 1995).

Over the next century, climate change in the Southwest is expected to result in higher temperatures and more variable precipitation (Garfin 2013, IPCC 2014). 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, as described above, have resulted in forests that are more vulnerable to stand-replacing 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.

Western spruce budworm outbreaks have been shown to alter forest composition and fuel loading. Following outbreaks, there is an increase of non-host species composition due to mortality of host species. As a result of mortality, there is an increase in surface fuels as dead trees lose their needles and fall to the forest floor, while canopy fuels decrease as a result (Fellin and Dewey 1982, Brookes *et al.* 1987, Hummel and Agee 2003). While landscape-scale studies tend to show a decrease in the incidence of fire occurrence and intensity following WSBW outbreaks in mixed conifer forests (Lynch and Moorcroft 2008, Flower *et al*. 2014, Meigs *et al*. 2015), it is also important to understand fire behavior changes during and after outbreaks to better inform land management decisions. The influence of WSBW on fire, including potential fire behavior, has not been studied in spruce-fir forests to our knowledge. Focusing on the spruce-fir forest type in northern New Mexico, we investigated the influence of WSBW outbreaks on stand structure and potential fire behavior. Our specific research objective was to document current stand structure and model both stand structure and potential fire behavior in stands experiencing an ongoing WSBW outbreak as compared to stands not experiencing an outbreak. We assessed current stand structure, including fuels, using inventory data when possible. We assessed potential fire behavior (proportion of live basal area killed, crowning and torching indices, and flame length) and predicted fuels (canopy bulk density and canopy base height) using model outputs across both stand types over the course of 50 years, from 2013 to 2063.

## METHODS

We analyzed and modeled a total of 10 high-elevation spruce-fir stands from northern and central New Mexico, USA, for this study (Tables 1, 2, and 3). Data was available from 4 stands with current WSBW defoliation; the remaining six stands were similar but data was collected prior to the WSBW outbreak (see additional details below). All 10 stands were located on either the Cibola National Forest or the Carson National Forest, New Mexico. Of the four stands with current defoliation, two stands were located on the Cibola National Forest, and two stands were located on the Carson National Forest. Stands were selected using US Department of Agriculture (USDA) disease and insect aerial survey maps (Forest Health Protection 2012) and subsequent ground truthing (Polinko 2014).

Soils on the Carson National Forest are predominately Marosa cobbly sands and rock outcrops, while the Cibola National Forest contains Parkay complexes (USDA NRCS 2014). Stands were predominately north facing and had an average slope of 17°. Stand elevations on the Cibola National Forest ranged from 3118 m to 3173 m, while those on the Carson National Forest ranged from 3018 m to 3296 m. Mean annual precipitation ranges

**Table 1.** Stand level metrics (mean and SD) for four spruce-fir WSBW outbreak stands located in northern New Mexico, USA. Different letters indicate significant differences between stands using Wilcoxon rank sign test (α < 0.05); no letters indicate no significant differences between stands.

| Stand | Bunker Hill | Cold Spring | Mount Taylor | Bobcat Canyon |
|---|---|---|---|---|
| **Detected defoliation (yr)** | 8.7 (3.2)[a] | 9.7 (0.9)[a] | 11.6 (2.0)[b] | 12.2 (1.5)[b] |
| **Defoliation (%)** | 11.0 (4.2)[a] | 12.8 (3.0)[a] | 30.7 (17.2)[b] | 20.5 (7.0)[b] |
| **Elevation (m)** | 3287.6 (44.2)[a] | 3172.8 (47.1)[b] | 3114.8 (58.6)[c] | 3017.4 (21.4)[d] |
| **Slope (%)** | 20.2 (8.3) | 16.3 (7.0) | 19.8 (10.4) | 17.4 (6.7) |
| **Canopy cover (%)** | 51.3 (15.8) | 51.4 (13.1) | 57.6 (15.0) | 51.5 (17.3) |
| **Aspect (°)** | 131.4 (120.4) | 141.6 (147.0) | 126.7 (142.5) | 138.1 (6.7) |

**Table 2.** Mean basal area and trees per hectare for stands in northern New Mexico under western spruce budworm outbreak and pre-outbreak conditions. Pre-outbreak stand values are shown using Forest Vegetation Simulator outputs from stands grown to 2013. Outbreak stand values as reported in Polinko (2014) and calculated using 2013 outbreak stand data. Different letters indicate significant differences between pre-outbreak and outbreak stands using Wilcoxon rank sign test ($\alpha < 0.05$). Percent dead and live were not included in statistical comparisons. Fuel loads (Mg ha$^{-1}$) were calculated from stands in northern New Mexico under western spruce budworm outbreak (outbreak). Fuel loading data was not available for the pre-outbreak time period. Standard errors shown in parentheses.

| | Mean basal area (m$^2$ha$^{-1}$) | | | Condition | |
|---|---|---|---|---|---|
| **Stand type** | **Dead** | **Live** | **Total** | **Dead (%)** | **Live (%)** |
| Pre-outbreak ($n$ = 6) | 4.1$^a$ (2.1) | 57.8$^a$ (5.4) | 61.8$^a$ (7.4) | 6.5 | 93.5 |
| Outbreak ($n$ = 4) | 20.8$^b$ (8.2) | 30.9$^b$ (7.0) | 50.8$^a$ (14.9) | 40.9 | 60.9 |
| | **Trees per hectare** | | | **Condition** | |
| **Stand type** | **Dead** | **Live** | **Total** | **Dead (%)** | **Live (%)** |
| Pre-outbreak ($n$ = 6) | 205.2$^a$ (97.9) | 2016.7$^a$ (535.5) | 2221.9$^a$ (568.4) | 9.2 | 90.7 |
| Outbreak ($n$ = 4) | 478.0$^b$ (105.7) | 500.0$^b$ (76.8) | 978.0$^b$ (106.5) | 48.9 | 51.1 |

| | Fuel loading (Mg ha$^{-1}$) | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **Stand type** | **1-hour** | **10-hour** | **100-hour** | **1000-hour (sound)** | **1000-hour (decayed)** | **Litter** | **Duff** | **Total** |
| Outbreak ($n$ = 4 ) | 0.62 (0.15) | 2.31 (0.42) | 6.56 (1.50) | 124.11 (49.86) | 67.50 (30.30) | 18.90 (2.20) | 46.58 (5.03) | 266.57 (86.16) |

from 56.8 cm to 59.6 cm on the Cibola National Forest and 62.0 cm to 72.4 cm on the Carson National Forest. Averaged mean annual temperatures range from −12.5 °C to 21.0 °C on the Cibola National Forest, and −15.5 °C to 21.0 °C on the Carson National Forest in the coldest and warmest months (Crookston and Rehfeldt 2008).

Sampled stands had >50 % WSBW host species composition: Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), white fir (*Abies concolor* [Ford. & Glend.] Lindl. ex Hildebr.), corkbark fir (*Abies lasiocarpa* [Hook.] Nutt. var. *arizonica* [Merriam] Lemmon), and Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco). They had not received anthropogenic treatment in the previous 20 years, and were of similar slope and aspect; elevation varied between stands (Tables 1, 2 and 3; see also Polinko 2014). Records of fire history in these stands could not be found, but due to the lack of fire scars, presence of large diameter aspen (*Populous tremuloides* Michx.), and dense conditions, these stands did not appear to have experienced recent fire. Stands were relatively even-aged with widely scattered older trees. The majority of trees were between 50 yr and 100 yr, but some individuals ranged up to 137 yr (Polinko 2014).

To compare outbreak conditions to non-outbreak conditions, we attempted to find similar spruce-fir stands that were not experiencing WSBW outbreaks in high-elevation spruce-fir ecosystems in the Southwest, but none could be found in 2012 or 2013. We instead used stand inventory data collected by the USDA Forest Service from pre-outbreak time periods in six similar spruce-fir stands on

**Table 3.** Species composition as a percent of total basal area ($m^2ha^{-1}$), and trees per hectare for stands in northern New Mexico under western spruce budworm pre-outbreak and outbreak conditions. Pre-outbreak data shown using Forest Vegetation Simulator outputs from stands grown from stand inventory data to 2013. Outbreak stand values as reported in Polinko (2014) and calculated using 2013 outbreak stand data. Standard deviation shown in parentheses. Different letters indicate significant differences between pre-outbreak and outbreak stands using Wilcoxon rank sign test ($\alpha < 0.05$); no letters indicate no significant differences between stands. ABLAA = *Abies lasiocarpa* var. *arizonica,* PIEN = *Picea engelmannii,* POTR = *Populous tremuloides,* PSME = *Pseudotsuga menziesii,* JUSP = *Juniperus* spp., PIAR = *Pinus aristata,* PIFL = *Pinus flexilis,* ABCO = *Abies concolor.* ABCO, JUSP, PIAR, and PIFL were not included in statistical comparisons due to their presence in only <3 stands.

| **Percent total live basal area by species** | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **Stand Type** | **ABLAA** | **PIEN** | **POTR** | **PSME** | **JUSP** | **PIAR** | **PIFL** | **ABCO** |
| Pre-outbreak<br>(*n* = 6) | 48.92[a]<br>(6.2) | 38.99[a]<br>(9.5) | 4.25[a]<br>(7.9) | 8.19[a]<br>(10.9) | 0.0<br>(0.0) | 0.11<br>(0.1) | 0.0<br>(0.0) | 0.0<br>(0.0) |
| Outbreak<br>(*n* = 4) | 10.22[b]<br>(8.3) | 48.29[a]<br>(32.4) | 14.07[a]<br>(8.6) | 25.09[a]<br>(40.5) | 0.0<br>(0.0) | 0.0<br>(0.0) | 1.85<br>(2.4) | 0.47<br>(9.4) |
| **Percent total live trees per hectare by species** | | | | | | | | |
| **Stand type** | **ABLAA** | **PIEN** | **POTR** | **PSME** | **JUSP** | **PIAR** | **PIFL** | **ABCO** |
| Pre-outbreak<br>(*n* = 6) | 4.65[a]<br>(3.9) | 3.78[a]<br>(1.6) | 18.3[a]<br>(19.4) | 0.1[a]<br>(0.1) | 0.0<br>(0.0) | 0.0<br>(0.0) | 0.0<br>(0.0) | 0.0<br>(0.0) |
| Outbreak<br>(*n* = 4) | 42.02[a]<br>(38.4) | 48.02[a]<br>(34.9) | 33.53[b]<br>(16.1) | 7.56[a]<br>(9.2) | 50<br>(57.7) | 25<br>(50.0) | 0.0<br>(0.0) | 37.5<br>(47.9) |

the Cibola and Carson 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.

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 aerial surveys (USDA Forest Service 2015). The ongoing outbreak in northern New Mexico began in the early 1990s (USDA Forest Service 2014). Polinko (2014) found that all four currently defoliated stands had been defoliated for a minimum of 10 continuous years.

### *Field Measurements*

Stands experiencing WSBW outbreak were sampled in 2013 using a randomized, systematic grid of ten 0.02 ha plots to record overstory tree data as a part of a larger study (Polinko 2014). Five 0.001 ha regeneration plots were nested inside each of the overstory plots to quantify regeneration; these were located at the plot center and 3.95 m from plot center in each cardinal direction. Measurements taken on all overstory trees (trees with diameter at breast height [DBH; 1.37 m] >12.7 cm) included DBH, species, condition (live or dead), height, height to live crown base, and percent defoliation. Defoliation of live trees by WSBW was measured using techniques adapted from Millers *et al.* (1991; sketch method). Cumulative defoliation was sketched on a transparency using the guidelines of Schomaker *et al.* (2007) and later digitally scanned at 300 dots per inch. Pixels were counted using the analysis tools in Adobe® PhotoShop CS5. The proportion of defoliated crown area to total crown area was recorded and used to estimate percent canopy defoliation. Although the measurements are presented as a continuous variable, the method only reduces bias between individual assessments and is still prone to bias, which may be large (Hollands and Dyre 2000). Aspect, elevation, and defoliation ratings were compared between stands using Wilcoxon sign rank test with significance determined at $\alpha \leq 0.05$.

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).

Fuels data 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° azimuth. Fuels were tallied by time-lag size class: 1-hour fuels (0 cm to 0.63 cm) from 1.52 m to 3.35 m, 10-hour fuels (0.63 cm to 2.54 cm) from 1.52 m to 5.18 m, 100-hour fuels (2.54 cm to 7.62 cm) from 1.52 m to 16.7 m, and 1000-hour fuels (>7.62 cm) from 1.52 m to 16.7 m. For 1000-hour fuels, diameter was recorded along with decay class on a scale from 1 to 5 (1 showing no sign of decay and 5 being extremely decayed; Maser *et al.* 1979). Duff and litter depths were taken at 1.52 m, 6.09 m, 10.66 m, and 15.24 m along each transect. Fuel loading was calculated on a scale of megagrams per hectare, and 1000-hour fuels were split into decayed (decay class 4 and 5), and sound (decay class 1, 2, and 3) classes. To calculate fine woody debris fuel load, we used specific gravities and average squared diameter values published in Sackett (1980) when possible, and values from van Wagtendonk *et al.* (1996) and Green *et al.* (1999) for species not included in Sackett (1980). For sound 1000-hour fuel load calculations, we used specific gravity values published in Green *et al.* (1999). We assumed an average specific gravity of 0.30 for decayed 1000-hour fuels (Brown 1974). Calculations were weighted by standing species composition because species for 1000-hour fuels were not recorded on site. We used average Sierra Nevada mixed-conifer litter and duff values (van Wagtendonk *et al.* 1998) because none were available for the Southwest.

Stand inventory data were provided by the USDA Forest Service for six stands visited prior to the start of the WSBW outbreak; inventory data provided were from the late 1980s and early 1990s. Inventory data included individual overstory tree data (species, DBH, total tree height, and live crown ratio) and understory tree data (species and trees per unit area). Stand level data provided included geographic coordinates, aspect, slope, and elevation. To obtain current conditions for pre-outbreak stands, we modeled the stands from the original inventory date to 2013 and extracted stand-level metrics from the FVS outputs. We calculated mean basal area (BA; $m^2\ ha^{-1}$), trees per hectare (TPH), and percent species composition for each stand and averaged across stands by stand type (pre-outbreak, outbreak). We also calculated mean percent species composition for live trees using BA and for dead trees using TPH; these were also averaged across stands by stand type. We compared means between stand types (pre-outbreak and outbreak) using Wilcoxon signed rank tests (significance set at $P \leq 0.05$) except for fuels data, for which we did not have enough available information to conduct statistical comparisons.

### Stand Structure and Fire Behavior Model Parameters

We used the Forest Vegetation Simulator (FVS), Central Rockies (CR) variant, and the Fire and Fuels Extension (FFE) (Keyser and Dixon 2008, Rebain *et al.* 2013) to model stand structure and fire behavior and assess differences between outbreak and pre-outbreak conditions. The FVS is a distant-independent, individual-tree forest growth and yield model that is widely used throughout the United States (Crookston and Dixon 2005). The Fire and Fuels Extension is a fire behavior model that utilizes the Rothermel (1972) surface fire model in conjunction with the Van Wagner (1977) crown fire initiation model. In addition, FFE-FVS uses fuel models described in Anderson (1982) and Scott and Burgan (2005) to model prescribed fire and wildfire in stands inputted to FVS. Regeneration is not initiated by default into FVS so we used seedling and sapling data collected during field work in outbreak stands to initiate regeneration in 2023, and 30 years later in 2053 (corkbark fir: 1235 TPH, Engelmann spruce: 346 TPH, Douglas-fir: 173 TPH, and quaking aspen: 1235 TPH). Standard FFE-FVS output for each 10-year time step includes TPH, basal area, mean tree size, crown base height, canopy bulk density, torching and crowning indices, surface flame lengths, and surface fuel loading.

We initiated FVS using both severe (ninety-seventh percentile) and moderate (eightieth percentile) fire weather data that was derived from three remotely activated weather stations (RAWS), deemed appropriate based on their elevation and proximity to the study areas. Fire weather was assessed from 1 April to 30 September beginning in 1992 (first year of available RAWS data) to 2016 (Table 4). We determined seasonality for our fire weather assessment using previous fire occurrence data for the Santa Fe, Cibola, and Carson national forests, which indicated that the majority of fires (>5 acres) occur from April to September (Bradshaw and McCormick 2000). Because fuel models in FFE are calculated based on the relationship between small fuels and large fuels (Rebain *et al*. 2013) in the CR variant, we selected two fuel models that were more appropriate in the timber litter (TL) fuel type category, based on fuel profiles witnessed in the field in outbreak stands: TL5 (high load conifer litter) and TL7 (large downed logs). Fire is

**Table 4.** Moderate and severe fire weather conditions used in the Fire and Fuels Extension-Forest Vegetation Simulator to model fire behavior. Information derived from Coyote, Jarita Mesa, and Brushy Mountain Remote Automated Weather Stations (RAWS) in New Mexico (http://raws.wrh.noaa.gov/roman/index.html) unless otherwise noted.

| | Moisture (%) | | |
|---|---|---|---|
| **Fuel type** | **Moderate (80$^{th}$ percentile)** | **Severe (97$^{th}$ percentile)** | **Source** |
| Duff | 50 | 15 | Rebain *et al.* 2013 |
| 1-hour fuels | 3 | 1 | RAWS |
| 10-hour fuels | 4 | 3 | RAWS |
| 100-hour fuels | 6 | 3 | RAWS |
| 1000-hour fuels (sound) | 9 | 4 | RAWS |
| Herbage | 17 | 3 | RAWS |
| Woody | 60 | 60 | RAWS |
| | **Fire weather** | | |
| | **Moderate** | **Severe** | |
| Wind speed (km hr$^{-1}$) | 11 | 27 | RAWS |
| Temperature (°C) | 19 | 27 | RAWS |

carried by high loads of conifer litter and slash in TL5 and by large downed logs in TL7. Rate of spread and flame length is low for both fuel models; however, the rate of spread is lower in TL7 than in TL5 and flame lengths are initially slightly higher in TL7 than in TL5 under low wind conditions, but as wind speed increases, flame lengths in TL5 become greater than in TL7 (T. Nicolet, USDA Forest Service, Payson, Arizona, USA, personal communication). We applied different weights to the two fuel models in pre-outbreak and outbreak stands, based on fuel profiles witnessed in the field (pre-outbreak: 85 % TL5, 15 % TL7; outbreak: 60 % TL5, 40 % TL7).

### *Fire Behavior Modeling*

Stand level data (overstory trees, seedlings, and saplings) were input into FVS for both pre-outbreak and outbreak stands, along with fuels data from outbreak stands. Prior to initiating model runs, pre-outbreak stands were projected forward from the initial inventory data to 2013 in FVS so that the impact of WSBW on fire behavior could be examined over the same time period as outbreak stands. Using FFE-FVS, we looked at the potential impact and severity of fire on pre-outbreak and outbreak stands under moderate and severe fire weather conditions every decade over a 50-year timespan, beginning in 2013 and ending in 2063. Potential fire reports were generated for all pre-outbreak and outbreak stands under moderate and severe fire weather conditions in order to determine predicted tree mortality, flame length, torching and crowning indices (wind speed, measured at 6.09 m above open ground, required to initiate individual tree torching or active crown fire), as well as canopy base height (CBH) and canopy bulk density (CBD). No statistical comparisons were conducted on the results of the fire behavior modeling.

## RESULTS

### *Current Conditions*

In stands currently experiencing WSBW outbreak, dead trees accounted for nearly half of the standing basal area (BA, $m^2$ $ha^{-1}$; Table 2). Engelmann spruce accounted for the greatest percentage (48 %) of live BA, followed by Douglas-fir, aspen, corkbark fir, limber pine (*Pinus flexilis* James), and white fir (Table 3). In contrast, pre-outbreak stands grown to 2013 in FVS were composed of >90 % live BA and TPH (Table 2). In pre-outbreak stands, corkbark fir accounted for the greatest percentage (49 %) of live BA, followed by Engelmann spruce, Douglas-fir, aspen, and Rocky Mountain bristlecone pine (*Pinus aristata* Engelm.) (Table 3).

Total surface fuel loading in outbreak stands was greater than 250 Mg $ha^{-1}$, with the majority (71 %) found in sound and decayed 1000-hour fuels. Duff and litter fuel loads accounted for almost 25 % of the total, while 1-hour, 10-hour, and 100-hour fuels accounted for <5 % of the total fuel load (Table 2).

Canopy base height (CBH) was nearly identical at about one meter in pre-outbreak and outbreak stands. Canopy bulk density (CBD) was over twice as dense in pre-outbreak stands as that of outbreak stands (Table 5).

### *Current Potential Fire Behavior*

Fire modeled in 2013—the year data was collected—resulted in mortality of more than a third of live BA under moderate fire weather conditions, and nearly half under severe fire weather conditions in outbreak stands. Pre-outbreak stands that were grown to 2013 in FVS sustained over 40 % BA mortality under moderate fire weather conditions, and resulted in a loss of all live BA under severe fire weather conditions. Torching indices were greater than moderate and severe fire weather

**Table 5.** Potential fire behavior and predicted canopy fuels for current (2013) conditions in northern New Mexico under western spruce budworm outbreak (outbreak) and non-outbreak (pre-outbreak) conditions, as modeled in the Fire and Fuels Effects-Forest Vegetation Simulator. Standard deviation shown in parentheses.

| | Potential fire behavior | |
|---|---|---|
| | **Pre-outbreak[1] ($n$ = 6)** | **Outbreak[2] ($n$ = 4)** |
| **Mortality (% of basal area)** | | |
| **Moderate** | 41.17 (5.7) | 36 (9.52) |
| **Severe** | 100.00 (0) | 48 (29.9) |
| **Flame length (m)** | | |
| **Moderate** | 0.36 (0.01) | 0.43 (0.01) |
| **Severe** | 24.26 (1.14) | 5.51 (8.3) |
| **Indices (km hr$^{-1}$)** | | |
| **Crowning** | 15.44 (2.68) | 29.33 (4.84) |
| **Torching** | 65.05 (16.56) | 48.95 (27.6) |
| **Predicted canopy fuels** | | |
| **Canopy bulk density (kg m$^{-3}$)** | 0.25 (0.05) | 0.11 (0.02) |
| **Canopy base height (m)** | 1.23 (0.25) | 1.23 (0.58) |

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

condition wind speeds, indicating that torching is unlikely. Crowning indices were less than torching indices for pre-outbreak and outbreak stands, and pre-outbreak stands had lower crowning indices than outbreak stands (Table 5).

### *Future Stand Structure and Potential Fire Behavior*

Canopy bulk density was higher in pre-outbreak stands than outbreak stands for the entire 50-year time frame. Over time, CBD in pre-outbreak stands decreased slightly, while outbreak stands increased (Figure 1e). Canopy base height varied throughout the 50-year model run and was similar between stand types (Figure 1f).

Over the 50-year time span of the model run, live BA mortality under moderate fire weather conditions remained mostly unchanged around 40 %, and there was little to no difference between pre-outbreak and outbreak stands (Figure 1a). Live BA mortality resulting from severe fire weather conditions remained at 100 % for pre-outbreak stands for the entire model run, while outbreak stands initially sustained about 50 % BA mortality, increasing to almost 100 %, by 2063 (Figure 1b).

Crowning indices were nearly 30 km hr$^{-1}$ in 2013 for outbreak stands and decreased to less than 25 km hr$^{-1}$ by 2063. Conversely, crowning indices for pre-outbreak stands were nearly 15 km hr$^{-1}$ in 2013, and slightly increased over the 50-year model run to 17 km hr$^{-1}$ (Figure 1c). Torching indices were variable throughout the 50-year model run, increasing from present day to 2023, decreasing in 2033, increasing again until 2053, and decreasing to its lowest level in 2063. Outbreak stands had lower torching indices than pre-outbreak stands in 2013, but were similar or higher for the remainder of the model run (Figure 1d).

Flame lengths under moderate fire weather conditions remained mostly unchanged

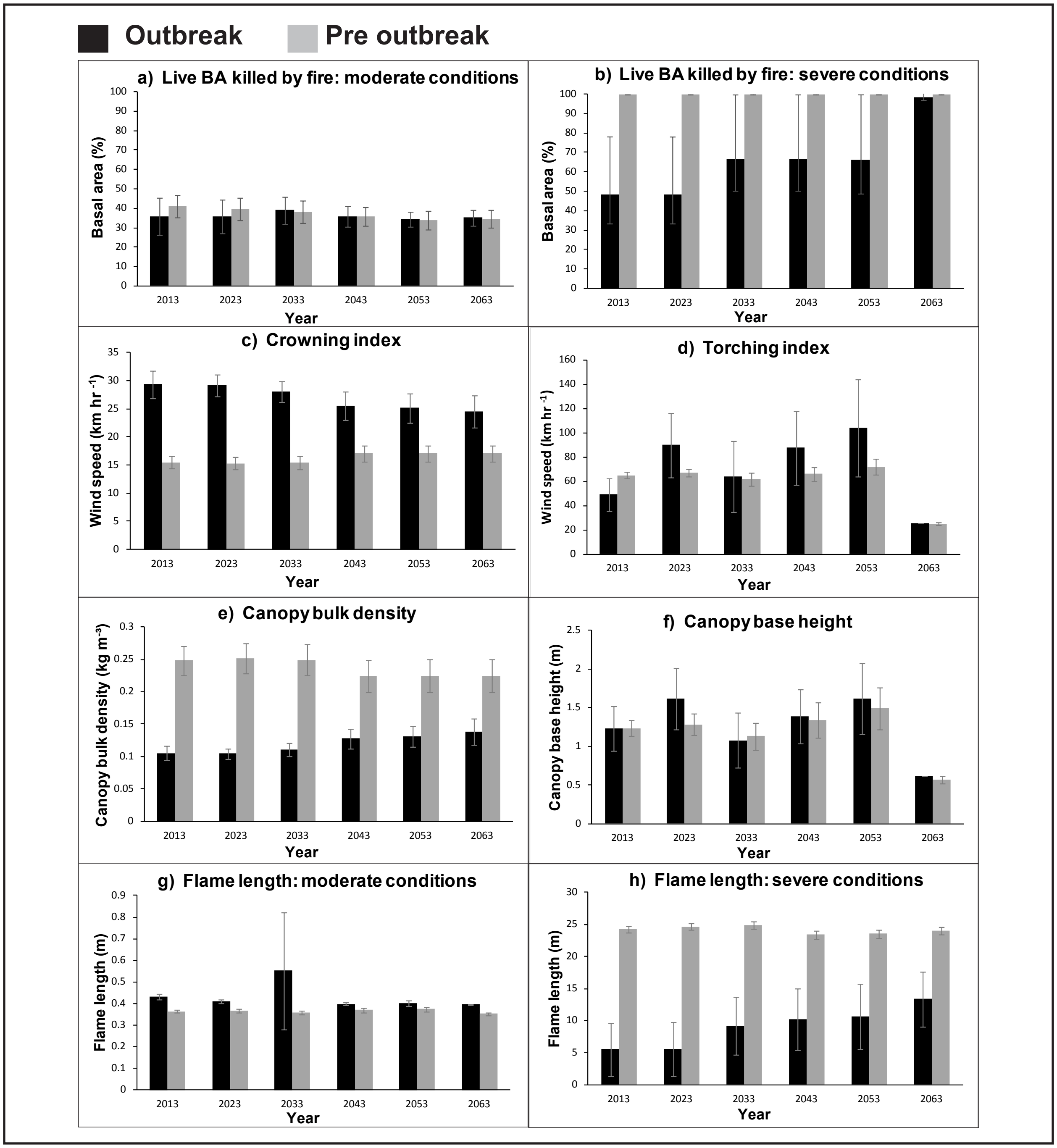


**Figure 1.** Potential fire behavior predicted from 2013 to 2063 using the Fire and Fuels Effects-Forest Vegetation Simulator (FFE-FVS). Pre-outbreak stands ($n$ = 6) are stands that were sampled prior to western spruce budworm (WSBW) outbreaks. Outbreaks stands ($n$ = 4) were sampled in 2013 during a current outbreak. Bars represent the means and lines represent standard deviation at 10-year intervals. Proportion of live basal area experiencing mortality as a result of wildfire under moderate fire weather conditions (a) and severe fire weather conditions (b). Crowning indices or wind speed required to result in a canopy fire based on stand conditions (c), and torching indices or wind speed required for torching of individual trees to occur (d). Canopy bulk density or the density of fuels in the canopy (e), and canopy base height, distance from the forest floor to the base of the continuous canopy (f). Flame lengths under moderate (g) and severe (h) fire weather condition.

throughout time for both pre-outbreak and outbreak stands at around 0.4 m, with an increase in flame length in outbreak stands for one decade, 2033 (Figure 1g). Under severe fire weather conditions, flame lengths were much higher in pre-outbreak stands than outbreak stands. Pre-outbreak stands had flame lengths of nearly 25 m, which remained mostly unchanged over time, while outbreak stands had flame lengths of 5 m, which increased to nearly 14 m by 2063 (Figure 1h).

## DISCUSSION

Our fire behavior (proportion of live BA killed, crowning and torching indices, and flame length) and fuels (canopy bulk density and base height) results indicate that pre-outbreak stands are highly vulnerable to stand-replacing fire under severe fire weather conditions, while outbreak stands also sustain substantial mortality, which increases through time. Spruce-fir ecosystems tend to experience a high-severity, low-frequency fire regime, regardless of insect presence, due to their dense stand structure and species' silvics (Arno 1980, Agee 1993, Margolis *et al.* 2011). Spruce and fir species tend to have shallow roots, 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 1993), which results in abundant ladder fuels, dense stands, and high surface fuel loading (Alexander 1987). Therefore, high-elevation spruce-fir stands that have not been recently logged or experienced fire tend to support fuel structures that are highly susceptible to stand-replacing, high-severity fire (Arno 1980, Alexander 1987, Agee 1993). Due to the persistent cool, wet conditions, severe fires in this ecosystem typically occur in the late summer and early fall during drought years when the snowpack is low and warm, dry conditions reduce fuel moisture content. In particular, these drought conditions occur in the Southwest when the El Niño Southern Oscillation and Pacific Decadal Oscillation are in their cool phases (Margolis and Swetnam 2013). Our results show that there is a greater difference in fire severity between moderate and severe fire weather conditions than between outbreak and pre-outbreak conditions, regardless of time period. This coincides with the other research done in this ecosystem that has found the fire regime in spruce-fir forests to be climate driven (Agee and Smith 1984, Agee 1993, Kulakowski *et al.* 2003).

Western spruce budworm in northern New Mexico has altered conditions of spruce-fir stands, increasing the amount of standing dead TPH and BA while decreasing live BA and TPH. Dead trees in the outbreak stands both remain standing and have begun falling, resulting in a high amount of 1000-hour fuels relative to the total surface fuel load (70%) and higher levels of standing dead BA and TPH in WSBW-affected stands. In addition, species composition of live trees has shifted from predominantly corkbark fir and Engelmann spruce, to Engelmann spruce, Douglas-fir, and aspen. Fuel loading data reported for Utah spruce-fir stands, in stands not experiencing a WSBW outbreak (Jorgensen and Jenkins 2010), was approximately 3 times less (at 84 Mg ha$^{-1}$) than our outbreak stands (at 267 Mg ha$^{-1}$). Sound and decayed 1000-hour fuels accounted for half of the total fuel load in the Utah stands and just over 70% in our outbreak stands. As dead trees continue to fall in the outbreak stands, 1000-hour fuels will continue to account for most of the surface fuels. Insect outbreaks, which disrupt the continuity of crowns, create gaps that may slow or limit growth of active crown fires. Decreases in canopy bulk density can also result in decreased torching of individual trees and subsequent crown fire occurrence and spread (Cohn *et al.* 2015). Outbreaks of WSBW result in greater mortality of small-diameter understory trees (Brooks *et al.* 1987), which may increase stand-level CBH, reducing the likelihood of torching. While this was not reflected in our study, it is likely that increased mortality of understory trees would result in higher CBH in other stands.

Previous research in the western United States on the interaction between insects and fire in spruce-fir forests have focused on the impact of spruce beetle (*Dendroctonus rufipennis* Kirby). Unlike WSBW outbreaks, spruce beetle outbreaks primarily cause mortality in large-diameter spruce and shift species composition to true fir and pine with high rates of spruce mortality (Veblen *et al.* 2000, Bebi *et al*. 2003, Kulakowski *et al.* 2003, Kulakowski and Veblen 2007). Outbreaks of spruce beetles in this ecosystem decrease live BA and canopy bulk density while increasing surface fuel loading and CBH (Jorgensen and Jenkins 2010). Active crown fire potential may also be reduced for decades following stand-level spruce beetle outbreaks (DeRose and Long 2009) or may have a weaker influence on fire severity than other factors (Andrus *et al.* 2016). These results are similar to trends found in our study that also show decreases in live BA and CBD, and increases in CBH and surface fuel loads.

Western spruce budworm outbreaks can have a significant impact on certain fire metrics and fuel loading in lower elevation forests. Hummel and Agee (2003) found that surface fuel loading and flame lengths are greater in stands at the end of a WSBW outbreak than those at the onset in mixed conifer forests. However, no significant differences were found in BA or TPH mortality due to fire from early to late outbreak conditions. This is contrary to our study, which has shown that, in spruce-fir forests, WSBW outbreaks can result in decreased flame lengths and BA mortality as compared to pre-outbreak conditions under severe fire weather conditions. However, the fire regime and stand structure of spruce-fir forests are characterized by heavy fuel loads and dense stand structures that support high severity stand-replacing fires regardless of WSBW activity.

The impact of WSBW defoliation on a single tree and its potential to torch or transfer heat is important in understanding the effect that defoliation has on fire spread in a stand. Cohn *et al.* (2014) found that >50 % defoliation of a tree by WSBW was found to transfer only 60 % of the heat as compared to heat transfer from trees that were not defoliated, and the greater the defoliation, the less heat transfer occurred. In addition, the amount of heat needed to ignite or torch a 50 % defoliated tree was greater than for an undefoliated tree (Cohn *et al*. 2014). The reduction of CBD by WSBW can thus hamper a fire's ability to kill individual trees and spread through the canopy. The FFE-FVS does not have the ability to model these fine-scale interactions between partial defoliation and fire spread. This suggests that, under outbreak conditions, potential fire behavior may be even more reduced than our model outputs show. However, FFE-FVS does model decreases in CBD due to WSBW-caused mortality; pre-outbreak stands consistently have higher CBDs.

Our study focused on the stand scale, providing information at a scale important to forest managers needing to prioritize stands for management activities. However, modeling exercises such as the one conducted for this study are constrained by modeling limitations, and need to be contrasted with observations and fire behavior analysis on the ground. Fire behavior models, including FVS-FFE, have a tendency to under-predict some metrics used to predict fire behavior, such as crowning index (Cruz and Alexander 2010). Adjustments to the default parameters for fire weather (e.g., fuel moisture, wind speeds) and selecting fuel models can help alleviate (but not eliminate) model bias (Cruz and Alexander 2010). We used the nearest RAWS weather stations available, which were lower in elevation than our specific stands. Weather data that better represents stand conditions would likely reduce model bias. Specific to FVS-FFE, initial stand data and regeneration assumptions will influence stand structural characteristics, such as CBD and CBH, through time. Regeneration was periodically included using actual stand data; we believe that is realistic in this forest type, given the silvics of the species involved and lack of stand-replacing disturbance. However, as trees continue to succumb to WSBW and climate shifts, regeneration, both the abundance and species present, is likely to shift as well, which will affect fire behavior (Tinkham *et al.* 2016). The current WSBW outbreak has been ongoing for 10 to 20 years in northern New Mexico. In our model runs, we assumed that the outbreak ended in 2013 and included no additional growth reduction or mortality from WSBW. Removing this assumption, including a reduction in regeneration, would likely yield predictions of fire behavior that move farther away from those of pre-outbreak stands. Finally, additional research is needed to confirm our results, given our small sample size ($n = 4$ outbreak stands).

While historic fire regimes in high-elevation spruce-fir forests followed a high-intensity, stand-replacing, low-frequency pattern, these patterns may shift under new climatic conditions. In the Southwest, climate change is predicted to result in higher temperatures and more variable moisture conditions along with increasing frequency and severity of drought events (Woodhouse *et al.* 2010, Garfin 2013, IPCC 2014). Although moisture is difficult to accurately model, precipitation patterns are predicted to shift away from snowfall, which will reduce snowpack size and longevity (IPCC 2014). This could alter the fire regime in spruce-fir forests (which typically only experience fire infrequently in drought years with low snowpacks [Agee 1993]) by increasing the instances of these advantageous fire conditions. Meigs *et al.* (2015) found that, while there was an overall decrease in fire occurrence following WSBW outbreaks, there was a higher likelihood of fire following outbreaks in extreme climate years. Increases in high-severity, stand-replacing fires could significantly impact human settlements and ecosystem function. Although it is outside the scope of this research, alterations in stand structure due to insect-caused mortality may result in decreased fuel moisture content due to increased solar radiation and alterations to wind flow and fire spread (Hoffman *et al.* 2015). We observed a general lack of robust regeneration in outbreak stands and little understory vegetation. Additional research should be conducted to determine if open conditions lead to more rapid desiccation of fuels in spruce-fir forests and increase the risk of successful fire ignitions and enhance fire's ability to spread in these pockets.

## CONCLUSION

When discussing the impacts of forest insects with forest managers, it is often assumed that high levels of mortality from insects result in an increase in crown fire behavior. Our results support the results of previous research in other forest types that WSBW outbreaks result in decreased fire behavior, even though high levels of mortality are still predicted. It is important to note the relatively large differences in fire behavior between moderate and severe fire weather that reinforce research that has shown fire to be climate driven in this ecosystem. However, as the discrepancy between outbreak and pre-outbreak stands shows, under severe fire weather conditions, canopy fuels play an important role as well. While there are differences between outbreak and pre-outbreak stands, fire under both moderate and severe fire weather behavior still results in substantial BA mortality, which indicates that WSBW outbreaks have not yet shifted spruce-fir forests away from a moderate-severity to high-severity fire regime.

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