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Western Spruce Budworm Defoliation Effects on Forest Structure and Potential Fire Behavior

Abstract

Forest composition and structure on the eastern slope of the Cascade Mountains have been influenced by decades of fire exclusion. Multi-layered canopies and high numbers of shade-tolerant true fir trees interact with western spruce budworm to alter forest structure and to affect potential fire behavior and effects. We compared measurements taken in 1992 (early budworm outbreak) and 2000 (late budworm outbreak) from 21 permanent plots located in a late successional reserve south of Mt. Adams in Washington. Canopy closure decreased significantly, from a mean of 78% in 1992 to 43% in 2000, but the coarse woody debris load increased significantly during the same period, from about 40 Mg ha^{-1} to 80 Mg ha^{-1}. Tree mortality was mostly in the smaller (<20 cm) diameter classes. Potential surface fire flame lengths increased significantly from 1.4 m in 1992 to 1.9 m in 2000, but changes in torching potential and independent crown fire behavior were not significant. Projections using the First Order Fire Effects Model indicate that a wildfire in conditions similar to those in 2000 would not be of stand replacement severity and would leave 148 trees ha^{-1} and more than 34 m^2 ha^{-1} of basal area.

Introduction

Biological and social concerns about the persistence of species associated with old forests of the U. S. Pacific Northwest helped to create a regional forest reserve network in 1994 (USDA and USDI 1994). These reserves-called late successional reserves (LSR)-are located on federal land throughout the range of the northern spotted owl (*Strix occidentalis caurina*) in Washington, Oregon, and California (Figure 1). One LSR objective is to protect late successional forest from large-scale fire, insect and disease epidemics, and major human impacts (USDA and USDI 1994). Here we evaluate how insect and fire interaction may affect this objective for a LSR in the eastern Washington Cascades physiographic province (Figure 1).

Forest structure and species composition are shaped by disturbance history and, in turn, influence disturbance dynamics. The mid-elevation, mixed-conifer forest on the east slope of the Cascade Mountains has a variable-intensity, mixed-severity fire regime (Agee 1993). Low-intensity surface fires favor dominance by ponderosa pine (*Pinus ponderosa*), western larch (*Larix occidentalis*), and Douglas-fir (*Pseudotsuga menziesii*), which have thick, fire-resistant bark (Arno et al. 1985, Prance and Prance 1993). Low- or moderate-intensity fires favor seral tree species such as Douglas-fir and western larch, over species like grand fir (*Abies grandis*), which has thinner bark and is thus more easily killed by fire (Agee 1993). Federal fire exclusion policies dating to the early 20th century (Dana and Fairfax 1980) have extended fire-free intervals in western states (Covington et al. 1994), and in the mid-elevation forests on the eastern slope of the Cascade Mountains this has resulted in higher numbers of shade-tolerant, fire-intolerant grand fir (Camp 1999).

Changes in species composition have been accompanied by structural changes in the amount and distribution of foliage and of dead or dying trees. Higher biomass of late successional trees like grand fir (Brookes et al. 1985) and more canopy layers (Brookes et al. 1987) increase forest susceptibility to outbreaks of insects such as the western spruce budworm (*Choristoneura occidentalis*) (Hessburg et al. 1994, Swetnam and Lynch 1993, Swetnam et al.1995). Trees may die if spruce budworm defoliation persists over several years (Edmonds et al. 2000), which contributes dead wood to the structural complexity and ecological function of the forest

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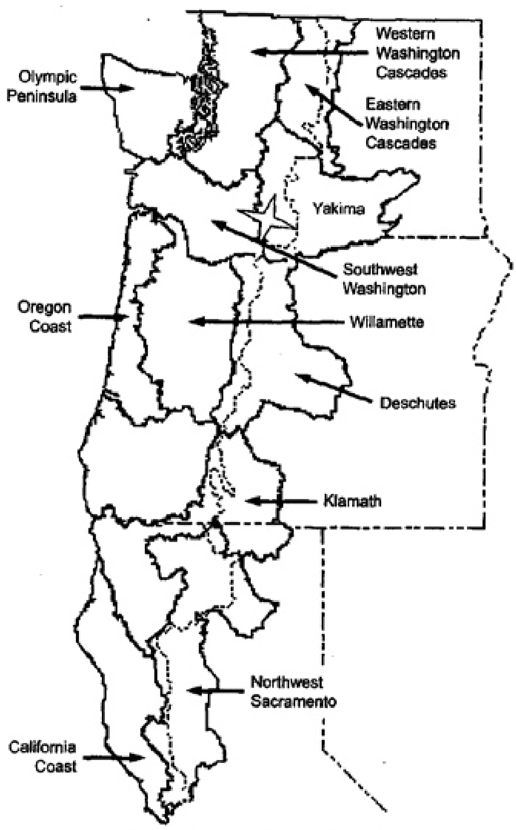

Figure 1. Physiographic provinces within the range of the northern spotted owl (source: FEMAT 1993). Star identifies location of this study.

(McComb and Lindenmayer 1999). Tree size and age diversity, canopy foliage distribution, and dead wood are all important components of forest structure (Spies 1998).

Forest structure interacts with fire behavior, which is a function of fuels, weather, and topography (Agee 1996). A fire may move through a forest as a surface fire, an independent crown fire, or some combination thereof (Van Wagner 1977). Intense surface fire may transition into a crown fire through a process called torching, which is a function of the height to live crown and foliar moisture content. Above threshold conditions of fireline intensity defined by these variables, the fire will move into the canopy. Independent crown fire spread is a function of fire rate of spread and the foliar density of the canopy (Scott and Reinhardt 2001).

In late successional reserves, developing or maintaining forest structure that supports species like the spotted owl is a key goal (USDA and USDI 1994). Large trees, downed wood, and dense, multilayered forest canopies are important because these features support owl nesting and foraging activities (Thomas et al. 1990). Spotted owl nest sites on the eastern side of the Cascade Mountains are reported to have a broader array of structural conditions than on the western side (Everett et al.1992), but common elements include closed canopy forests with large Douglas-fir trees, downed wood, and multiple canopy layers (Buchanan et al. 1995, King et al. 1997). Depending on their magnitude and intensity, budworm defoliation and fire might affect these elements of owl habitat in different ways. The spruce budworm modifies forest structure by increasing mortality of defoliated understory trees, by predisposing larger trees to attack by bark beetles (*Dendroctonus* or *Scolytus* spp.), and by altering the amount of canopy foliage (Hadley and Veblen 1993). The duration and frequency of budworm outbreaks have increased with decreasing forest fire frequency in the western states (Anderson et al. 1987). Fire modifies forest structure along a gradient, thinning smaller to larger trees and thin- to thicker-barked species as fireline intensity increases. We investigated the interaction of budworm and fire by measuring how live and dead forest structure is changing in association with budworm defoliation and by simulating how these changes affect potential fire behavior and effects.

Study Site

Our study site is in south-central Washington state in the Gifford Pinchot National Forest (Hummel et al. 2001) (Figure 1). The approximately 80-ha site covers Smith Butte, a cinder cone related to Mt. Adams that lies 13 km south of the mountain and about 3 km west of the Yakama Indian Reservation. The elevation of Smith Butte ranges from 1120 to 1300 m and slopes, which average 22°, range from 10° to 31°. The volcanic parent materials are covered by coarse-textured, sandy loam soils ranging from 0.15-1.2 m deep (Gifford Pinchot National Forest Soil Resource Inventory 1971). Vegetation varies by aspect and elevation, but generally resembles the grand fir/creeping snowberry/vanillaleaf (*Abies grandis/Symphoricarpos mollis/Achlys triphylla*) or grand fir/big huckleberry (*Abies grandis/Vaccinium membranaceum*) associations described in Franklin and Dyrness (1988) and Topik (1989).

Several species of conifer root pathogens occur in scattered pockets across the study site (Filip 1980). The nearest weather records are from the USDA Forest Service Mt. Adams Ranger Station in Trout Lake, Washington, which is located about 10 km southwest and 600 m lower than Smith Butte. Trout Lake receives about 116 cm of annual precipitation, and snow generally covers Smith Butte from November to April.

Current forest conditions on Smith Butte are atypical for the area because, although Douglas-fir, ponderosa pine, and western larch have been harvested from surrounding forest, the butte has not been logged. A map from the 1940s shows the forest types on Smith Butte by aspect: mature Douglas-fir on the west flank, large-pole ponderosa pine on the south flank, and mixed conifer on the east flank. These forest types, which the map illustrates but does not explicitly define, were roughly equal in size (20-28 ha) (Bright 1941). Current conditions are a legacy of these relict forest types, and the butte is, therefore, a type of refugium (Camp et al. 1997) where research opportunities abound. Indeed, Smith Butte was considered for inclusion in a network of Research Natural Areas (RNA) (Franklin et al. 1972) and a grid of 21 sample points was established in 1992. Although vegetation conditions at each point were measured and the center location permanently marked, Smith Butte never became a designated RNA. Efforts have been revived to obtain RNA status, in part because a meadow on top of the butte provides habitat for the mardon skipper (*Polites mardon*), a butterfly listed as endangered by the state of Washington (Potter et al. 1999).

In 1994, Smith Butte was included in the 6000 ha Gotchen LSR and became subject to the Gifford Pinchot National Forest objectives for the reserve (USDA Forest Service 1997). Also in 1994, annual aerial detection surveys began recording a budworm outbreak in the Gotchen LSR. Budworm had been increasing from endemic levels since the late 1980s, and by 1992 defoliation was apparent from the ground (Jim White, USDA Forest Service, personal communication 2000). Annual defoliation has been documented since 1992 (Willhite 1999). Since 1999 annual counts of budworm larvae and adult moths indicate a sharp decline for populations in the vicinity of Smith Butte (Beth Willhite, USDA Forest Service, personal communication 2001).

Smith Butte is adjacent to a spotted owl nest site that has been monitored for a decade. Although the nest site previously provided suitable habitat, by 2001 successful owl reproduction at had not been documented for at least 3 yr (Mendez-Treneman 2001). This decline occurred simultaneously with the ongoing budworm defoliation and inspired our research questions.

Objectives

Our objectives were to investigate how spruce budworm defoliation on Smith Butte has altered forest structure and how the changes might influence fire behavior and effects. It is difficult to predict the course of the current budworm outbreak, but for this study we consider 1992 to be early in the outbreak and 2000 to be late. We tested two null hypotheses relative to spotted owl habitat elements: 1) canopy closure has not changed and 2) the basal area of large trees has not changed. We tested five null hypotheses relative to fire behavior and effects: 1) coarse woody debris levels have not changed, 2) potential surface fire behavior has not changed, 3) torching potential has not changed, 4) independent crown fire potential has not changed and 5) potential tree density and basal area losses to wildfire have not changed.

Methods

The 21 sample points installed on Smith Butte in 1992 were remeasured in 2000. We recorded the location, slope, aspect, and plant association for each point. Each sample point consisted of three nested plots and a fuel transect. The largest plot had a variable radius, based on a 40 (English unit) basal area factor (BAF) prism (Bell and Dilworth 1988). Inside this variable radius plot were two fixed plots: one 0.02 ha and the other 0.008 ha. Each plot shared the same center stake. The 100 m fuel transect extended from the center stake in the direction of the next sample point. Supplemental fuels data on fuelbed depth and aboveground biomass were collected in 2001 using the estimation procedures in NEWMDL (Burgan and Rothermel 1984), a program that creates a fuel model suitable for fire behavior prediction from field-collected data. Data from 1992 and 2000 were compared using various statistical tests and software packages. Data from the variable radius plot were used to test hypotheses about changes in canopy closure and conifer basal area between 1992 and 2000, whereas data from the fixed plots and the fuel transect were used to test hypotheses about changes in fire behavior for the same period. Details about the methods and analyses are provided below.

Trees and Snags

In the variable radius plot, data were collected on canopy closure and on all trees and snags larger than 12.5 cm dbh (diameter at breast height or 1.3 m). Canopy closure was estimated using a spherical densiometer (in 1992) and a moosehorn densitometer (in 2000). Densiometers may overestimate actual cover in mature conifer stands by 3-9% compared to densitometers (Ganey and Block 1994). Two canopy closure measurements were made, each 3.6 m from the center (adjusted for slope), north and south of the plot center stake. The average of these two measurements is the point value. Sample trees and snags were selected using standard variable plot sampling methods (Bell and Dilworth 1988). An average of 6-8 trees per point resulted from using the 40 BAF prism. For each sample tree species, status (live or dead), diameter and height were measured or estimated. For each sample snag, the measured attributes included species, diameter, and height.

Shrubs, Herbs, and Tree Regeneration

In the fixed-radius plots, all shrub and herb species were recorded, together with an estimate of cover to the nearest 10% and a tally of conifer regeneration. Estimates of shrub and herb cover were made by using the comparison charts for visual estimation of foliage cover by Terry et al. (1955) as adopted by the USDA Forest Service Pacific Northwest Region 6 Area Ecology Program (Topik 1989).

Fuels

Along the 100 m transect, fuels data were collected based on Brown (1974). Fuels are the dead twigs, branches, and boles of trees and shrubs that have fallen and lie on the ground. Calipers were used to measure all coarse woody debris (> 7.6 cm in diameter) within 2 m of the ground over the entire transect. Smaller fuels and duff depth were sampled on three 2.7 m segments. The first interval began at 40 m from the center stake, the second at 60 m, and the third at 80 m. Data on fine fuels were collected in each sampling segment. The 1-hour fuel load (0-0.62 cm diameter) was tallied over the first 0.91 m of the segment, the 10-hour fuel load (0.63-2.54 cm diameter) was tallied over the first 1.8 m, and the 100-hour fuel load (2.55-7.62 cm diameter) was tallied over the entire 2.7 m.

Surface Fire Behavior

Two components of surface fire behavior (fuelbed depth and aboveground live biomass) were not directly measured in either 1992 or 2000, so we estimated them by using shrub and herb cover data collected in 2000 together with biomass estimates in 2001. We assume biomass changed minimally between 2000 and 2001. For 1992 biomass values, the 2000 cover data and the 2001 biomass estimates were used to develop regression equations for herb and shrub loadings. These equations are:

HerbLoad (t/ac) = 0.08181 Ln (Herb Cover [%]) $r^2 = 0.63$
ShrubLoad (t/ac) = 0.11170 Ln [Shrub Cover [%]) $r^2 = 0.36$

We used these relationships to estimate 1992 herb and shrub loads from the 1992 cover data. Based on our estimates, we assigned the fuelbed depth of the Northern Forest Fire Laboratory (NFFL) Model 10 (Albini 1976) to 1992. For 2000, we used our estimates of fuelbed depth from 2001.

Analysis

Habitat Elements

Changes in canopy closure, the basal area of large conifers, and down wood between 1992 and 2000 were evaluated by subtracting the 1992 estimates of canopy closure, basal area by size class, sound coarse woody debris, rotten coarse woody debris, and total coarse woody debris from the 2000 estimates for each plot.

Canopy Closure

A paired t-test was used to compute the mean difference in canopy closure between 1992 and 2000 ($P = 0.05$ for all analyses). Due to missing values, 13 paired plots were available for analysis (Table 1).

Basal Area

Basal area per tree was computed from measured diameters and the values summed into 12.7 cm diameter classes for all live grand fir and Douglas-fir

TABLE 1. Difference in canopy closure (%) between 1992 and 2000 for points on Smith Butte.

Point number	1992	2000	Difference
1	70	75	5
2	60	33	-27
3	80	32	-48
5	90	26	-64
9	80	30	-50
13	60	42	-18
14	70	27	-43
15	90	60	-30
16	90	16	-74
17	70	57	-13
19	80	47	-33
20	80	62	-18
21	90	50	-40

trees. The difference in means between 1992 and 2000 of diameter classes larger than 63.5 cm dbh was compared using a one sample, two-sided paired t-test for the 16 plots on which large trees existed.

Fuels and Fire Behavior

Changes in fire and woody debris parameters between 1992 and 2000 were evaluated by subtracting the 1992 values for surface fire flame length, critical torching flame length, actual flame length, critical canopy bulk density, actual canopy bulk density, sound coarse woody debris, rotten coarse woody debris, and total coarse woody debris from the 2000 values. One-sample, two-sided t-tests with the null hypothesis that the difference was zero were performed on each parameter.

Surface fire behavior predictions were made through the TSTMDL program of BEHAVE (Burgan and Rothermel 1984). Fuel models for each plot were developed by altering the NFFL Model 10 (Albini 1976) with new fuel loads and fuel depths. Slope was entered as the percent slope normal to the contour at each plot location. Weather was estimated from FireFamily (Bradshaw and McCormick 2000) summaries of historic fire weather at the nearest weather station (Trout Lake) at the 70-percentile and 95-percentile fire weather (Table 2). Wind speeds were adjusted down by a factor of 0.9 in stands with high canopy cover (>70%) with lower adjustment factors in more open stands with lower canopy cover (Rothermel 1983).

TABLE 2. Fire weather for Trout Lake.

Variable	70th Percentile	95th Percentile
1-hr fuel moisture (%)	6.85	4.01
10-hr fuel moisture (%)	10.59	6.00
100-hr fuel moisture (%	12.95	8.71
Herbaceous fuel moisture (%)	90.77	55.01
Woody fuel moisture (%)	120.61	92.95
6.1 m wind speed (km hr^{-1})	14.5	22.5

Torching and independent crown fire behaviors were assessed by the technique of Van Wagner (1977). We enabled torching if the predicted fireline intensity from BEHAVE (95 percentile) exceeded the critical fireline intensity defined by Van Wagner (1977):

$$I_0 = \{0.01BLC(460 + [26FMC])\}^{3/2}$$

Where I_0 = critical surface fireline intensity (kW m^{-1})
BLC = base to the live crown (m)
FMC = foliar moisture content (percent)

Foliar moisture was treated as a constant 90% in these analyses, based on 95-percentile moisture of woody plants derived from local weather records. Base to the live crown was estimated from individual plot tree data in 1992 and 2000 as evaluated by the Fire and Fuels Extension of the Forest Vegetation Simulator (FFE-FVS) (Beukema et al. 2000). Fireline intensity can be scaled to flame length; our results are expressed in terms of surface fire flame length and critical flame length.

Independent crown fire activity was assessed by comparing critical levels of canopy bulk density (CBD_c) to actual levels defined by FFE-FVS (CBD_a) (Beukema et al. 2000). We use canopy bulk density instead of crown bulk density (Scott and Reinhardt 2001), but in previous literature the two terms are interchangeable. Critical levels were defined by comparing actual to critical canopy bulk densities (Van Wagner 1977, Williamson 1999). Independent crown fire behavior depends on maintaining a minimum level of heat flux into the unburned fuel ahead of the fire. The heat flux is a function of fire rate of spread, the bulk density of the crown, and heat of ignition:

$$E = R(CBD)h$$

Where E = net horizontal heat flux (kW m^{-1})
R = rate of spread (m sec^{-1})
CBD = canopy bulk density
h = heat of ignition (kJ kg^{-1})

The equation can be expressed in terms of mass flow rate:

$$S = R(CBD) = E/h$$

Where S = mass flow rate (kg m^2 sec^{-1})

Van Wagner (1977) empirically defined a minimum mass flow rate (S) of 0.05 kg m^2 sec^{-1} below which independent crown fire spread was not possible. Therefore, a critical canopy bulk density (CBD_c), defined as the canopy bulk density below which independent crown fire spread is unlikely, is:

$$CBD_c = 0.05/R$$

Rates of forward spread (R) were estimated for each plot based on Rothermel (1991). The 95-percentile fire weather was used to estimate surface fire rate of spread on each plot using NFFL Fuel Model 10 (Albini 1976) and a 0.4 adjustment to open windspeed, with that value multiplied by 3.34 to obtain R. The calculated CBD_c then was compared to the CBD_a from the FFE-FVS program to determine whether crown fire should be enabled for that plot.

Potential Fire Effects

The tree list data from 1992 and 2000, along with the flame lengths from BEHAVE using 95-percentile fire weather, were used as inputs to the First Order Fire Effects Model (FOFEM) (Reinhardt et al. 1997). Absolute and percent change in basal area due to a simulated wildfire in the 1992 forest and the forest in 2000 were calculated from the percentage mortality of each tree list species and size combination on each plot. If torching or independent crown fire were predicted for a plot, the post fire density and basal area was set to zero. Basal area and density proportional loss were analyzed through a paired ttest using an arcsin-square root transformation of the proportion.

Results

Canopy Closure

We measured a significant decrease in canopy closure between 1992 and 2000 (P<0.0001). On average, canopy closure decreased from 78% in 1992 to 43% in 2000 (Table 1). The 95% confidence interval calculated around the mean difference indicates that the change varies from -48 to -22%. This decrease is much greater than possible error due to different sampling methods in the two periods. It likely stems from a combination of defoliation on large trees and mortality of smaller ones. Tree density decline was primarily in the smaller size classes (Figure 2).

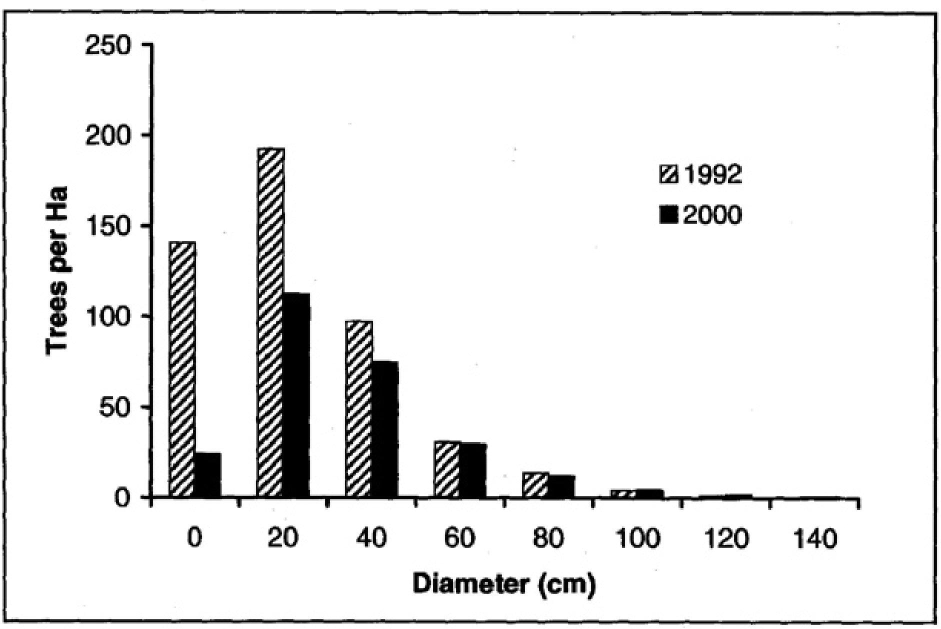

Figure 2. Tree density early (1992) and late (2000) in the western spruce budworm defoliation event.

Basal Area of Large Trees

No significant difference in the basal area of large Douglas-fir or grand fir trees was found between 1992 and 2000. The mean basal area of large trees (>63.5 cm dbh) was 4.64 m^2 ha^{-1} in 1992 com pared to 5.05 m^2 ha^{-1} in 2000.

Fuels and Fire Behavior

Downed Wood

The mean downed coarse woody debris load in 1992 was about 40 Mg ha^{-1}, about 60% of which was sound (Table 3). By 2000, the load was significantly greater in all categories and fuel bed depth indicates this higher loading was tightly packed. Sound coarse woody debris increased by an average of 14.2 Mg ha^{-1} ($P = 0.02$), rotten material increased by 24.3 Mg ha^{-1} ($P = 0.001$), and the total coarse woody debris increased by the sum of these differences (38.5 Mg ha^{-1}; $P = 0.001$). In eight years, the coarse woody debris load nearly doubled.

TABLE 3. Coarse woody debris (Mg ha^{-1}) at Smith Butte in 1992 and 2000.

Category	1992	2000	Difference
Sound	23.15	37.35	14.21
Rotten	17.67	41.97	24.30
Total	40.82	79.32	38.50

Surface Fire Behavior

Predicted surface fire flame lengths significantly increased (Table 4, Figure 3) from 1.4 m in 1992 to 1.9 m in 2000 ($P = 0.0003$). This was largely a function of increasing fuel load, because topography and weather inputs remained constant. Midflame windspeed increased somewhat because of a more open canopy.

Torching and Crown Fire Behavior

In 1992, four of the 21 plots had torching enabled. Most of the plots did not, however, because of the large difference between the critical flame length needed for torching and the lower actual flame length predictions (Table 4). In 2000, the number of plots where torching was, enabled declined to two. None of the four plots in which torching was

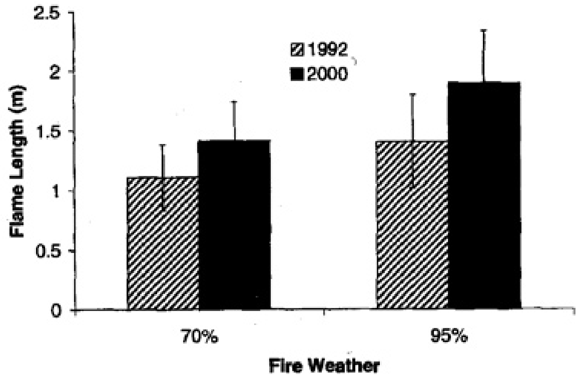

Figure 3. Average surface fire flame lengths across 21 Smith Butte plots in 1992 and 2000. Error bars indicate one standard deviation.

TABLE 4. Fire behavior and crown character estimates for 1992 and 2000.

Variable	1992	2000	Difference
Flame length (m)	1.4	1.9	0.5
Critical flame length (m)	3.0	3.3	0.3
Height to live crown (m)	6.9	8.9	2.0
Actual canopy bulk density (kg m^{-3})	0.11	0.12	0.01
Critical canopy bulk density (kg m^{-3})	0.19	0.19	0

enabled in 1992 had torching enabled in 2000. This variable response is supported by statistical analyses, which showed that height to live crown and critical torching flame lengths did not differ significantly between 1992 and 2000. Critical flame lengths decreased on 7 plots, increased on 10 plots, and showed no change on 1 plot.

Crown characteristics that contribute to independent crown fire potential did not change significantly during this time period (Table 4). Actual canopy bulk densities as measured by FFE-FVS did not change significantly between 1992 and 2000, and critical canopy bulk densities remained stable.

Potential Fire Effects on Forest Structure

Early in the budworm outbreak, a wildfire burning in 95-percentile fire weather would have removed about 22% of the density and 21 % of the basal area (Figure 4). Moderate flame lengths (Table 4) and stand dominance by large trees contribute to

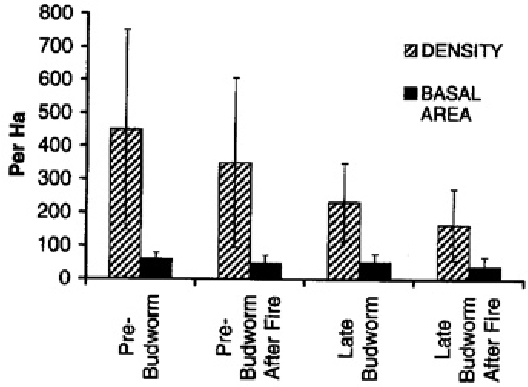

Figure 4. Basal area ($m^2 ha^{-1}$) and density (trees ha^{-1}) in the early budworm condition (1992), the early budworm condition if burned by a 95-percentile weather fire, the late budworm condition (2000), and the late budworm condition if burned by a 95-percentile weather fire. Error bars indicate one standard deviation.

this low predicted loss. Between 1992 and 2000, tree mortality was mostly in the smaller diameter classes (Figure 2) and the residual stand thus maintained a large average diameter. Greater surface fire behavior did not have significant impacts on the proportional loss of density and basal area, which in 2000 averaged 29% and 25% respectively. Absolute density and basal area were larger in 1992 than in 2000, but proportional loss did not differ significantly for either density or basal area between 1992 and 2000. Our FOFEM predictions indicate that a wildfire in conditions similar to those measured in 2000 would leave residual stands on Smith Butte stands with 148 trees ha^{-1} and more than 34 m^2 ha^{-1} of basal area.

Discussion

Forest structure, including canopy closure and levels of downed wood, has changed significantly on Smith Butte following 8 yr of budworm defoliation. Changes in forest fire behavior resulting from human or natural disturbance can be expressed as changes in surface fire, torching potential, and independent crown fire potential. Our assessment of fire behavior changes on Smith Butte includes all three, with mixed results. Coarse woody debris levels are significantly higher, and this increase in fuels contributes directly to a predicted increase in surface fire flame lengths. In contrast, the effects on torching potential and crown fire behavior were moderated by the pattern of tree mortality.

We expected more extreme flame lengths, given both the large surface fuel loads in 2000 and the significant increase since 1992. Other studies have predicted much greater losses in defoliated stands due to the buildup of fuels from trees killed by budworm (Mutch et al. 1993). Two factors may have moderated our predicted flame lengths. Fuelbed depth is the first moderating factor. During our 2001 site visit, we rarely saw places with fuelbeds deeper than 0.60 m. This means that the high amount of woody debris was tightly packed, and that these compressed fuels created an unfavorable packing ratio. If, in our analysis, we had made the fuels less compressed and thus deepened the fuelbed, fire behavior would have increased, particularly for the 2000 measurements. The second moderating factor was the relatively mild 22.5 km hr^{-1} windspeed at 95-percentile weather, and the reduction factor of 0.9 used for the closed canopy stands. If we had used higher windspeeds typical of weather stations east of our study site, or had used a less conservative windspeed reduction factor of 0.7 or 0.8, fire predictions would have resulted in more intense surface wildfire behavior.

The amount and pattern of tree mortality affected both fire behavior and stand structure, and therefore had complex effects on predicted fire severity. The proportional loss in tree density and basal area after a fire was predicted to be the same for 1992 and 2000, even though there was almost a 50% decrease in tree density late in the budworm event (Figure 4). A substantial density of snags will add to the down coarse woody debris totals over the next decade regardless of whether a fire occurs, and an additional 30% reduction in tree density is predicted if a wildfire burns the area. While coarse woody debris is not a factor in surface fire potential models, intense fires may receive an important contribution of energy from large fuel (Rothermel 1991). This additional fuel could play a role in extending the duration and increasing the severity of fires above what our results predict. In contrast, the response of shrub and herb loads should continue to increase in the more open forest, which can dampen the effect on surface fire behavior (Agee et al. 2000). While it is not clear how fire behavior will change, it is clear that ongoing tree mortality will continue to affect potential fire severity on Smith Butte.

Although the residual live tree graph (Figure 2) appears similar to what might result from either a mechanical low thinning (Smith 1962) or fire, the effect of the small tree mortality associated with budworm defoliation is different. Mechanical thinning typically removes many stems at once, whereas the mortality on Smith Butte occurred over several years. Instead of being removed, the dead trees are either falling to the ground or remaining as snags. The net effect of a fire would also be less residual coarse woody debris than that associated with budworm defoliation, because fuel existing prior to the budworm outbreak would be consumed, even as new mortality occurred. In both instances, more woody debris results from the small tree mortality associated with budworm defoliation. Our simulations suggest that current debris loads will not result in stand replacement fires, but more surface woody debris will be accruing over time as current snags fall and perhaps more are created.

The structural elements that moderate potential fire severity on Smith Butte, namely large trees that are fire-resistant because of thick bark and tall crowns, are also elements considered important for spotted owl habitat. Immediately following a disturbance, structural elements like canopy closure or basal area per hectare might not be sufficient for spotted owl habitat, but such effects might be short term (decades) rather than long term (centuries). Future effects on owl habitat would be very different if the large trees on Smith Butte did not survive either a wildfire or subsequent insect or pathogen attacks. Increasing levels of large woody debris may benefit owls by improving conditions for key prey species, but the fire risks and hazards associated with such levels are not well understood.

A remeasurement of the Smith Butte plots is planned to assess how continued mortality and fuel inputs affect fire behavior. Since several root pathogens are also present at the site, we cannot be completely certain how much mortality is attributable to budworm, but the remeasurement should help characterize the lag effects of budworm defoliation.

It is important to remember that the conditions on Smith Butte are uncommon in the area. Indeed, forests with large diameter, early-seral, fire resistant tree species are not common in eastern Washington, owing to harvest history (Covington et al. 1994). Our projections suggest that within such relict patches, fire behavior may not result in stand-replacement fire severity. If this is the case, the conditions on Smith Butte may offer some structural guidelines for managing fire severity in the eastern Washington physiographic province.

Although we have empirical data for changes between measurement periods, we relied on models for our fire behavior assessments. While BEHAVE appears relatively robust due to an extended history of use, the FFE extension to FVS is new and has several key assumptions related to important fire behavior variables. Height to live crown is a critical variable in the prediction of torching potential. It is defined in FFE as that height in the crown where canopy bulk density reaches 0.011 kg m^{-3} in a 3 m horizontal slice through the crown (Scott and Reinhardt 2001). There are little data to validate this assumption. Canopy bulk density is expressed as the highest 5 m running mean of layers 0.3 m thick through the crown, yet another possible way to express it would be as the average through the crown from the base of live crown to the tree tops. Both of these would be subjective, but the latter would result in a decreased predicted crown fire behavior. Selection of an appropriate metric for height to live crown and canopy bulk density is a difficult challenge in the heterogeneous stand structures commonly found in forests in the eastern Cascade Mountains. The last model we used was FOFEM, which predicts mortality from a given flame length or scorch height based on bark thickness and crown dimensions. It is density independent and does not consider secondary mortality due to bark beetles or other agents, so it underpredicts mortality that may occur from synergistic disturbances. These are the strongest models we currently have, but their outputs should be used cautiously.

Acknowledgments

We appreciate the fieldwork of C. Bluestone and K. Hlavka, and the helpful reviews of: A. Camp, S. Campbell, E. Forsman, M. Raphael, D. Tilton, J. White, and B. Willhite. The USDA Forest Service programs for Forest Health Management, the Northwest Forest Plan, and Creating Opportunities funded this study.

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Received 19 March 2002.
Accepted for publication 16 November 2002.

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