Forest Structure and Fire Susceptibility in Volcanic Landscapes of the Eastern High Cascades, Oregon

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Abstract: *Multidimensional forest health initiatives—those designed to restore and protect forest integrity in the large sense—require, among other things, a better understanding of the relationship between the structure of forests and (1) their susceptibility to wildfires outside the range of natural variability and (2) the level of treatment (if any) required to lower susceptibility into an acceptable range. Within the ponderosa pine zone (Pinus ponderosa Dougl. ex Laws.) on the eastern slopes of Oregon's Cascade Range, an average of 86% of trees >5 cm in diameter at breast height (dbh) were less than 101 years old, the density of young trees correlating negatively with the density of old trees (>150 yrs). Species composition differed significantly between age classes: 72% of trees older than 150 years were ponderosa pine, whereas 83% of trees younger than 101 years were either grand fir (*Abies grandis *[Dougl.] Lindl.) or lodgepole pine (*Pinus contorta *Dougl. ex). Loud Crown bulk density calculated to reflect the multiple canopy layer structure of these stands (CBDm) averaged 0.049 kg/m^3, compared to 0.093 kg/m^3 using the standard approach (*$p < 0.001$*), indicating that the latter significantly overestimates risk of active crown fire in stands with complex canopy structure. Nevertheless, modeling with CBDm predicts that under severe but plausible weather conditions (late summer fuel moisture and 48-kph winds) crown kill would exceed 70% on 5 of 14 plots and 50% on an additional 5 plots. Models also predict that thinning trees of <20 cm dbh, coupled with controlled burning to reduce logging slash, would prevent torching (fire moving from the ground into crowns) on all plots, even under extreme fire conditions (low fuel moisture and 80-kph winds). The amount of thinning required to prevent spread from crown to crown (active crown fire) varied widely among plots and depended on the degree of risk deemed acceptable. The relationship between stand basal area and the critical wind speed required to maintain active crown fire followed a negative power law (*$r^2 = 0.92$*), implying that over certain ranges small changes in basal area result in relatively large changes in susceptibility (assuming that changes in basal area are due to addition or removal of smaller trees). Landsat thematic mapper correlates reasonably well with CBDm (*$r^2 = 0.80$*) and may provide a tool for rapid risk assessment.*

Key Words: fire exclusion, forest fire, forest health, ponderosa pine, stability, thinning, tree invasion

Estructura del Bosque y Susceptibilidad al Fuego en Paisajes Volcánicos de Cascadas Altas del Este, Oregon.

Resumen: *Las iniciativas multidimensionales de salud de bosques (es decir, diseñadas para restaurar y proteger la integridad del bosque en sentido amplio) requieren, entre otras cosas, un mejor entendimiento de la relación entre la estructura de bosques y (1) su susceptibilidad a incendios más allá de su variabilidad natural y (2) el nivel de tratamiento (si fuera necesario) requerido para reducir la susceptibilidad a niveles aceptables. En la zona de pino ponderosa (*Pinus ponderosa *Dougl. ex Laws.) en las laderas orientales de la Sierra Cascade, Oregon, un promedio de 86% de los árboles de > 5 cm de diámetro a la altura del pecho (dap) tenían menos de 101 años de edad, la densidad de árboles jóvenes se correlacionó negativamente con la densidad de árboles viejos*

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(de > 150 años). La composición de especies difirió significativamente entre clases de edad: 72% de los árboles mayores de 150 años eran pinos ponderosa, mientras que 83% de los árboles menores de 101 años eran Abies grandis *[Dougl.] Lindl o* Pinus contorta *Dougl. ex Loud. La densidad de la masa del dosel (DMDm), calculada para reflejar la estructura del dosel de capas múltiples de esos bosques tuvo una media de 0.049 kg/m^3. Mediante el método estándar (*$p < 0.001$*), la densidad de la masa del dosel promedió 0.093 kg/m^3, lo que indica que la última sobreestima significativamente el riesgo de fuego activo en la corona de bosques con dosel con estructura de capas múltiples. Sin embargo, el modelo que utiliza DMDm predice que bajo condiciones climatológicas severas, pero plausibles, (humedad de combustible al final del verano y vientos de 48 kph) la mortandad de coronas excedería 70% en 5 de 14 parcelas y 50% en 5 parcelas adicionales. Los modelos también predicen que el corte de árboles de <20 cm dap, combinado con fuego controlado para reducir residuos de la tala, prevendría el fuego antorcha (fuego que se mueve del suelo hacia las coronas) en todas las parcelas, aun bajo condiciones extremas de fuego (baja humedad en el combustible y vientos de 80 kph). La tala requerida para prevenir la propagación de corona a corona (fuego de corona activo) varió ampliamente entre parcelas y dependió del grado de riesgo considerado aceptable. La relación entre el área basal y la velocidad de viento crítica requerida para mantener el fuego de corona activo siguió una ley exponencial negativa (*$r^2 = 0.92$*), lo que implica que, en algunas escalas, cambios pequeños en el área basal resultan en cambios relativamente grandes en la susceptibilidad (suponiendo que los cambios en el área basal se deben a la adición o remoción de árboles más pequeños). El mapa temático de Landsat correlaciona razonablemente bien con DMDm (*$r^2 = 0.80$*) y puede proporcionar una herramienta para la evaluación rápida de riesgo.*

Palabras Clave: estabilidad, exclusión de fuego, invasión de árboles, incendio forestal, modelaje de incendios, pino ponderosa, salud del bosque, tala

Introduction

It is now widely accepted that many dry forest types in western North America are overstocked and vulnerable to catastrophic crown fires (Mutch et al. 1993; Arno et al. 1995; Agee 1997; Covington 2000; Allen et al. 2002). A number of questions remain, however, especially concerning spatial variation (patchiness, variation over environmental gradients) and the number and size of trees that should be cut to lower risk without compromising other environmental values (Allen et al. 2002). Further, few studies have actually documented age-class distributions in dry forests. Finally, the ability to accurately model the relation between fire risk and stand structure has been limited because the standard approach to calculating crown bulk density (CBD), a major determinant of crown-fire risk, was not designed for the multiple canopy layers that characterize dry forests. Addressing these questions is high priority for at least two reasons: (1) to provide guidance in prioritizing and developing landscape strategies for reducing fire risk and (2) to better evaluate trade-offs between reducing fire risk and protecting environmental values that could be jeopardized by logging too much or in the wrong places. In short, to make wise choices we need better assessments of the situation on the ground.

We report the size, species, and age-class distribution of trees in mature and old-growth forests on the eastern slopes of the Cascade Range in Oregon and model current susceptibility to crown fires with a new technique for assessing CBD in stands with multiple canopy layers. We then simulate how various levels of thinning from below—beginning with smaller trees and working up—alter the risk of crown fires and explore the use of remote imagery in assessing fire risk.

Study Area and Methods

Our study was conducted within the Mount Bachelor Volcanic Chain, Deschutes National Forest, Oregon. The area is characterized by late Pleistocene–early Holocene basaltic lava flows, volcanic ash, and pumice deposits; soils are in the early stages of development. Slopes are gentle, on our plots varying from flat to 13%, with an average of 4%. Elevation within the area encompassed by our study ranges from 1300 to 1950 m, with all but two plots falling below 1600 m. The majority of the study area consists of old-growth forests, the overstory consisting primarily of ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) but with patches of other species, and the understory consisting of ponderosa pine, grand fir (*Abies grandis* [Dougl.] Lindl), or lodgepole pine (*Pinus contorta* Dougl. ex Loud), depending on locale. Mountain hemlock (*Tsuga mertensiana* [Bong.] Carr.) forests replace ponderosa pine at the higher elevations (represented by our two highest plots). Being on relatively flat ground, forests at low and mid-elevations are relatively easily accessible and have a long history of selective logging of individual large trees.

Following Franklin and Dyrness (1973), we refer to the zone where ponderosa pine is a major overstory species, either actually or potentially, as the ponderosa pine zone, regardless of whether ponderosa pine is a seral or climax species. In our study area, that includes all plots between 1350 m and 1600 m in elevation, including some with significant numbers of older Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco.) or western white pine (*Pinus monticola* Dougl.). In terms of potentially climax species, our mid-elevation sites fall into a mixed-conifer zone with grand fir or Douglas-fir as major climax species (Franklin & Dyrness 1973; Agee 1994). Historic fire-return intervals ranged from <10 to approximately 25 years in lower elevations, depending on locale, and reached >100 years in the mountain hemlock zone (Bork 1985; Agee 1994).

We selected eight sites representing a range of elevations, soil types, and lava types. At each site we installed two plots separated from one another by 100 m, each having a radius of 25 m (0.2 ha). On each plot except the two at highest elevation (in the mountain hemlock zone), we recorded species and measured the height of all trees taller than 50 cm and diameter at breast height (dbh) of all trees at least 1.3 m tall. Heights of all but the shortest trees were measured with a clinometer. In the mountain hemlock zone we recorded only species and dbh. A total of 3153 living trees >50 cm in height (2750 >5 cm dbh) were recorded on the 16 plots. In selecting plots we attempted to avoid areas with evidence of past selective logging, although that was not possible in some cases. Of the 16 plots, 5 had evidence of past logging (cut stumps): 2 had only one large tree removed, 1 had two trees removed, and both plots at one site had experienced substantial past logging (14 and 21 stumps). We recorded the diameter and species (from bark characteristics) of all stumps.

We established four circular subplots of 2-m radius in each plot for sampling trees <50 cm tall. Subplots were located in each of the four cardinal directions of the plot, with centers located half the distance from the center to the edge of the main plot. Crown lengths were measured with a clinometer on 1196 trees representing all species except mountain hemlock. These were used to develop species-specific equations relating crown length to dbh and height, from which crown lengths of unmeasured trees were calculated.

We took increment cores from 1777 trees distributed among 13 plots, most >5 cm dbh. We took two cores at right angles from each tree and measured sapwood area and bark thickness in the field. Cores that did not hit tree center were discarded. Cores were sealed in plastic straws and returned to Oregon State University for microscopic age determination. Ages of uncored trees were modeled with species-specific equations of tree size versus age developed from the aged trees.

We used equations from Gholz et al. (1979) to estimate foliar biomass per tree. All equations were species-specific except for grand fir, for which we used a pooled *Abies* equation, and western white pine, for which we used the sugar pine *(Pinus lambertiana* Dougl.*)* equation. We used a generic pine equation for ponderosa pine smaller than 15 cm dbh. Leaf area was determined from sapwood area with species-specific equations (Waring et al. 1982).

Calculating Crown Bulk Density

Crown bulk density (CBD) is defined as the dry weight of available canopy fuels per unit canopy volume, where foliage is generally considered the primary available fuel during the spread of a crown fire (Van Wagner 1977). Crown bulk density is commonly calculated as foliage weight divided by canopy depth, where average live crown length is frequently used to estimate canopy depth (Alexander 1988; Graham et al. 1999). The accuracy of this approach depends on how evenly foliage is distributed within vertical canopy space. Using average live crown length as a measure of vertical canopy is especially problematic in stands such as ours with multiple canopy layers.

We calculated CBD in 1-m vertical layers from the bottom to the top of the canopy on each plot, only for trees >5 cm dbh. The approach was to use data on crown location in vertical space (calculated as the difference between tree height and height to base crown) coupled with a constant of proportionality (discussed below) to apportion total plot foliar biomass (calculated from allometric equations) among the different 1-m layers. For a given layer (L_i, spanning the vertical distance i to $i + 1$ m), we tested each tree on a plot to determine whether its crown occupied the entire 1-m interval. The next step was to translate the number of crowns present in each layer to foliar biomass in that layer. To do that we made two assumptions: (1) the foliar biomass contained in L_i is proportional to the total stem basal area at height i (designating this sum as BA_i), and (2) basal area of a given tree at height i = (basal area at breast height) × (1 − i/total height), so that the basal area of a given tree declines linearly with height. The proportionality constant used to convert BA_i to crown biomass in L_i was

$$(\text{total plot foliar biomass})/\,\Sigma i\ \mathrm{BA}_i .$$

This ensured that the sum of foliar biomass over all L_i equaled the total plot foliar biomass.

The calculations described above may be expressed mathematically as

$$F_i = \mathrm{FT} * \Sigma_t\{\mathrm{BABH}_{ti} \times (1 - i/\mathrm{H}_t)\}/\Sigma_i \Sigma_t\{\mathrm{BABH}_{ti} \times (1 - i/H_t)\},$$

where F_i is the foliar biomass in the ith layer, FT is the total plot foliar biomass, BABH_{ti} is the basal area at breast height of tree t, where only trees with crowns in L_i are included, and H_t is the height of tree t.

The contribution of trees <5 cm dbh was estimated with the same allometric equations and by assuming that their crown length was equal to their height. Because for some species the allometric equations were not derived for trees of that size, this is only an approximation. The contribution of trees of <5 cm dbh to CBD was small, however, and therefore error in the calculations for that size class would not significantly affect the modeling outcomes.

The CBD in each layer was calculated as foliar biomass in that layer divided by total plot area, and CBD for the plot was taken as the mean of CBDs in the five contiguous 1-m layers with the highest CBD values. For comparison, we also calculated CBD using the standard approach: foliar biomass/square meter divided by average crown length. Unless otherwise noted, our use of the term crown bulk density (or CBDm; *m* for multiple) refers to that calculated in 1-m sections. Calculations using average crown length are denoted as CBDa.

Calculating Canopy Base Height and Live-Conifer Ground Fuels

Canopy base height is "the lowest height above the ground at which there is a sufficient amount of canopy fuel to propagate fire vertically into the canopy" (Scott & Reinhardt 2001). As with crown bulk density, this variable presents some difficulties in stands with multiple canopy layers. We defined canopy base height as the lowest height above which at least 0.011 kg/m^3 of available canopy fuels was present (Beukema et al. 1997), calculating to the nearest 0.1 m by linear interpolation between the 1 m CBD values.

We used allometric equations to calculate the foliar biomass of all trees of ≤5 cm dbh and >50 cm tall, substituting basal diameter for trees <1.4 m in height. These were put into models as live ground fuels (see below).

Simulated Thinnings

The effects of thinning on crown-fire risk were calculated by deleting from the data all trees below a given diameter, then calculating CBD and canopy base height of remaining trees. Crown biomass of logged trees was calculated to provide an estimate of fuel loading resulting from untreated logging slash. We simulated thinning of all trees of ≤20, 25, 30, or 35 cm dbh. For each simulation, CBD and canopy base height following logging were calculated as described above.

The dry weights of 100-hour (2.54–7.62 cm), 10-hour (0.62–2.54 cm), and 1-hour (<0.62 cm) time-lag fuels added to the forest floor in each simulated thinning were estimated as follows. We calculated foliar biomass (a component of 1-hour fuels) of thinned trees from allometric equations as described above. Branch biomass was distributed among the three fuel classes by taking advantage of the fractal (self-similar) structure of trees. We assumed that for trees of any size, the ratio of foliage to branch biomass for branches of diameter *x* would be similar to the ratio of foliage to wood for trees with dbh *x*, the latter being calculable from allometric equations (Gholz et al. 1979). This approach gives no more than a crude estimate for 1-hour time-lag woody fuels (branches <0.62 cm diameter) because none of the allometric equations had a data range extending that low. For grand fir, the same is true for 10-hour fuels. Allometric equations for pines and Douglas-fir were valid to 2.5 cm and 1.8 cm, respectively, so they should give acceptable estimates of 100- and 10-hour fuels. Because foliage seems likely to compose the bulk of 1-hour fuels, at least in conifers, and foliage was accurately estimated from allometric equations applied to tree dbh, we believe that inaccuracies in the estimates of branches of <0.62 cm had a relatively minor effect on the total calculated biomass of 1-hour fuels.

Modeling Fire Risk

We used Nexus (Scott & Reinhardt 2001) to model parameters related to crown-fire risk for plots as measured and for each thinning simulation. Nexus is an Excel program that links surface- and crown-fire prediction models. Crown bulk density, canopy base height, fuel loading and moisture, wind, and slope steepness are used to predict various aspects of fire behavior, including the likelihood of spread from ground to crown (torching) and from crown to crown (active crowning). We ran two models for each thinning simulation (including unthinned), one with logging slash added to the forest floor ("no-fuels treatment") and one that simulated a broadcast burn by reducing overall forest-floor fuels (described in more detail below). All simulations used slopes corresponding to our sites, and fuel moisture conditions corresponding to late-summer drought: 100% foliar moisture (Agee et al. 2002) and 3%, 4%, and 6% for 1-hour, 10-hour, and 100-hour fuels, respectively (Rothermel 1991). We ran models with various open winds (defined in the model as 6.1 m above the tallest trees) up to 80 kph, well above what is considered high wind in the study area. Our intent in using high wind speeds and very dry fuels was to simulate extreme conditions. We also report threshold wind speeds required to initiate torching and active crown fire.

To adjust open winds to mid-flame wind speeds, Nexus uses a wind-reduction factor, which is simply the ratio of the midflame wind speed to the open wind speed. For unthinned stands we used the default wind-reduction factor, which is 0.1 (midflame wind speeds that were 10% of open wind speeds). To reflect the effect of thinning on wind speeds within stands, we increased the wind-reduction factor in proportion to the reduction in foliar biomass. For example, if foliar biomass was reduced 30% by thinning, we used a wind-reduction factor of $0.1/(1 - 0.3) = 0.14$. This simple procedure would not be valid for canopy reductions of >90%, but none of our simulations approached that.

We used Fuel Model 10 (Anderson 1982) to estimate forest-floor fuels in unthinned stands (6.6 Mg/ha, 1-hour fuels; 4.4 Mg/ha, 10-hour fuels; and 11 Mg/ha, 100-hour fuels) and Fuel Model 10 plus 1-, 10-, and 100-hour fuels from the crowns of thinned trees (calculated as described above) in the various thinning scenarios without fuel treatment. In the latter cases, we used the Nexus default parameters (e.g., heat content, surface-to-volume ratios) for Fuel Model 10, except in the case of fuel bed depth, which was adjusted upward to account for the addition of logging slash. The latter was accomplished by using the same ratio of fuel depth to fuel weight as was used in NEXUS Fuel Model 10. For simulated underburned plots (thinned and unthinned), we used Fuel Model 8 (Scott & Reinhardt 2001), which corresponds to closed timber litter and has exactly half the fuelbed mass as Fire Model 10 (3.3 Mg/ha, 1-hour fuels; 2.2 Mg/ha, 10-hour fuels; 5.5 Mg/ha, 100-hour fuels) and no live fuels.

Results

Density and Age-Species Structure

Density of trees ≥5cm dbh ranged from 335 to 1415 trees per ha (tph) and averaged 868. When all trees of >50 cm in height were considered, density ranged from 600 to 4900 and averaged 2200. Of the total variance in density, 75% was due to differences among sites and 25% to differences between plots within sites. Two sites in the 1400- to 1500-m elevation zone (separated by 2 km) had exceptionally high densities of lodgepole pine, and accounted for most of the among-site variance.

All plots had a similar multiaged structure, with the overwhelming majority of trees younger than 101 years and several to many age classes of older trees (see below). Figures 1a and 1b show histograms of establishment year (age a tree attained breast height), by species, for the 14

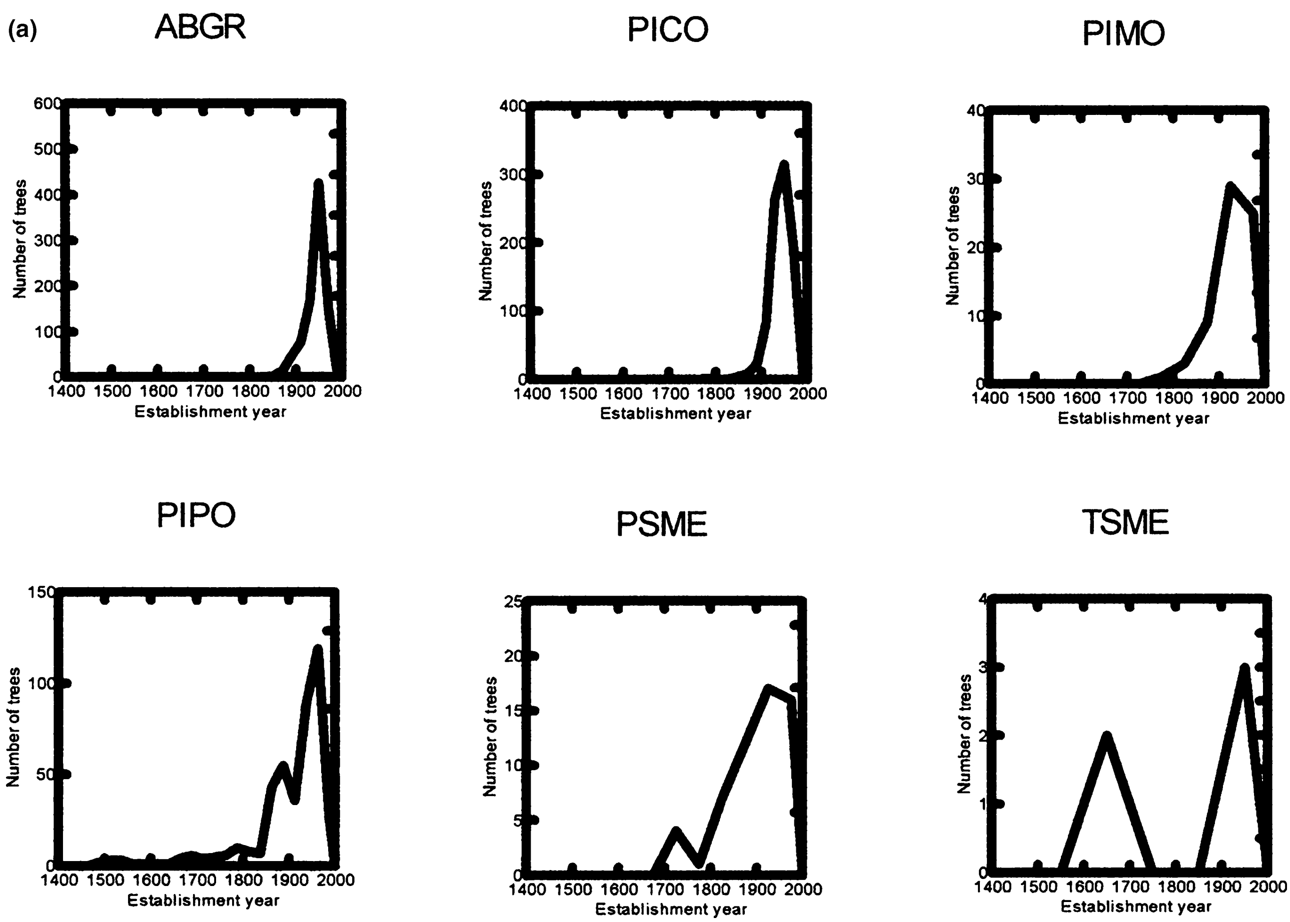

Figure 1. Establishment year (EST YEAR), by species (count), for trees on the east slopes of the Cascade Range, Deschutes National Forest, Oregon: (a) ponderosa pine zone (14 plots) and (b) mountain hemlock zone (2 plots). Abbreviations: PIPO, Pinus ponderosa*; ABGR,* Abies grandis*; PICO,* Pinus contorta*; PIMO,* Pinus monticola*: TSME,* Tsuga mertensiana*; PSME,* Pseudotsuga menziesii*. (continued)*

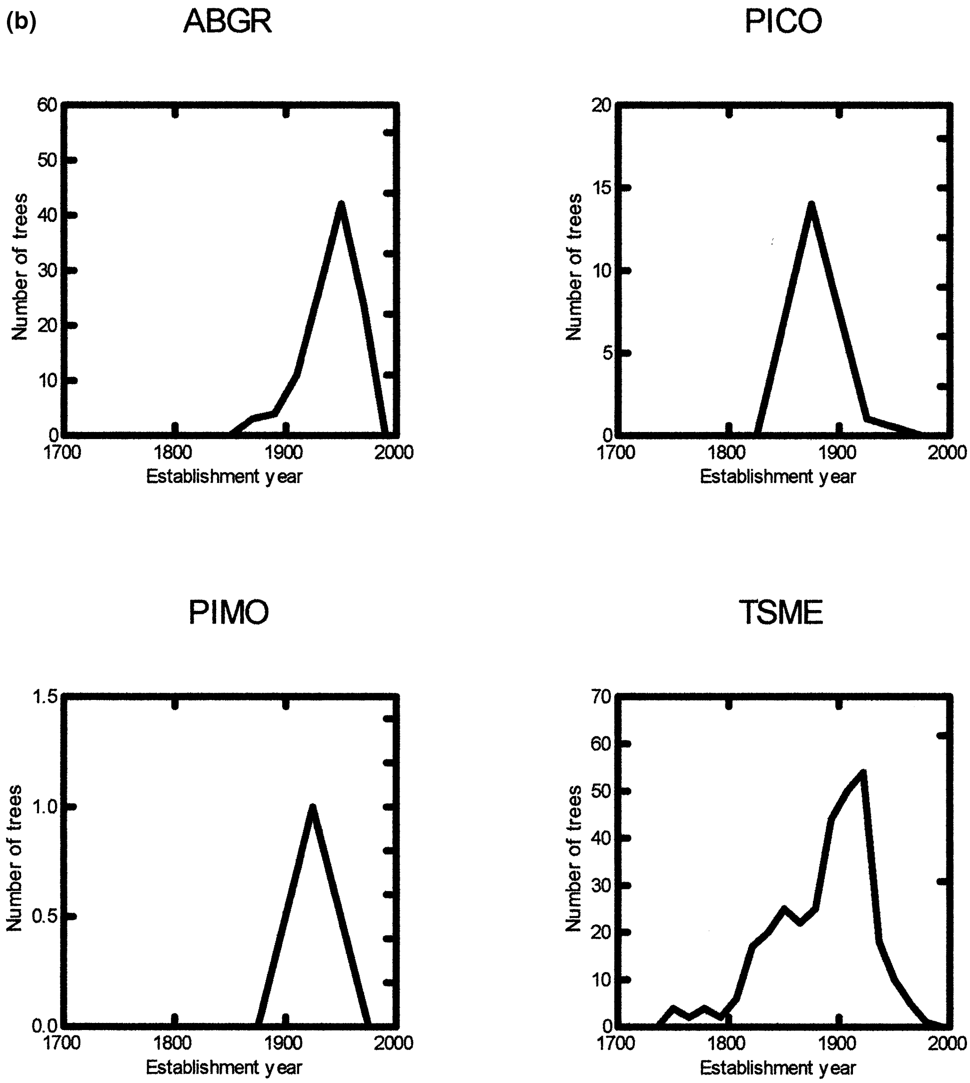

Figure 1. (continued)

plots in the ponderosa pine zone and the 2 plots in the mountain hemlock zone, respectively (y-axis scales of different species differ by up to an order of magnitude). In the ponderosa pine zone, the proportion of total stems $\geq$5 cm dbh that were less than 101 years old ranged from 58% to 98%, with all but 2 plots $\geq$80%. The proportion of younger trees ($\geq$5cm dbh) in the two plots within the mountain hemlock zone were somewhat lower at 53% and 69%. Plots in the mountain hemlock zone had a more even distribution of age classes than those in the ponderosa pine zone, suggesting a more constant and less episodic recruitment of new trees than at lower elevations.

Age-Size Relationship

Among the six species, only Douglas-fir had a reasonably tight relationship between tree age and dbh (data not shown). For other species, and especially for ages of > 100 years, trees of a given age spanned a wide range of sizes. However, trees older than 100 years were rarely smaller than 20 cm dbh for all species except lodgepole pine.

Species Composition of Older Cohorts

Ponderosa pines, which ranged up to 502 years old, were the oldest and most widely distributed of the older trees, occurring on 12 of 14 plots in the ponderosa pine zone. However, there occasionally were also grand fir up to 160 years of age, lodgepole pine up to 222 years, white pine up to 201 years, and Douglas-fir up to 290 years. We recorded three old-growth mountain hemlock (290–320 years old) on lower-elevation plots.

Ponderosa pine was the sole species in the older cohort on five plots and the dominant (over two-thirds of older stems) on another three. Three plots were dominated by older Douglas-fir (60–70% of older stems), and the remaining three plots in the ponderosa pine zone had either a more balanced mixture (two plots) or no older trees at all (one plot). Western white pine greater than 150 years old occurred only on one site with skeletal soils, where it attained a reasonably high density (15 trees on one plot and 5 on the other; 75 and 25 tph, respectively). The two plots in the mountain hemlock zone had by far the greatest density of trees older than 150 years (average 29/plot, 145/ha), all mountain hemlock.

We found both landscape- and plot-level diversity in the age-class distribution of older ponderosa pine. Principle components analysis revealed a tendency for different age classes to segregate across the landscape, trees older than 250 years forming one group (along with ponderosa pine younger than 101 years), trees between 150 and 250 years a second, and trees between 100 and 150 a third. However, this reflected a general tendency and not distinct separations, because older ponderosa pine on most plots spanned a wide range of ages. To estimate the age-class diversity among older trees on a given plot, we grouped trees into 10-year age classes. The 13 plots with ponderosa pine older than 100 years averaged 6 10-year age classes among the older cohorts, with individual plots containing from 1 to 12 age classes. For the most part, older ponderosa pine spanned a wide age range on a given plot, with 10 plots having a range of >100 years and 4 plots a range >200 years between the oldest and the youngest tree at least 100 years old.

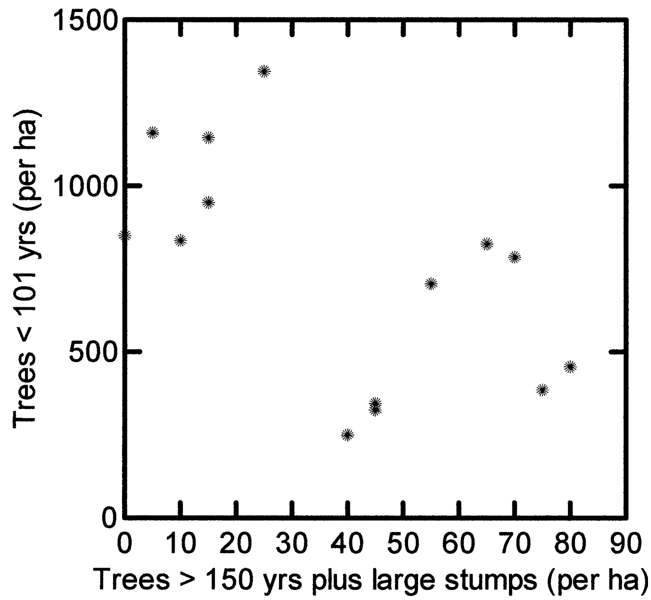

Figure 2. Relationship between the stocking density of trees (tph, trees per hectare) <101 years old and the stocking density of trees >150 years old plus the density of large stumps, east slopes of the Cascade Range, Deschutes National Forest, Oregon.

Species Composition of Younger Cohorts

Except for the lowest- and, to a lesser extent, the highest-elevation sites there was little correspondence between the species composition of older and younger cohorts. On the lowest-elevation site, 78% of trees <101 years old were ponderosa pine (the rest were lodgepole pine). On the highest-elevation site, three out of four trees between 50 and 100 years old were mountain hemlock (the dominant older tree) and the rest were grand fir; however, that proportion was reversed in the <51-year cohort, in which 80% of trees (≥5 cm dbh) were grand fir. Younger age classes in the remaining six sites were heavily dominated by either grand fir or lodgepole pine, which composed 46% and 44%, respectively, of trees <101 years old and ≥5 cm dbh. The proportions were somewhat different for seedlings (<50 cm tall), with roughly equal numbers of ponderosa and lodgepole pines and fewer grand fir (data not shown).

The species composition of the younger age classes (<101 years old) was variable. Ponderosa pine dominated on the lowest-elevation site, lodgepole pine on two sites at 1437 and 1446 m elevation (but separated by 2 km), and white pine (of all ages) on one site at 1376 m (the only site with skeletal soils). Younger age classes on the other four sites were dominated by grand fir (along with mountain hemlock at the highest-elevation site). In the case of grand fir, lodgepole pine, and Douglas-fir, young and old age classes of the same species did not occur on the same plot.

In the ponderosa pine zone, there was a striking bimodal pattern in the density of young trees in relation to that of older trees. Plots with more than four living trees greater than 150 years old (20/ha) averaged 93 trees <101 years old (465/ha), whereas plots with four or fewer living old trees averaged 203 trees <101 years old (1015/ha) ($p < 0.001$). When past stocking of the older cohort was reconstructed by adding in large stumps (>60 cm base diameter), the general pattern was unaltered, but there was a more distinct separation between high and low stocking classes in the older cohort (Fig. 2). (Based on our ring data, Douglas-fir and ponderosa pine stumps 60 cm in diameter have a two-thirds chance of being at least 150 years old; most stumps were significantly larger than 60 cm).

Crown Bulk Density and Height to Crown Base

For the 14 plots in the ponderosa pine zone, CBD calculated based on the densest 5-m vertical layer (CBDm) was 47% lower than when calculated based on average crown length (CBDa) (0.049 vs. 0.093 kg/m^3; $p < 0.001$ by paired t test). The slopes of the lines relating leaf-area index to the two CBD measures were virtually identical, indicating that the difference in the two was consistent across all stocking densities in our study. The fit between leaf-area index and CBDm was much tighter than that between leaf-area index and CBDa ($r^2 = 0.90$ vs. $r^2 = 0.50$). When only larger trees were considered (by "thinning" smaller trees and then calculating CBD), the relationship between the two methods was reversed, CBDm yielding higher estimates than CBDa. For trees >25 cm dbh, CBDm was 9% higher than CBDa, and for trees >35 cm dbh, CBDm was 24% higher than CBDa. Cases in which CBDm is larger than CBDa may be the result of limiting CBDm to the densest 5-m layer.

Crown bulk density correlated reasonably well with stand basal area ($r^2 = 0.74$), and the correlation increased significantly when the density of grand fir was added as a predictive variable along with basal area ($r^2 = 0.87$). For a given basal area, CBDm increased as the density of grand fir increased. Because the different tree species contributed differently, CBDm was not simply related to stocking density. For example, plots where lodgepole pine dominated the younger age classes had by far the highest stocking densities but only moderately dense crowns.

Whereas local variation (differences among plots at a given site) accounted for 25% of the total variance in density, it composed only 10% of the variance in CBDm. For six of the seven sites in the ponderosa pine zone, there was a highly significant positive correlation between elevation and CBD ($r^2 = 0.74, p < 0.001$). The exception was a high-elevation site with low CBDm that was characterized by (1) a relatively high stocking of older Douglas-fir and ponderosa pine (living), (2) the greatest number of large stumps of any site, and (3) a high stocking density of small, young trees (mostly grand fir). This pattern suggests that logging had opened the canopy and allowed an influx of younger trees, which were not yet large enough to produce a high CBD.

Crown base height varied from 0.2 m to 1.6 m, 9 of the 14 plots being above 0.8 m. In general, there was a poor relationship between crown base height and leaf-area index; however, the lowest crown base heights occurred on plots with relatively low leaf-area index, and vice versa. The highest crown base heights occurred on plots with high leaf-area indices.

Fire Susceptibility of Untreated Stands

Based on model predictions, plots vary widely in the degree of crown fraction burned during fire (Fig. 3). However, an active crown fire (approximately 1.0 crown fraction burned) does not occur on any plots until wind speeds reach 40 km/hour at fuel moisture levels corresponding to late summer drought.

Effects of Thinning and Surface-Fuel Reduction

Thinning had dramatically different effects on crown-fire susceptibility depending on whether or not surface fuels were treated. Without fuels treatment, thinning all trees of <20 cm dbh lowered the average fraction of crown burn per plot at all modeled wind speeds of >0 km/

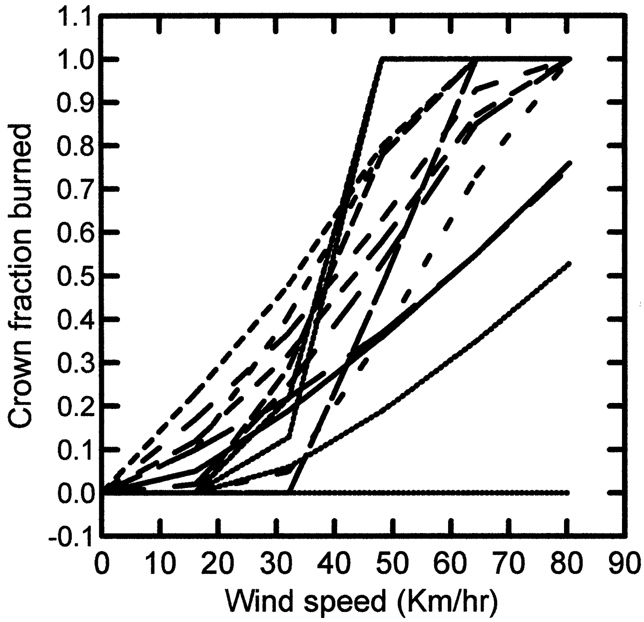

Figure 3. Crown fraction burned versus wind speed for plots as measured (i.e., no simulated thinning or fuels reduction) on east slopes of the Cascade Range, Deschutes National Forest, Oregon. Lines represent individual plots.

hour (all p values < 0.03, paired t test), with proportional reductions in the average fraction of crown burn ranging from 88% at a wind speed of 16 km/hour to 21% at 80 km/hour (Fig. 4). Generally, heavier thinning without fuels treatment produced relatively minor and statistically insignificant incremental reductions in crown fraction burned relative to the 20-cm-level thinning. However, at wind speeds of 48 and 64 km/hour, removing all trees of <35 cm dbh lowered crown-fraction burn by 25–30% compared with the fraction of crown burn for the <20-cm-dbh thinning. In scenarios without surface-fuels treatment, plot responses to different thinning levels varied greatly at both the landscape and the local scale. For the heaviest thinning and highest simulated wind speed, crown-fraction burn ranged from 0 (on three plots) to 100% (active crown fire on six plots). At two sites, both plots had active crown fires, whereas at two other sites one plot had an active crown fire and the other either 0 or <20% crown-fraction burn.

In contrast to the above, simulating a surface-fuels treatment (with Fuel Model 8) without thinning resulted in ground fires (crown-fraction burn = 0) on all but one plot at the highest modeled wind speed (80 km/hour). For the one odd plot, thinning trees of <20 cm dbh coupled with fuels treatment sufficed to keep fire on the ground at the highest wind speed.

It does not necessarily follow, however, that stands without crown fire in our simulations are resistant to crown fires moving in from outside their boundaries. To

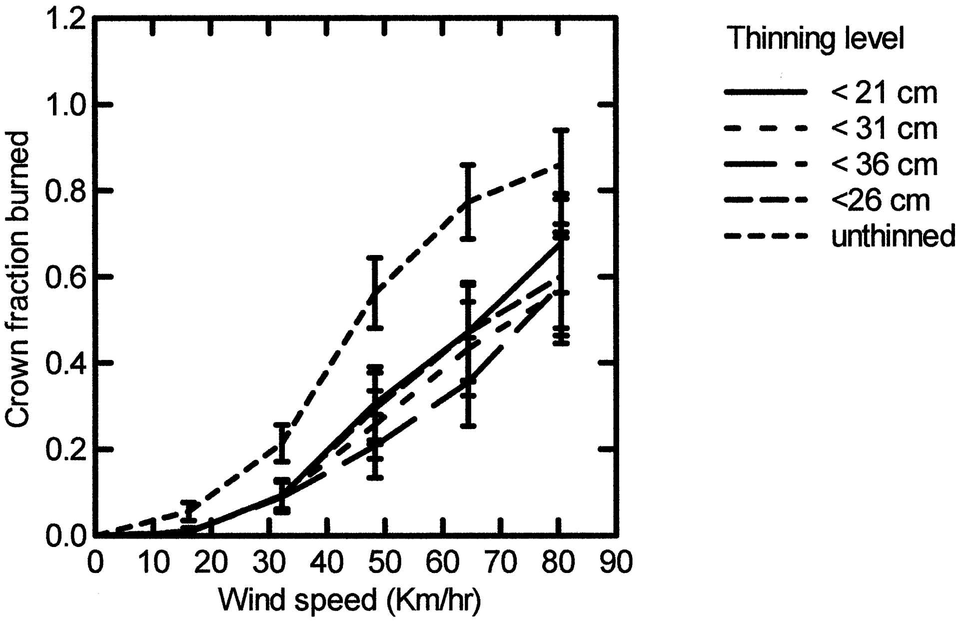

Figure 4. Effect of thinning to different diameter-at-breast-height (dbh) maxima on average crown fraction burned as a function of wind speed; with no simulated fuel treatment. Data and modeling for stands on the east slopes of the Cascade Range, Deschutes National Forest, Oregon. Bars are SE. Top curve is mean of untreated plots, as measured in the field. Successively lower lines represent the following thinning regimes, respectively (dbh >5 cm in all cases): all trees <21 cm dbh, <26 cm dbh, <31 cm dbh, and <36 cm dbh.

explore this point further, it is necessary to examine both components of crown-fire susceptibility: (1) torching, or propagation of fire from the ground into the crown of single trees, and (2) active crown fire, or the spread of fire from crown to crown. Treatment of surface fuels affects only torching, whereas thinning has multiple effects. Thinning reduces the chance of torching by raising crown base height; however, if logging slash is not adequately treated, increased flame lengths increase the chance of torching. By lowering CBD, thinning reduces the chances of crowning at a given wind speed, but at the same time it results in higher wind speed within stands. When our simulations included a surface-fuels treatment, at least some plots at all thinning levels would support "conditional" crown fires, which is to say that crowns were sufficiently dense to support a crown fire, but crown base height was high enough and surface fuels low enough to prevent fire from moving into crowns, even at the highest modeled wind speed. Where stands were in such a "conditional" state, increasing resistance to active crown fire moving across the landscape required further reductions in CBD.

How much thinning is required on our plots to achieve a given level of resistance to active crown fire? With no thinning, less than one-third of the plots would support active crown fire in winds <60 km/hour, and about one-half the plots would support active crown fire in winds of 75 km/hour (Fig. 5). What if the objective is to lower CBD sufficiently to create high resistance on all plots? The relationship between stand basal area and the critical wind speed required to support an active crown fire is illustrated in Fig. 6 with data taken from all our simulations—all plots and all thinning levels, including unthinned. This relationship is valid only for stands thinned from below to achieve a given basal area. Stand basal area is used here because managers can readily measure and manipulate it and it correlates reasonably well with CBD when the latter is calculated to reflect complex canopy structure (CBDm; $r^2 = 0.74$). The relationship between critical wind speed and basal area follows an inverse power law: critical wind speed is proportional to basal area$^{-0.66}$($r^2 = 0.92$). For basal area below about 25 m^2/ha, critical wind speed increases sharply with decreasing basal area. Below 20 m^2/ha, wind speeds >80 km/hour are necessary to support an active crown fire, and at basal areas of <15 m^2/ha, the critical wind speed rises above 90–100 km/hour.

Bringing basal area below 20 m^2/ha would require widely different levels of thinning on our plots (Fig. 7). Three are below that level without thinning, and an additional four could be brought below that level by thinning trees of <25 cm dbh. On three plots, however, it would require thinning all trees of <55–60 cm dbh. Achieving basal areas of $\leq$20 m^2/ha would require cutting some trees older than 100 years on five plots and some trees older than 150 years on one plot. Not surprisingly, plots requiring heavier thinning of large trees had a combination of high basal area and a relatively large proportion of that in big trees, the latter also tending to be older (>100 years). Plots with both high basal area and high stocking density required less thinning in larger size classes, especially if grand fir composed a significant proportion of the stocking.

Growth of Residuals

Would relatively light thinning, meaning thinning of smaller trees, be a temporary measure, with CBD soon returning to prethinning levels? To answer that question,

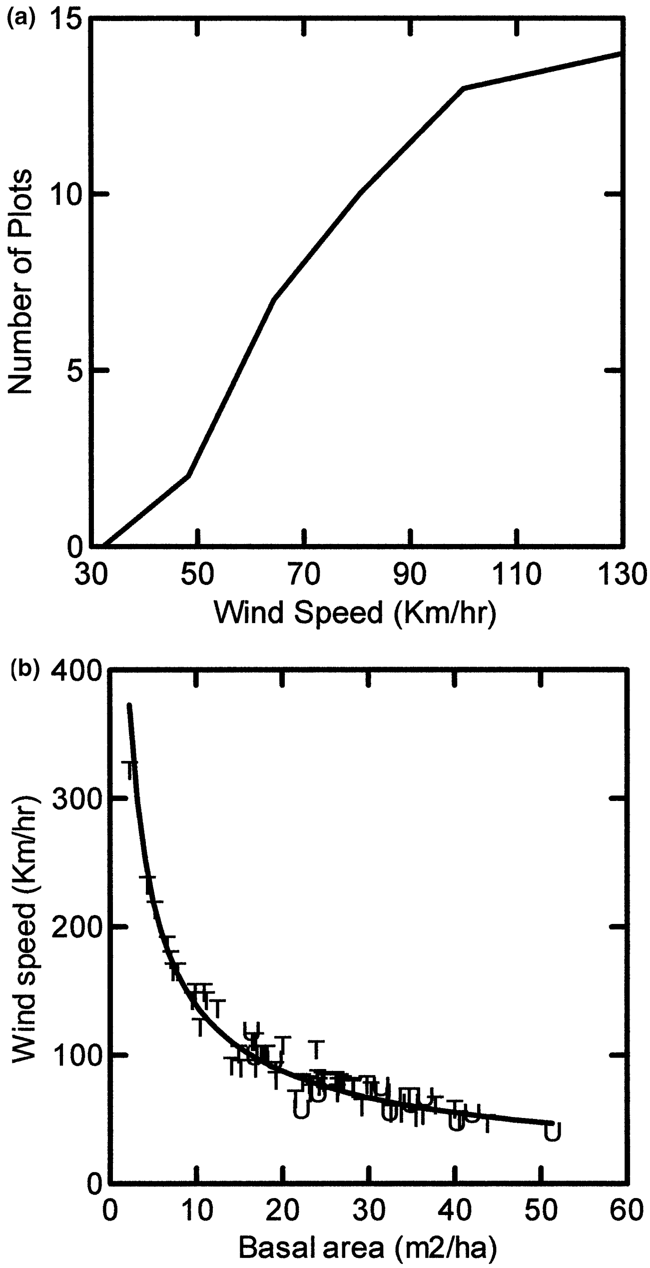

Figure 5. (a) Cumulative distribution of plots with active crown fire as a function of wind speed. Plots as measured (i.e., no simulated thinning or fuels reduction) on east slopes of the Cascade Range, Deschutes National Forest, Oregon. (b) Relationship between stand basal area and critical wind speed for active crown fire (U, untreated; T, thinned) on east slopes of the Cascade Range, Deschutes National Forest, Oregon.

we modeled CBD 20 and 40 years into the future following a thinning that left all trees of $\geq$25 cm dbh. We used the relationship between dbh in 1885 and growth increment between 1885 and 1905, modeled from increment-core ring widths, to estimate dbh growth for 20-year increments.

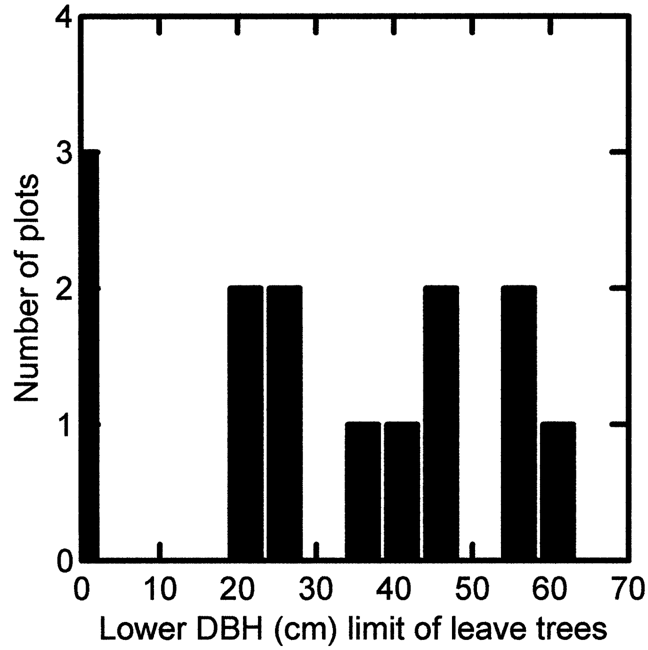

Figure 6. Level of thinning required to attain a basal area of 20 m^2/ha on plots in the ponderosa pine zone on east slopes of the Cascade Range, Deschutes National Forest, Oregon.

We chose the 1885–1905 period to better reflect the more open stand conditions after thinning. We used actual and predicted dbhs to calculate crown biomass and crown lengths for 1995, 2015, and 2035, and then used those variables to calculate average crown length (CBDa) for the same periods. Use of CBDa rather than CBDm is legitimate in this case because, as discussed earlier, the two measures differ relatively little for trees of >25 cm dbh.

After 40 years, the CBD on nearly two-thirds of the plots was still too low to support active crown fire in winds below 80 kph, and more than half of these (five plots) would withstand winds considerably higher than 80 kph (Fig. 7). On two other plots, CBD increased sufficiently by years 20–25 to support active crown fire in 48 km/hour winds, and a third plot reached that threshold by year 40.

Remote Sensing of Canopy Bulk Density

The close correlation between CBDm and leaf-area index raises the possibility that at least one aspect of crown-fire hazard can be measured remotely with Landsat thematic mapper (TM). We explored this on our plots with TM data from 1992 corrected for topography and atmospheric factors (Huang 1998). Among our plots, 80% of the variation in CBDm was explained by the following equation:

$$\text{CBDm} = -0.011 + 0.002 * \text{band4} - 0.004 * \text{band7}$$

Landsat band 4, which spans 0.76–0.90 μm (near infrared), is sensitive to vegetation biomass. Band 7 spans 2.08–2.35 μm (mid-infrared) and is sensitive to moisture

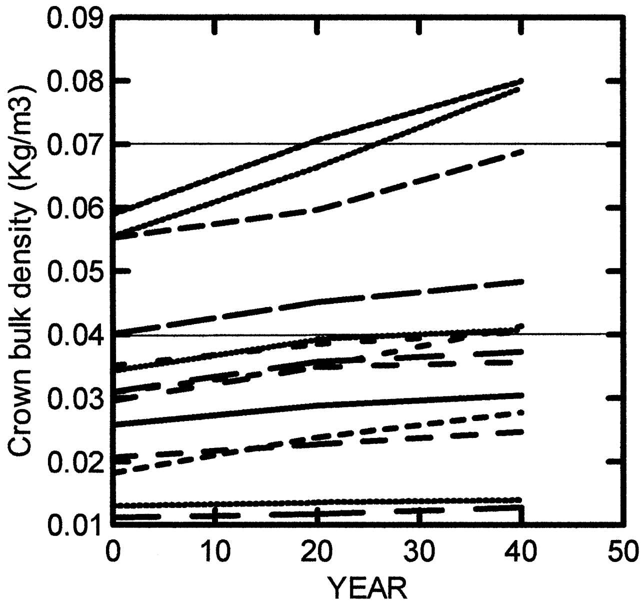

Figure 7. Simulated recovery rate of crown bulk density (CBD) after thinning all trees >5 cm diameter at breast height (dbh) and <25 cm dbh on east slopes of the Cascade Range, Deschutes National Forest, Oregon. Each line represents an individual plot. Horizontal bars represent threshold wind speeds for active crown fire: lower bar = 80 km/hour, upper bar = 48 km/hour. Area below the lower horizontal bar requires winds in excess of 80 km/hour to support active crown fires. Area between the lower and upper horizontal bars will support active crown fires in winds between 48 and 80 km/hour.

variation in plants and soils, with reflectance decreasing as water content increases (http://edc.usgs.gov/guides/landsat_tm.html).

Discussion

Trees <100 years old composed a large majority of total density on all our plots in the ponderosa pine zone, as is commonly believed for dry forests of western North America. On all but the lowest- and highest-elevation sites, there was a dramatic shift in species composition between older and younger age classes. Older trees were primarily ponderosa pine, with pockets of Douglas-fir and western white pine, whereas grand fir and lodgepole pine dominated the younger age classes. Such a shift is expected when fire is excluded from areas where ponderosa pine is seral to other conifers. The high-elevation mountain hemlock forests have a history of less frequent but more severe fires than forests in the ponderosa pine zone (Agee 1994); hence, they should be less affected by the fire exclusion of the twentieth century. Relatively large numbers of grand fir established over the last 50 years on plots in the mountain hemlock zone may or may not indicate the beginnings of a shift in species composition at high elevations. Although our study site in the mountain hemlock zone was approximately 400 m higher in elevation than the commonly recognized upper limits of the grand fir zone (Franklin & Dyrness 1973), we found six grand fir trees between 100 and 150 years old on one of the plots (30/ha), indicating a relatively long presence of that species in at least some high-elevation areas. Nevertheless, it is possible that some combination of fire exclusion and a warming climate has facilitated upward migration of the fir over the last 50 years.

Establishment dates for all species within the ponderosa pine zone (all but the highest-elevation site) show a rise beginning in the mid-1800s and peaking between 1930 and 1950. Establishment of both ponderosa pine and Douglas-fir increased sharply beginning in the mid 1800s, corresponding with the period of EuroAmerican settlement, importation of livestock into the area, and declining populations of Native Americans. Interestingly, the establishment of both those species dipped for a period in the early 1900s, a time when grand fir and lodgepole pine establishment was sharply accelerating, and also the time when the U.S. Department of Agriculture Forest Service (USFS) was beginning to control wildfire. However, patterns of establishment during the nineteenth century must be interpreted cautiously for grand fir and lodgepole pine, which are relatively short-lived species. Our nineteenth-century estimates of presence for those two species should be considered lower bounds only.

Ponderosa pine, Douglas-fir, and western white pine commonly live for several centuries; therefore, the sharp increase in these species in the mid-1800s, especially prominent in ponderosa pine, is likely to be real and to signal an environmental change favoring increased tree establishment coincident with EuroAmerican settlement and predating active fire control by the USFS. Similar patterns have been seen elsewhere in the interior West, the earliest impacts being linked to introduction of cattle and declines in burning by Native Americans (Mutch et al. 1993; Covington et al. 1994; Fule et al. 1997).

Our data show that, even in areas uniformly far from their historic disturbance regime (Fire Regime Condition Class III Hann & Bunnell 2001), there may be a great deal of landscape heterogeneity in the degree of risk and the treatments required to lower risk. Although all plots had a superabundance of young trees, density varied significantly at both the landscape and local scales. At the landscape scale, high densities occurred in the relatively narrow elevational band where lodgepole pine was abundant and low densities on both plots at the site with skeletal soils, which was also the only site with abundant western white pine. Considerable variation also occurred locally (between plots) at sites where ponderosa pine or grand fir dominated the younger cohorts. However, CBD correlated more closely with basal area ($r^2 = 0.74$) and, especially, leaf area ($r^2 = 0.90$) than with stocking density, reflecting the well-known fact that species differ in crown characteristics (Graham et al. 1999). For a given density, grand fir produced higher CBD than lodgepole pine; this is not surprising given the fact the former is significantly more shade-tolerant and maintains longer, denser crowns than the latter.

Lowering the risk of torching and active crown fire may require different levels of treatment. The model predicts that reducing surface fuels by 50% (from Fuel Model 10 to Fuel Model 8), with no thinning, is sufficient to prevent torching (fire moving from ground to crowns) on 13 of 14 plots in the ponderosa pine zone, even in high winds. A light thinning (dbh < 20 cm) coupled with surface-fuel reduction is sufficient to prevent torching on the fourteenth plot. That result may or may not apply specifically to our plots, because height to base crown was a measured variable but surface-fuel loading was taken from a model. It is, however, consistent with what has been seen elsewhere. On the 2002 Hayman fire in Colorado, an extreme crown fire dropped to the ground when it encountered a large area (several thousand hectares) that had been recently underburned to reduce surface fuels and kill small trees (Hayman Fire Case Study Analysis, available at http://www.fs.fed.us/rm/hayman_fire/).

Reducing the risk of active crown fire may necessitate heavier thinning, depending on stand structure and the acceptable degree of risk. Crown bulk density on the majority of our plots is too low to support active crown fire even in winds up to 50–60 kph, but some plots are vulnerable at lower wind speeds. In extreme fire conditions, such as the >100-kph winds that occurred on the 2002 Hayman fire (www.wilderness.org/ourissues/wildfire/hayman.cfm), most or all of our plots would support active crown fire. Our study was not designed to address landscape-level fire risk. However, for a hypothetical landscape with a range of stand structures and crown bulk densities similar to our plots, protecting the entire landscape against such extreme conditions would require levels of thinning ranging from relatively light to relatively heavy, where heavy thinning implies removing all trees up to 60 cm dbh. Because most trees in that size range would be older than 100 years, and some perhaps older than 150 years, trade-offs emerge between lowering risk and cutting trees in a rare age class. Moreover, the negative relationship between the density of young and old trees suggests that cutting large, old trees to reduce current risk could exacerbate future risk by allowing a dense understory to develop. Ultimately, such trade-offs must be evaluated in the context of specific landscape conditions and models that relate condition to spread. Referring again to the hypothetical landscape with a pattern similar to that of our plots, one-third or less of the area in a high-risk situation may be tolerable, depending on where the high-risk areas were located.

Thinning from below must retain some young trees of desired species if stands are to retain a healthy age structure. Accomplishing that without leaving the landscape vulnerable to crown fire is a central challenge and requires attention to spatial detail. In our study, significant numbers of young ponderosa pine (<50 years old) occurred only at the lowest-elevation site, probably because of reduced competition from other tree species. On the other sites in the ponderosa pine zone, restoring a viable ponderosa pine forest requires areas with low enough canopy cover to allow shade-intolerant ponderosa pine to establish and grow. The relatively low CBD of such areas reduces the risk of small trees acting as foci for converting ground fires to active crown fires. This is not an argument for cutting large, old trees or opening stands excessively, which, as we pointed out earlier, could have the unwanted effect of allowing too much regeneration and which our data indicate is not necessary to achieve acceptable levels of ponderosa pine regeneration. At our lower elevation site, abundant, young ponderosa pine (<50 years) occurred in a range of ages and sizes beneath overstories (trees > 25 cm dbh), with basal areas ranging from 18 to 25 m^2/ha. On all but one of our plots in the ponderosa pine zone, the basal area in trees older than 100 years was either within or below that range. The primary limitation to regeneration of ponderosa pine on our sites is likely to be the same thing that creates crown-fire hazard, a dense midstory of grand fir or lodgepole pine.

Our predictions about the relationship between stand structure and risk of active crown fire depend on the validity of our CBD calculations. Both logic and data argue that the technique we used provides a better estimate of CBD in stands with complex canopy structure than the traditional approach of using average crown length. In stands with a wide range of tree sizes, average crown length is relatively meaningless in predicting foliage packing, an assertion supported by the fact that our approach yields CBDs that correlate much more closely with leaf area and stand basal area than CBDs calculated based on average crown length.

The choice to calculate CBD over a 5-m vertical distance was somewhat arbitrary, but fortuitous. Examination of the vertical profiles for each plot showed that, although CBD in a few plots peaked within a vertical span of 2 to 3 m, in most plots the highest CBD occurred within a broad plateau spanning approximately 4–6 m. Consequently, had we chosen a vertical distance shorter than 5 m, the calculated CBD would have increased somewhat but not a great deal, whereas with a wider span they would have dropped significantly. In other words, the CBDs we calculated are at or close to the stand maxima. The vertical distance most relevant to spread of fire through crowns is a question of fire physics and beyond the scope of this paper.

How generalizable are our results? The stand structures typifying our plots—a superabundance of young, small trees—have been reported throughout the dry forests of the West, and invasion by young pines, true firs, and Douglas-fir is a common feature of many low- and mid-elevation western forests (Mutch et al. 1993; Covington et al. 1997; Allen et al. 2002). The soils of our study area are formed from relatively young lavas (a few thousand years old) and hence are more poorly developed than might be found in other areas. However, the large range of leaf-area indices on our plots, 2–10 m^2/m^2, reflect a wide variety of moisture and nutrient conditions, although mortality on the more heavily stocked plots indicates that those may still be coming into equilibrium with site conditions. Nevertheless, it seems reasonable to hypothesize that, for areas with similar stocking and species composition, the predictive relationships we found between stand leaf area (or basal area) and canopy bulk density would apply reasonably well. Because the potential for active crowning varies with slope, however, the relationship between canopy bulk density and active crown fire predicted by our modeling may not apply in areas with steeper slopes than those of our study sites (all under 10%).

Our findings support the argument that wide application of highly specific thinning rules is at best crude and at worst contraindicated in heterogeneous landscapes (Graham et al. 1999). As we discussed earlier, cutting larger trees in a stand is likely to create future problems and should not be done if long-term landscape health is the primary objective (Allen et al. 2002). However, trees of a given dbh may be "large" on one site and only midsized on another. For example, the largest cohort was composed of trees >40 cm dbh on some of our plots, whereas on others the largest trees were no more than 25–35 cm dbh. Thinning guidelines using a relatively small diameter limit (e.g., 20 cm) would significantly reduce the risk of crown fire on some of our plots but not on others, whereas a higher diameter limit would remove more trees from some plots than necessary. Moreover, restoration should strive for landscape heterogeneity to protect habitat and other environmental values (Tiedemann et al. 2000; Allen et al. 2002). These considerations argue for maintaining flexibility as to how general goals translate into what is and is not thinned in any given stand.

To be workable, flexibility requires at least two things: (1) multidisciplinary planning and shared understanding regarding overall objectives and what sorts of stand and landscape structures best meet those objectives and (2) an open, adaptive process subject to validation, feedback and adjustment if necessary. Generalizable relationships (e.g., between elevation and basal area) and rapid-assessment tools (e.g., remote imagery) can facilitate landscape-level planning and treatment prioritization within locales and help ensure that local decisions are consistent with landscape objectives. Once landscape objectives are established, thinning from below to achieve basal areas that meet those objectives would seem a sensible and workable approach.

Throughout, it is important to bear in mind that the integrity of forests and forested landscapes has many dimensions. Any kind of intervention to improve stand health can also have negative impacts, including soil compaction, nutrient loss, damage to streams, spread of invasive weeds, and loss of critical habitat (Tiedemann et al. 2000; Allen et al. 2002). In most cases, careful planning and execution can minimize such impacts or avoid them altogether. The challenge in restoring forest health is to fix one set of problems without exacerbating others or creating new ones (Perry 1995).

Acknowledgments

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Literature Cited

Agee, J. K. 1994. Fire and weather disturbance in terrestrial ecosystems of the eastern Cascades. General technical report PNW-GTR-320. U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Agee, J. K. 1997. The severe weather wildfire–too hot to handle? North West Science **71:**153–156.

Agee, J. K., C. S. Wright, N. Williamson, and M. H. Huff. 2002. Foliar moisture content of Pacific Northwest vegetation and its relation to wildland fire behavior. Forest Ecology and Management **167:**67–66.

Alexander, M. E. 1988. Help with making crown fire hazard assessments. General technical report INT-143. U.S. Department of Agriculture Forest Service, Intermountain Research Station, Ogden, Utah.

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecolological Applications **12:**1418–1433.

Anderson, H. E. 1982. Aids to determining fuel models for estimating fire behavior. General technical repost INT-122. U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah.

Arno, S. F., J. H. Scott, and M. G. Hartwell. 1995. Age-class structure of old growth ponderosa pine/Douglas-fir stands and its relationship to fire history. Research paper INT-RP-481. U.S. Department of Agriculture Forest Service, Intermountain Research Station, Ogden, Utah.

Beukema, S. J., D. C. Greenough, C. E. Robinson, W. A. Kurtz, E. D. Reinhardt, N. L. Crookston, J. K. Brown, C. C. Hardy, and A. R. Stage. 1997. An introduction to the fire and fuels extension to FVS. Pages 191–195 in R. Teck, M. Mouer, J. Adams, editors. Proceedings of the forest vegetation simulator conference. General technical report INT-373. U.S. Department of Agriculture Forest Service, Intermountain Forest Range and Experiment Station, Ogden, Utah.

Bork, J. 1985. Fire history in three vegetation types on the east side of the Oregon Cascades. Ph.D. thesis. Department of Forest Science, Oregon State University, Corvallis.

Covington, W. W. 2000. Helping western forests heal. Nature **408:**135–136.

Covington, W. W., R. L. Everett, R. Steele, L. L. Irwin, T. A. Daer, and A. N. D. Sinclair. 1994. Historical and anticipated changes in forest ecosystems of the inland West of the United States. Journal of Sustainable Forestry **3:**13–63.

Covington, W. W., P. Z. Fule, M. M. Moore, S. C. Hart, T. E. Kolb, J. N. Mast, S. S. Sackett, and M. R. Wagner. 1997. Journal of Forestry **95:**23–29.

Franklin, J. F., and C. T. Dyrness. 1973. Natural vegetation of Oregon and Washington. General technical report PNW-8. U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Fule, P. Z., W. W. Covington, and M. M. Moore. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine. Ecological Applications **7:**895–908.

Gholz, H. L., C. C. Grier, A. G. Campbell, and A. T. Brown. 1979. Equations for estimating biomass and leaf area of plants in the Pacific Northwest. Research paper 41. Forest Research Laboratory, Oregon State University, Corvallis.

Graham, R. T., A. E. Harvey, T. B. Jain, and J. R. Tonn. 1999. The effects of thinning and similar stand treatments on fire behavior in western forests. General technical report PNW-GTR-463. U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Hann, W. J., and D. L. Bunnell. 2001. Fire and land management planning and implementation across multiple scales. International Journal of Wildland Fire **10:**389–403.

Huang, J. 1998. Spatial characterization and analysis of forests in the Mount Bachelor Volcanic Chain, Central Oregon. Ph.D. dissertation. Department of Forest Science, Oregon State University, Corvallis.

Lehmkuhl, J. F., P. F. Hessburg, R. D. Ottmar, M. H. Huff, R. L. Everett, E. Alvarado, and R. Vihnanek. 1995. Assessment of terrestrial ecosystems in eastern Oregon and Washington: the eastside forest ecosystem health assessment. Pages 87–99 in R. L. Everett and D. M. Baumgartner, editors. Ecosystem management in western interior forests. Department of Natural Resources, Washington State University, Pullman.

Mutch, R. W., S. F. Arno, J. K. Brown, C. E. Carlson, R. D. Ottmar, and J. L. Peterson. 1993. Forest health in the Blue Mountains: a management strategy for fire-adapted ecosystems. General technical report PNW-GTR-310. U.S. Department of Agriculture Forest Service, Portland, Oregon.

Perry, D. A. 1995. Landscapes, humans, and other system level considerations: a discourse on ecstasy and laundry. In Proceedings of a workshop on ecosystem management in western interior forests. D. Baumgartner and R. Everett, editors. Washington State University, Cooperative Extension Unit, Pullman.

Rothermel, R. C. 1991. Predicting behavior and size of crown fires in the northern Rocky Mountains. Research paper INT-438. U.S. Department of Agriculture Forest Service, Intermountain Forest Range and Experiment Station, Ogden, Utah.

Scott, J. H. and E. D. Reinhardt. 2001. Assessing crown fire potential by linking models of surface and crown fire behavior. Research paper RMRS-RP-29. U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.

Tiedemann, A. R., J. O. Klemmedson, and E. L. Bull. 2000. Solution of forest health problems with prescribed fire: are forest productivity and wildlife at risk? Forest Ecology and Management **127:**1–18.

Van Wagner, C. E. 1977. Conditions for the start and spread of crown fire. Canadian Journal of Forest Research **7:**23–34.

Waring, R. H., P. E. Schroeder, and R. Oren. 1982. Application of pipe model theory to predict canopy leaf area. Canadian Journal of Forest Research **12:**556–560.