CSIRO PUBLISHING

International Journal of Wildland Fire **2012**, *21*, 1004–1013
http://dx.doi.org/10.1071/WF11121

Fuel treatment effects on tree mortality following wildfire in dry mixed conifer forests, Washington State, USA

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Abstract. Fuel reduction treatments are increasingly used to mitigate future wildfire severity in dry forests, but few opportunities exist to assess their effectiveness. We evaluated the influence of fuel treatment, tree size and species on tree mortality following a large wildfire event in recent thin-only, thin and prescribed burn (thin-Rx) units. Of the trees that died within the first 3 years, most died in the first year regardless of treatment. First-year mortality was much higher in control and thin-only units (65 and 52%) than in thin-Rx units (37%). Cumulative third-year mortality followed a similar trend (78 and 64% in control and thin-only units) *v.* 43% in thin-Rx units. Percentage crown scorch is a strong predictor of mortality and is highly dependent on fuel treatment. Across all treatments, *Pinus ponderosa* had a lower probability of post-fire mortality than did *Pseudotsuga menziesii*. Finally, the probability of beetle attack on surviving trees was highest in large-diameter trees within thin-only treatments and lowest within thin-Rx treatments. This study contributes further evidence supporting the effectiveness of thinning and prescribed burning on mitigating post-fire tree mortality. We also present evidence that a combination of thinning and prescribed burning is associated with lower incidence of post-fire bark beetle attack.

Additional keywords: bark beetles, crown scorch, *Pinus ponderosa*, *Pseudotsuga menziesii.*

Received 22 August 2011, accepted 14 June 2012, published online 1 August 2012

Introduction

Under a warming climate and concern about hazardous fuel accumulations, fuel reduction treatments increasingly are used to mitigate future wildfire severity in dry forests of western North America. Existing studies generally agree that prescribed burning following mechanical thinning is effective at mitigating wildfire severity (Finney *et al.* 2005; Strom and Fulé 2007; Safford *et al.* 2009; Wimberly *et al.* 2009; Prichard *et al.* 2010; Johnson *et al.* 2011). By reducing fuel continuity and potential energy release of canopy and surface fuels, fuel treatments can reduce potential fire behaviour and effects (Agee and Skinner 2005; Peterson *et al.* 2005; Johnson *et al.* 2011). Numerous studies have developed models that evaluate causes of direct and delayed tree mortality following fire (Peterson and Arbaugh 1986; Ryan *et al.* 1988; Stephens and Finney 2002; McHugh and Kolb 2003; Sieg *et al.* 2006; Thies *et al.* 2006; Hood and Bentz 2007). To date, few models include fuel treatment as a predictor of tree mortality following fire (but see Ritchie *et al.* 2007 and Prichard *et al.* 2010), and most models rely on post-fire measurements such as crown scorch. A better understanding of the role of fuel treatments in mitigating wildfires may assist in designing effective treatments, strategically placing treatments and developing models that use pre-fire measurements and treatment records to predict potential mortality following wildfire.

In a review of tree mortality models, Fowler and Sieg (2004) reported that the most common predictors of direct and delayed tree mortality include tree size, crown damage and bole scorch. Direct tree mortality (i.e. in the first year following fire) can be caused by torching, crown scorch, cambial damage from intense surface fires, and root damage from surface and ground fires (Ryan *et al.* 1988; Ryan and Amman 1994). Even trees that survive the direct effects of fire may sustain injuries that render them more susceptible to drought, insects and disease (Fowler and Sieg 2004; Thies *et al.* 2006). Many conifer tree species in dry forests of western North America are considered fire resisters and possess adaptations to wildfire, including thick bark, abscission of lower branches and developed root systems that help them survive the direct effects of a wildfire (Agee 1993; Baker 2009). Although large-diameter trees generally have more defences against direct fire effects than do small-diameter trees, they can be vulnerable to delayed mortality (i.e. mortality that occurs beyond 1 year following fire) due to drought stress, insects and pathogens (McHugh and Kolb 2003; Hood and Bentz 2007; Kolb *et al.* 2007). For management and planning it is important to be able to predict direct and delayed mortality due to fire, and how that may differ with fuel treatments.

Existing models of tree mortality are quite robust with a high correlation between post-fire severity measures such as percentage crown scorch volume, bole char and tree mortality (Regelbrugge and Conard 1993; Ryan and Amman 1994; McHugh and Kolb 2003; Sieg *et al.* 2006; Hood and Bentz 2007; Hood *et al.* 2007). Several studies have compared the relative resistance of tree species to fire (Ryan and Reinhardt 1988; Rigolot 2004; Fernandes *et al.* 2008). However, to date, only one published study (Wyant *et al.* 1986) has compared post-fire mortality for different tree species, and that study found no significant differences in mortality between Douglas-fir (*Pseudotsuga menziesii*) and ponderosa pine (*Pinus ponderosa*) 2 years following a prescribed burn within a single burn unit. There is clearly a gap in understanding how to predict post-fire tree mortality using pre-fire measurements and management actions, as well as in understanding how species may differ in their probability of mortality.

In a companion study (Prichard *et al.* 2010) we evaluated the effectiveness of two fuel treatments (mechanical thinning and mechanical thinning followed by a broadcast burn) on mitigating severity of effects from the 2006 Tripod Complex fires. The fires occurred in dry, mixed conifer forests in north central Washington State. Three years post-fire, tree mortality differed significantly between the two treatments: nearly 60% of trees survived in thin and prescribed burn units *v.* less than 20% in thin-only (thin) and control units. Considering only large-diameter trees (>20-cm diameter at breast height, DBH), close to 75% survived in thin and prescribed burn units *v.* 36 and 29% in thin and control units. Results include a simple model of tree mortality with fuel treatment as a predictor variable but do not include effects of tree diameter, species and year since fire.

In this study, we develop a set of logistic regression models that evaluate both direct and delayed tree mortality following the 2006 Tripod Complex fires. Our objective was to determine the primary drivers of direct and indirect tree mortality following wildfire. We use four modelling approaches: (1) evaluating the effect of crown scorch on tree mortality; (2) testing whether post-fire tree mortality 1, 2 and 3 years following the wildfire is dependent on fuel treatment and tree diameter; (3) comparing predicted mortality of Douglas-fir and ponderosa pine as a function of fuel treatment and diameter and (4) evaluating the effect of treatment, tree diameter and species on probability of bark beetle attack on surviving trees.

Methods

Study area

The 2006 Tripod Complex fires burned over 70 000 ha in the Okanogan–Wenatchee National Forest, Washington State (Fig. 1). The majority of the fire area burned with moderate to high severity in high elevation forests (>1300 m) dominated by lodgepole pine (*Pinus contorta* var. *latifolia*), Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*). The south-western portion of the fire burned at low to mid elevations and involved past fuel treatment units, including commercial thinning and shelterwood harvests, some of which had been prescribed burned before the wildfires. Common tree species at low to mid elevations include ponderosa pine, Douglas-fir, Engelmann spruce and western larch (*Larix occidentalis*).

We sampled tree mortality and severity measures in eight mechanically thinned units, eight mechanically thinned and prescribed burned units and eight unmanaged controls. Thin treatments included mechanical thins from below and shelterwood harvests within 10 years before the wildfire. Thin-Rx treatments were mechanically harvested followed by a broadcast burn within 6 years before the wildfire. Units were whole-tree harvested using tractor logging with the exception of four thin-only units that were helicopter logged (Prichard *et al.* 2010). Although most of the harvested biomass was removed from units, some logging residue remained on site. A detailed description of our sampling design and individual units is included in Prichard *et al.* (2010). A 2006 Burned Area Reflectance Classification image was used to confirm that all units included in our analysis were burned by the wildfire and were not surrounded by unburned forest. Units were located at low to mid elevations and were dominated by Douglas-fir and ponderosa pine.

Fig. 1. Location of the 2006 Tripod Complex fires.

Sampling methods

Tree sampling methods are summarised here and detailed more fully in Prichard *et al.* (2010). A minimum of 10% of each unit was sampled using circular plots along systematic grids. We used a nested plot sampling design to accommodate variable tree densities. Treated units (e.g. thin and thin-Rx) were sampled using 0.2-ha plots. Control units were sampled using 0.08-ha plots to account for generally much higher tree densities. The following measurements were collected for each sampled tree: DBH (cm), crown base height (m), height to live crown (m), tree height (m), maximum height of crown scorch (m), minimum and maximum bole char (m), percentage of the crown volume that was scorched or consumed by fire, status (live *v.* dead) and tree burn severity index (1 = unburned, 2 = scorched foliage, 3 = lightly burned (some foliage and twigs burned), 4 = moderately burned (foliage and small stems consumed) and 5 = severely burned (only charred stems remain)). Trees were recorded as live if they had any green foliage in their crowns. During the summers of 2008 and 2009, plots that had live trees in 2007 were revisited to sample subsequent tree mortality. The 2009 survey included detailed observations of each sampled tree, including damage agents such as fire scars, conks and evidence of bark beetle activity (presence of pitch tubes and frass). The initial objectives of our field sampling did not include an inventory of bark beetle evidence. However, the widespread outbreak of Douglas-fir beetles (*Dendroctonus pseudotsugae*), starting in 2008 and peaking in 2009 following the wildfires, prompted us to include sampling of beetle evidence in our third year of sampling. Only trees that were still living in 2007 were monitored in 2009. Trees with >10 pitch tubes were noted as having beetle evidence. Cambial observations to identify bark beetle species were not made in this study. Red frass observed on Douglas-fir tree boles was evidence of Douglas-fir beetle attacks (Hagle *et al.* 2003), whereas ponderosa pine and lodgepole pine could have been attacked by a variety of bark beetles, including western pine beetle (*D. brevicomis*), mountain pine beetle (*D. ponderosae*), engraver beetles (*Ips* spp.) and red turpentine beetle (*D. valens*). Tree data were summarised by unit and percentage tree mortality was calculated as the percentage of dead *v.* total trees in each unit. A total of 5358 trees were sampled, including 499 lodgepole pine, 1274 ponderosa pine and 3072 Douglas-fir.

Statistical analysis

Exploratory data analysis

We calculated summary statistics of tree mortality, crown scorch and beetle evidence by species and fuel treatment for qualitative comparisons between treatments. Summaries by major tree species and fuel treatment include percentage mortality in 2007, cumulative mortality in the final sampling year (2009), percentage of trees with moderately to severely burned crowns and mean percentage crown scorch. Minor species included any species with fewer than 10 individuals per treatment type and are tallied in the all trees category. Percentage of trees with bark beetle evidence is similarly summarised by major species and treatment type. No statistical comparisons were made on summarised data.

Generalised estimating equations and model selection

All statistical analyses were conducted in the R statistical software (R Development Core Team 2010). Because we expected within-unit correlation in tree mortality, we used generalised estimating equations (GEE) with a logit link (implemented with geeglm in the geepack R package; Højsgaard *et al.* 2005) and trees grouped by unit. Inference using GEE allows for consistent estimators even with unspecified correlation structures (Liang and Zeger 1986). Wald test statistics using the robust standard error estimates were used for inference for regression model coefficients.

For models with more than one predictor variable, we used forward model selection by first testing individual main effects, then two-way interactions, then any possible higher-order interactions (as many as 3-way interactions among our models). Because GEE use quasi-likelihood based metrics for model selection, metrics such as Akaike's information criterion (AIC) cannot be directly used. Pan (2001) proposes a quasi-likelihood alternative to AIC (QIC), with a similar penalty term for model complexity. Lower values of QIC are preferred in model selection. Final model selection depended on both the QIC values and a measure of model performance. Model performance was evaluated by the receiver operator characteristic (ROC) area under the curve values using the ROCR package in R (Sing *et al.* 2009). These ROC values range from 0 to 1. A value of 0.5 indicates the model performs no better than chance (Hosmer and Lemeshow 2000; Hood and Bentz 2007). Values >0.7 indicate that the model is acceptably better than chance, with higher values indicating better performance. In our final model selection, for subsets of models with similar QIC values we compared the ROC values. If the addition of another model coefficient resulted in only marginal improvement in both the QIC and ROC values, we did not include that coefficient. Final model selection did not include any variables or interactions that were not significant.

Crown scorch analysis

We evaluated inclusion of crown scorch as a predictor of tree mortality because it is an important variable in most published tree mortality models. Crown scorch in our dataset exhibits a bimodal distribution, with over 61% of the trees listed as either 0% or 100% crown scorch, making inference of crown scorch problematic. An overwhelming majority of trees assigned 0% crown scorch survived the fire, and those with 100% crown scorch did not survive the fire, resulting in a nearly 1–1 classification. To address this uneven distribution, we divided the data into three crown scorch classes (0%, intermediate and 100%) and excluded trees with moderate to severe crown consumption. We then performed a Chi-square contingency table test to determine if crown scorch classification is independent of fuel treatment. To predict the log-odds probability of mortality with crown scorch we used only the trees with intermediate crown scorch measures in the GEE model. Because other fire severity measures such as maximum bole char are highly correlated with percentage crown scorch, we did not evaluate models that combined percentage crown scorch with other measures of fire severity. Due to its uneven distribution and high correlation with treatment, crown scorch was not used as a predictor variable in subsequent analyses.

Delayed mortality continuation ratio model

We evaluated the effect of treatment and DBH across the three sampling years (2007 to 2009) using a continuation ratio model (CRM). The CRM is appropriate for hierarchical data and is a version of the Cox proportional hazards model (Harrell *et al.* 1998). We used an extended version of the CRM that allows for relaxation of the proportional hazards assumption (the assumption that the effect of the predictor variables is constant across the hierarchy; Harrell *et al.* 1998). In our application, the CRM evaluates the probability of mortality in a given year conditional on survival the previous year. We performed logistic regression using a GEE on the values Y_{ij}, to estimate the predictors of μ_{ij}. The model is:

$$\log\left(\frac{\mu_{ij}}{1-\mu_{ij}}\right) = \alpha_{ij} + \boldsymbol{\beta}_j \mathbf{X}_{ij}$$

where μ_{ij} is the probability individual i is classified as dead at time point j, given that individual has survived before time point j. α_j is the intercept value for each year evaluated, $\mathbf{X}_{ij}$ is the

Table 1. Summary of tree tallies, percentage mortality in 2007, cumulative percentage mortality in 2009, percentage of trees with moderately to severely burned crowns and mean and standard deviation (s.d.) of percentage crown scorch by tree species and treatment (control, thin-only and thin and prescribed burn (thin-Rx))

	Douglas-fir	Lodgepole pine	Ponderosa pine	Subalpine fir	Western larch	All species
Control						
Total trees (*n*)	1174	215	188	183	–	1892
2007 mortality (%)	58	85	55	80	–	65
2009 mortality (%)	72	95	63	98	–	78
Consumed crowns (%)	4	12	3	4	–	7
Mean (s.d.) crown scorch (%)	72 (39)	92 (24)	73 (36)	94 (17)	–	78 (36)
Thin-only						
Total trees (*n*)	947	103	573	68	–	1706
2007 mortality (%)	60	88	28	85	–	52
2009 mortality (%)	75	91	39	87	–	64
Consumed crowns (%)	7	17	2	10	–	6
Mean (s.d.) crown scorch (%)	76 (36)	94 (19)	57 (39)	91 (25)	–	71 (37)
Thin-Rx						
Total trees (*n*)	980	185	516	12	97	1796
2007 mortality (%)	37	63	28	75	19	36
2009 mortality (%)	46	72	34	83	20	44
Consumed crowns (%)	16	16	6	0	4	13
Mean (s.d.) crown scorch (%)	40 (43)	66 (45)	48 (39)	91 (18)	15 (36)	44 (43)

design matrix of predictor variables for individual *i* at time point *j* and $\boldsymbol{\beta}_j$ is the matrix of model coefficients for each year. If there are no significant interactions between year and the remaining predictor variables, then there is a common $\boldsymbol{\beta}$ matrix across all of the years.

Comparison of Douglas-fir and ponderosa pine mortality

We used GEE to evaluate the probability of mortality for the two most common tree species (Douglas-fir and ponderosa pine). We included only those units that contained at least 10 sampled individuals of each species, and we predicted the overall mortality of each species by year 2009. The third most common species, lodgepole pine, was unevenly distributed across units and was not suitable for inclusion in this analysis. The available predictor variables are species and treatment as factors, and diameter as a continuous predictor variable. Model selection was performed as described above.

Beetle attack analysis

In our final set of models, we evaluated the effect of fuel treatment and tree diameter on the probability of beetle attack on trees still alive in 2009. Because observations of beetle attack were made in 2009, beetle damage could not be used as a predictor of tree mortality across sampling years. Confining the analysis to living trees in 2009 reduced the total sample size and removed species effect from the final model, due to the sparse distribution of live trees by species across units.

Results

Summary statistics on tree mortality

The total number of surveyed trees is similar across treatments, but control units contain a greater number of understorey and hardwood species (*Alnus* spp., *Populus balsamifera* var. *balsamifera* and *Salix* spp.) that are absent or much rarer in treated units (Table 1). Most tree deaths occurred in the first year regardless of treatment, but mortality was much higher in control and thin-only units (65 and 52%) compared with thin-Rx units (36%). We found a similar trend among treatments in the percentage of trees that had died by the year 2009 (cumulative mortality; 78% for control, 64% thin-only, 44% thin-Rx). Cumulative lodgepole pine mortality is high in both control and thin-only units but somewhat lower in thin-Rx units (95 and 91% *v.* 72%). Subalpine fir is almost absent in thin-Rx units, so no comparison was possible across treatment types. Ponderosa pine exhibits the lowest mortality of all species. Still, thinning treatments clearly reduced mortality in ponderosa pine with 63% in control, 39% in thin-only and 34% in thin-Rx units. High ponderosa pine mortality in control units is associated with substantial crown scorch (73%). Douglas-fir mortality is comparable between control and thin-only units (72 and 75%) with much lower mortality in thin-Rx units (46%).

Table 2. Model of tree mortality with crown scorch as a predictor variable

The final model form is $\log(P/(1-P)) = \boldsymbol{\beta}_0 + \boldsymbol{\beta}_1\mathbf{X}_1^2$ where P = probability of mortality and $\mathbf{X}_1^2$ is the square of crown scorch (%) excluding trees with 0 and 100% crown scorch

Coefficient	Estimate	Robust s.e.	Wald	*P*(Wald)
$\boldsymbol{\beta}_0$ (intercept)	−2.80	0.121	539	<0.001
$\boldsymbol{\beta}_1$ (crown scorch^2)	0.00048	2.19×10^{-5}	473	<0.001

Percentage crown scorch

Percentage crown scorch as a single variable is a strong predictor of mortality (Table 2) with a ROC of 0.849. The

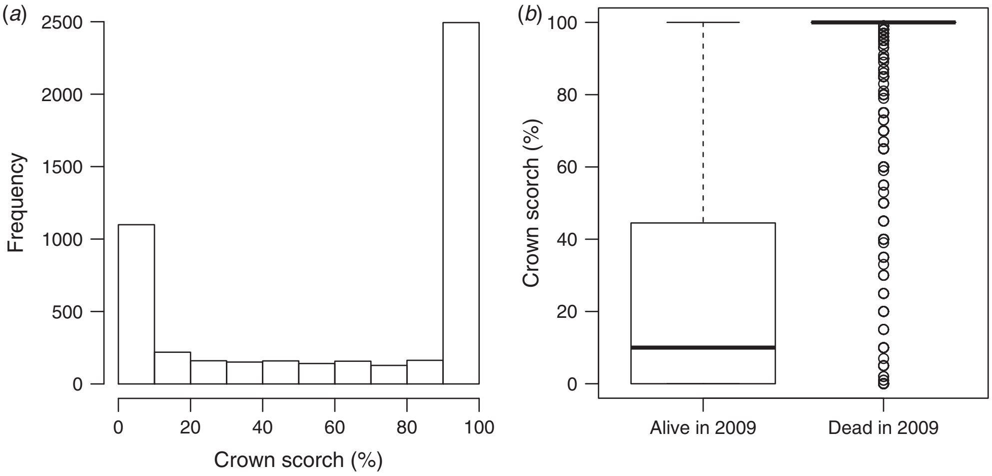

Fig. 2. (*a*) Histogram of crown scorch values across all units, showing a bimodal distribution of crown scorch. (*b*) Boxplot of crown scorch values to compare trees classified as alive in 2009 and trees classified as dead in 2009. The boxes encompass the interquartile range (IQR), the centre line the median, the vertical lines indicate ±1.5*IQR and the points data values outside of 1.5*IQR, showing that an overwhelming majority of trees classified as dead had 100% crown scorch.

distribution of crown scorch is highly skewed in our dataset: over 61% of trees have crown scorch values of either 0 or 100% and only a third had intermediate values (Fig. 2). Percentage crown scorch is also highly dependent on fuel treatment. Using three categories of crown scorch (0%, intermediate and 100%) and fuel treatment (control, thin-only and thin-Rx), a Chi-square contingency test demonstrates that whether a tree has 0%, intermediate or 100% crown scorch is significantly dependent on fuel treatment. Based on the residuals of the Chi-square test (Fig. 3), control units have more trees than expected in the 100% crown scorch class; whereas the thin-Rx units have more trees than expected in the 0% crown scorch class. The thin-only treatment has more trees than expected in the intermediate class.

Continuation ratio model (CRM) of tree mortality

The continuation ratio model (CRM) of tree mortality explains differences in predicted tree mortality across subsequent sampling years and treatment (Fig. 4, Table 3). Probability of tree mortality is lowest in thin-Rx units across all sample years, and trees in thin-only units have a slightly lower probability of mortality than do controls. One year post-fire, small-diameter trees are much more likely to die as a direct result of the wildfire than are larger trees across all three treatments, as evidenced by the significantly negative slope associated with DBH. Two years post-fire, the probability of subsequent tree mortality is significantly lower than of direct mortality (2007; see the $\boldsymbol{\beta}_1$ coefficient in Table 3) but still decreases significantly with increasing tree diameter. Three years post-fire, the probably of subsequent tree mortality is low for all treatments but in thin-only and control units, and large-diameter trees have a somewhat greater (but not significant) probability of mortality than do small trees. In contrast, the 2009 probability mortality in thin-Rx units, given survival in 2007 and 2008, is near zero across the range in sampled diameters. The ROC value for the final selected model is 0.838.

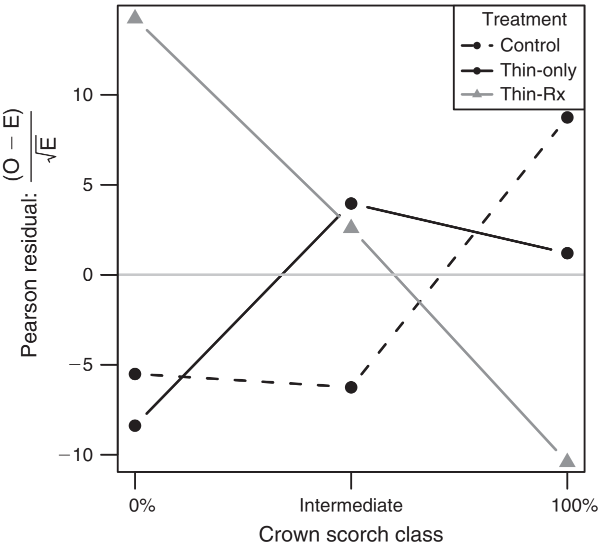

Fig. 3. Pearson residuals of the Chi-square contingency test, where positive values indicate more trees than expected and negative values, fewer trees than expected. Each line connects the residual value for each treatment. The control units have more trees than expected in the 100% crown scorch class, whereas the Thin-Rx units have more trees than expected in the 0% crown scorch class.

Mortality by species

The mortality by species model compares the probability of Douglas-fir and ponderosa pine mortality 3 years post-fire (Fig. 5). For either species there is no significant difference in mortality between control and thin-only units (Table 4), but the thin-Rx treatment has significantly lower mortality than the other treatments. Predicted mortality of ponderosa

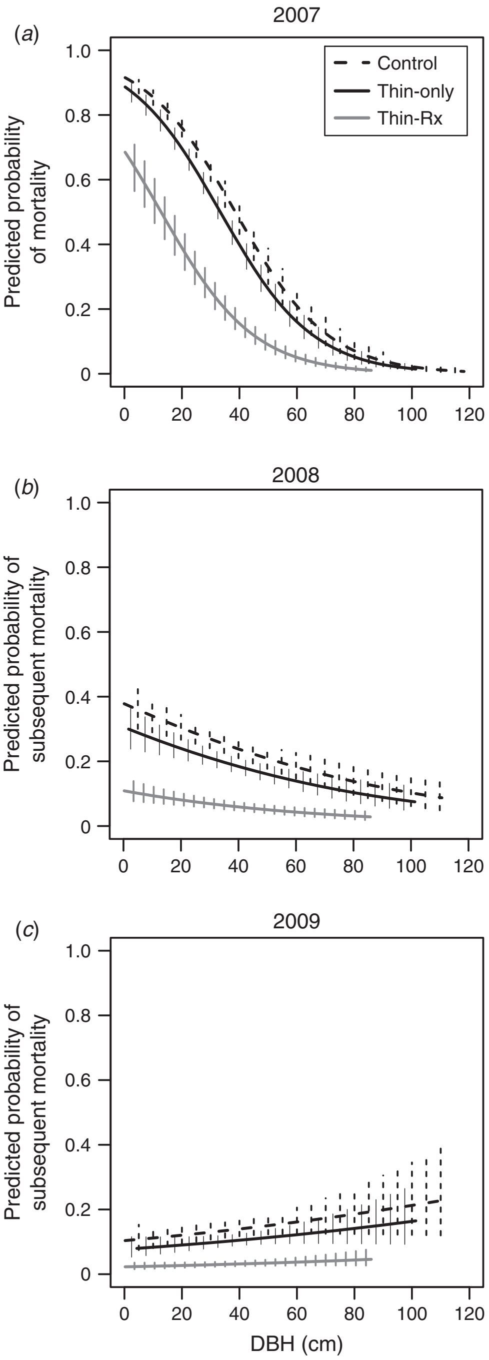

Fig. 4. Predicted probability (0–1) of tree mortality 1, 2 and 3 years post-fire modelled by treatment type and tree diameter using a continuation ratio model. Vertical lines represent 95% prediction intervals as estimated by the esticon() function in R. The 2007 model (*a*) predicts the probability of tree mortality 1 year post-fire, the 2008 model (*b*) predicts probability of subsequent mortality given survival in 2007 and the 2009 model (*c*) predicts probability of subsequent mortality given survival in 2008.

pine is significantly lower than that of Douglas-fir. Interactions (e.g. species × DBH or treatment × species) are not significant. The ROC for this model is 0.780.

Table 3. Continuation ratio model (CRM) of tree mortality

The final equation form is $\log(P/(1-P)) = \boldsymbol{\beta}_0 + \boldsymbol{\beta}_1\mathbf{X}_{11} + \boldsymbol{\beta}_2\mathbf{X}_{12} + \boldsymbol{\beta}_3\mathbf{X}_2 + \boldsymbol{\beta}_4\mathbf{X}_{31} + \boldsymbol{\beta}_5\mathbf{X}_{32} + \boldsymbol{\beta}_6\mathbf{X}_{11}\mathbf{X}_2 + \boldsymbol{\beta}_7\mathbf{X}_{12}\mathbf{X}_2$ where P = probability of mortality; $\mathbf{X}_{11} = 1$ if year 2008, 0 otherwise; $\mathbf{X}_{12} = 1$ if year 2009, 0 otherwise; $\mathbf{X}_2$ = DBH (cm); $\mathbf{X}_{31} = 1$ if a thin-only unit, 0 otherwise and $\mathbf{X}_{32} = 1$ if a thin-Rx unit, 0 otherwise

Coefficient	Estimate	Robust s.e.	Wald	*P*(Wald)
$\boldsymbol{\beta}_0$ (intercept)	2.397	0.126	362.532	<0.001
$\boldsymbol{\beta}_1$ (year 2008)	−2.890	0.174	275.212	<0.001
$\boldsymbol{\beta}_2$ (year 2009)	−4.554	0.251	329.846	<0.001
$\boldsymbol{\beta}_3$ (DBH)	−0.062	0.004	219.945	<0.001
$\boldsymbol{\beta}_4$ (thin-only)	−0.325	0.091	12.689	<0.001
$\boldsymbol{\beta}_5$ (thin-Rx)	−1.605	0.125	165.987	<0.001
$\boldsymbol{\beta}_6$ (2008*DBH)	0.045	0.005	73.685	<0.001
$\boldsymbol{\beta}_7$ (2009*DBH)	0.071	0.007	113.855	<0.001

Summary statistics on beetle evidence

Of the trees still living in 2009, 20% in control units, 34% in thin-only units and 8% in thin-Rx units exhibit evidence of beetle attack (Table 5). Few lodgepole pine trees survived to 2009, but those remaining had variable evidence of bark beetles: no beetles in control units, 44% in thin-only units and 6% in thin-Rx units. The percentage of trees with bark beetle evidence is comparable between ponderosa pine and Douglas-fir, but with markedly lower percentages in thin-Rx units than in thin and control units. Beetle evidence is highest in thin-only and lowest in thin-Rx units; fewer than 10% of trees are affected in thin-Rx units, regardless of species.

Beetle attack model

The probability of beetle attack 3 years post-fire increases significantly with increasing tree diameter (Table 6). Probability of beetle attack is dependent on fuel treatment with greatest probability of attack in thin-only treatments and lowest probability in thin-Rx units (Fig. 6). There is no significant interaction between tree diameter and treatment. The ROC for this model is 0.728.

Discussion

Existing models of tree mortality following fire generally include tree diameter and either crown scorch or bole char as key predictor variables (Peterson and Arbaugh 1986; Regelbrugge and Conard 1993; Ryan and Amman 1994; McHugh and Kolb 2003; Sieg *et al.* 2006; Hood and Bentz 2007). Our study also found that percentage crown scorch is a strong predictor of tree mortality. The skewed distribution of crown scorch towards values of 0 and 100% in our study is somewhat unusual and may be partially explained by differences in fire severity between treatments. The incidence of 100% crown scorch was much higher in control and thin-only units than in thin-Rx units (Fig. 3), leading to a high incidence of 100% crown scorch (Prichard *et al.* 2010). In many of the thin-Rx units, it was difficult to distinguish fire effects such as ground and bole charring from the recent prescribed burns *v.* the wildfire. However, it

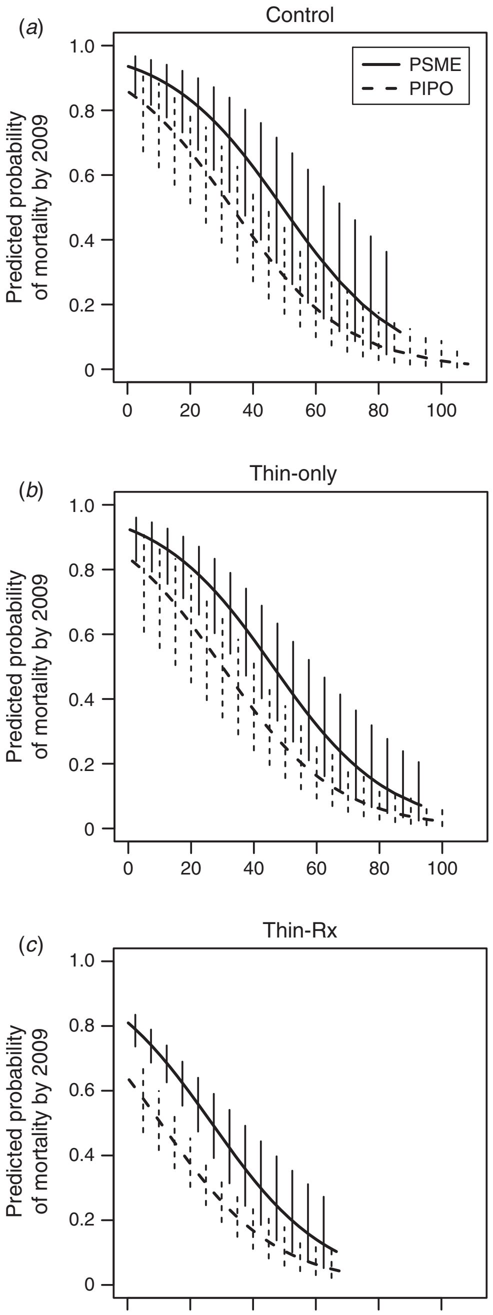

Fig. 5. Predicted probability of tree mortality through 2009 by tree species and fuel treatments including (*a*) control, (*b*) thin-only and (*c*) thin-Rx units. Vertical lines represent 95% prediction intervals as estimated by the esticon() function in R. PIPO, *Pinus ponderosa* (ponderosa pine); PSME, *Pseudotsuga menziesii* (Douglas-fir).

appeared that the wildfires generally burned intensely outside of thin-Rx units and wrapped around the edges of these units with only minor surface fires (e.g. patchy, discontinuous underburns) in the interior, leading to a low incidence of crown scorch in these units.

Table 4. Predicted probability of mortality by species 3 years post-fire

The equation form is: $\log(P/(1-P)) = \boldsymbol{\beta}_0 + \boldsymbol{\beta}_1\mathbf{X}_1 + \boldsymbol{\beta}_2\mathbf{X}_2 + \boldsymbol{\beta}_3\mathbf{X}_{31} + \boldsymbol{\beta}_4\mathbf{X}_{32}$ where P = probability of mortality by 2009 (3 years post-fire); $\mathbf{X}_1$ = DBH (cm); $\mathbf{X}_2$ = species (1 for Douglas-fir, 0 for ponderosa pine); $\mathbf{X}_{31} = 1$ if a thin-only unit, 0 otherwise and $\mathbf{X}_{32} = 1$ if a thin-Rx unit, 0 otherwise

Coefficient	Estimate	Robust s.e.	Wald	*P*(Wald)
$\boldsymbol{\beta}_0$ (intercept)	1.807	0.447	16.380	<0.001
$\boldsymbol{\beta}_1$ (DBH)	−0.054	0.009	36.447	<0.001
$\boldsymbol{\beta}_2$ (Species, Douglas-fir)	0.884	0.181	23.754	<0.001
$\boldsymbol{\beta}_3$ (thin-only)	−0.179	0.448	0.159	0.690
$\boldsymbol{\beta}_4$ (thin-Rx)	−1.230	0.379	10.555	0.001

Our results suggest that differences in mortality among the treatments are due to the variability in the crown scorch and consumption. In particular, the thin-Rx fuel treatment appears to have modified fire behaviour so as to reduce crown scorch, thereby decreasing the probability of tree mortality. Some existing models of tree mortality, including those in FOFEM (Ryan and Reinhardt 1988; Ryan and Amman 1994), are based on surveyed tree damage after prescribed burns, which likely lead to a more even distribution of crown scorch than we observed. Other studies that predict tree mortality following wildfire events report much lower overall mortality and also demonstrate a more uniform range of percentage crown volume scorched than in this study (e.g. McHugh and Kolb 2003; Sieg *et al.* 2006; Hood and Bentz 2007). However, in a study that includes fuel treatment as a predictor of tree mortality, Ritchie *et al.* (2007) also report stark differences between crown scorch and mortality in untreated forest *v.* units treated with thinning and prescribed fire.

Our sampling design in treated and untreated fuels enabled us to evaluate models using stand data (i.e. tree diameter, species and fuel treatment) that could be collected before an actual fire event. Crown damage is the primary cause of tree mortality and also is strongly dependent on fuel treatment (Fig. 3). We previously determined that the probability of tree mortality 3 years post-fire is significantly reduced in treated units relative to controls, with much greater reductions in thin-Rx units than in thin-only units (Prichard *et al.* 2010). Our findings are similar to those reported by Ritchie *et al.* (2007), but their tree mortality model represents 1 year post-fire and uses distance from treatment edge as a predictor variable in addition to tree diameter. The CRM model presented in this study evaluates mortality as a function of years since fire (conditional on survival the previous year), treatment and tree diameter (Fig. 4). Direct mortality, as measured 1 year post-fire, is predominantly in small-diameter trees. The probability of subsequent mortality is still highest for small-diameter trees 2 years post-fire, but by the third year post-fire, the probability of subsequent tree mortality is low across all treatments and is slightly higher (but not significantly so) for large trees in control and thin-only units than for smaller trees, although this trend is not significant.

Two factors likely contributed to the reversal in diameter effect from 1 to 3 years post-fire. First, with lower crown heights

Table 5. Summary of the number of live trees in 2009, percentage of trees with bark beetle evidence, and mean and standard deviation (s.d.) of tree diameters by tree species and treatment (control, thin-only and thin and prescribed burn (thin-Rx))

	Douglas-fir	Lodgepole pine	Ponderosa pine	Western larch	All species
Control					
Live trees in 2009 (n)	324	10	69	–	410
Bark beetles (%)	20	0	19	–	20
Mean DBH (cm)	36 (15)	18 (8)	40 (18)	–	36 (15)
Thin-only					
Live trees in 2009 (n)	240	9	348	–	606
Bark beetles (%)	32	44	37	–	34
Mean DBH (cm)	42 (16)	29 (11)	41 (16)	–	41 (16)
Thin-Rx					
Live trees in 2009 (n)	534	52	337	78	1007
Bark beetles (%)	9	6	7	9	8
Mean DBH (cm)	31 (16)	12 (13)	31 (14)	32 (19)	30 (16)

Table 6. Final model to predict probability of beetle attack given survival through 2009

The equation form is: $\log(P/(1-P)) = \boldsymbol{\beta}_0 + \boldsymbol{\beta}_1\mathbf{X}_1 + \boldsymbol{\beta}_2\mathbf{X}_{21} + \boldsymbol{\beta}_3\mathbf{X}_{22}$ where P = probability of bark beetle evidence for trees still living in 2009; $\mathbf{X}_1$ = DBH (cm); $\mathbf{X}_{21} = 1$ if a thin-only unit, 0 otherwise and $\mathbf{X}_{22} = 1$ if a thin-Rx unit, 0 otherwise

Coefficient	Estimate	Robust s.e.	Wald	P(Wald)
$\boldsymbol{\beta}_0$ (intercept)	−2.14	0.19	123.72	<0.001
$\boldsymbol{\beta}_1$ (DBH)	0.02	0.00	26.34	<0.001
$\boldsymbol{\beta}_2$ (thin-only)	0.71	0.16	19.65	<0.001
$\boldsymbol{\beta}_3$ (thin-Rx)	−0.89	0.18	25.15	<0.001

and thinner bark, small-diameter trees are more vulnerable to direct fire effects than are large-diameter trees (Agee 1993; Fowler and Sieg 2004). Second, large-diameter trees may be particularly susceptible to secondary mortality agents including bark beetles and drought stress (McHugh and Kolb 2003; Wallin *et al.* 2003). Pre-existing fire scars and basal accumulations of litter and bark slough around large-diameter trees can contribute to longer fire residence times and damage cambial tissue and fine root systems (Kolb *et al.* 2007). Resulting heat injury to boles and fine roots may not directly kill large-diameter trees but can predispose them to mortality from insects, pathogens and drought (McHugh and Kolb 2003). In this study, we found that 3 years post-fire, large-diameter trees within thin-only and control units had slightly higher (but non-significant) probability of dying than did small-diameter trees, but that the probability of tree death in thin-Rx units was near zero across a range of tree diameters. Trees in control and thin-only units that survived the wildfire experienced greater crown scorch than thin-only units, which in turn may have increased their susceptibility to secondary mortality agents.

The prevalence of Douglas-fir and ponderosa pine across all units allowed us to statistically compare the probability of mortality between the two species (Fig. 5). The resulting model is not surprising given what is known about the fire adaptations of these tree species: ponderosa pine has a lower probability of mortality following fire than Douglas-fir across all three treatments. With thick bark and tendency to shed lower branches at a young age, ponderosa pine and western larch are more resistant to fire than are Douglas-fir (Agee 1993; Baker 2009), and they indeed have the lowest mortality of all species (Table 1). The uneven occurrence of western larch across sampling units prevented its inclusion in a species-based mortality model. Although we were unable to include beetle attack as a predictor of mortality, it is possible that the high incidence of Douglas-fir beetle following the wildfires may have contributed to higher mortality in Douglas-fir than in ponderosa pine. Most Douglas-fir trees with beetle evidence were likely attacked by Douglas-fir beetle, whereas ponderosa pine may have been attacked by a less lethal species or more successfully pitched out attacking beetles (Six and Skov 2009).

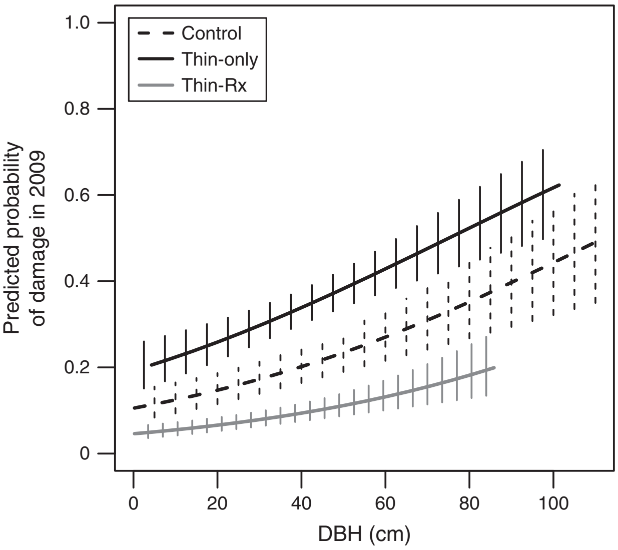

Fig. 6. Predicted probability of bark beetle attack 3 years post-fire (dataset includes all trees recorded as living in 2008 and sampled in 2009). Vertical lines represent 95% prediction intervals as estimated by the esticon() function in R.

The beetle attack model is based on observations of beetle evidence 3 years post-fire and predicts the probability of beetle attack as a function of tree diameter and fuel treatment. Probability of beetle attack is clearly highest for large-diameter trees. This result has also been reported in other studies (Wallin *et al.* 2003; Cunningham *et al.* 2005; Hood and Bentz 2007) and reflects the preference of bark beetles for larger-diameter trees with thicker phloem. Many bark beetle species, including Douglas-fir beetle, preferentially attack larger-diameter trees and are attracted to trees with fire injuries (Wallin *et al.* 2003; Kolb *et al.* 2006; Hood and Bentz 2007). We also found that the probability of beetle attack is highest in thin-only treatments and lowest in thin-Rx units. Because trees in thin-Rx units were generally subjected to lower fire severity, as evidenced by significantly less crown scorch than thin-only and control units, trees were likely less vulnerable to bark beetle attack (Wallin *et al.* 2003; McHugh and Kolb 2003; Six and Skov 2009).

Management implications

This study contributes to growing evidence of the influence of fuel treatments and species on post-fire tree mortality. Models of post-fire tree mortality that use stand data collected before a fire event could have numerous applications, including fuel treatment planning and prioritisation, predicting levels of mortality in a managed wildfire, post-fire restoration, habitat and water quality assessments (Brown *et al.* 2003), and salvage and hazard tree prioritisation (Peterson *et al.* 2009). For example, areas that have been recently treated with prescribed burning may be less vulnerable to direct and indirect effects of wildfire and have greater odds of long-term survival following the event than do thin-only treatments or unmanaged forests. Beetle attack models may be used to evaluate areas of particular management concern for secondary insect attack following fire or to prioritise areas for treatment to make them more resilient to future wildfires and subsequent bark beetle outbreaks (Fettig *et al.* 2007; Six and Skov 2009).

Suitable independent datasets were not available for model testing and development, and without rigorous testing against independent datasets, we do not know how applicable these models are to other locations and wildfire events. Each wildfire event occurs within a unique set of circumstances, and several factors likely limit broad extrapolation of our models. First, the Tripod Complex fires burned within an extreme fire weather event, including a record-setting month of August with high temperatures, low relative humidity and no recorded precipitation (Western Regional Climate Center, http://www.wrcc.dri.edu, accessed 6 July 2012). For this reason, percentage crown scorch volume and tree mortality in untreated and thin-only units may have been particularly high. In contrast, thin-Rx treatments appear to have been effective at mitigating wildfire severity even under these extreme fire weather conditions. Second, the fire event was not followed by prolonged drought. Extended drought following the wildfires could have further stressed surviving trees and altered observed patterns and levels of tree mortality. Third, source populations of Douglas-fir beetles were present following the fire event and likely exerted a strong influence on delayed mortality of Douglas-fir. Depending on the availability of source populations, other fire events may not be followed by outbreaks of Douglas-fir beetle or other bark beetle species (Fowler and Sieg 2004; Sieg *et al.* 2006). Finally, the prescribed burns in our study were all documented as successful with over 90% coverage and consumption of fine woody fuels. Including less successful prescribed burns or burns that had been conducted more than 10 years before the wildfire could have weakened our models of tree mortality based on fuel treatment.

In conclusion, this study contributes further evidence of the effectiveness of thinning and prescribed burning on reducing post-fire tree mortality. Results demonstrate a strong correspondence between percentage crown scorch and fuel treatment and may assist in the future development of operational models that use variables that can be collected before a wildfire event (e.g. tree size, species and fuel treatment) to estimate potential tree mortality following wildfires. We also present evidence that a combination of thinning and prescribed burning is associated with a lower incidence of bark beetle attack following wildfire. Rigorous testing and model validation would be required to evaluate the applicability of these models to other dry forest ecosystems.

Acknowledgements

This study was funded by the Joint Fire Science Program and the US Forest Service, Pacific Northwest Research Station. We thank members of the Fire and Environmental Research Applications team including Travis Freed, Cameron Balog, Jon Dvorak, Amy Jirka, Phil Monsanto, Joe Restaino, Shawn Smith and Aarin Sengisirirak for field assistance, Paige Eagle for database support and Robert Norheim for GIS support. We thank John Daily, Gary Reed, Tom Ketchum, Meg Trebon, Tom Leuschen and Rick Lind for information on the Tripod Complex fires. We thank Connie Mehmel, Roger Ottmar, David Peterson, John Daily and Clint Wright and three anonymous reviewers for constructive reviews of this manuscript.

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