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A comparison of landscape fuel treatment strategies to mitigate wildland fire risk in the urban interface and preserve old forest structure

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ABSTRACT

We simulated fuel reduction treatments on a 16,000 ha study area in Oregon, US, to examine tradeoffs between placing fuel treatments near residential structures within an urban interface, versus treating stands in the adjacent wildlands to meet forest health and ecological restoration goals. The treatment strategies were evaluated by simulating 10,000 wildfires with random ignition locations and calculating burn probabilities by 0.5 m flame length categories for each 30 m × 30 m pixel in the study area. The burn conditions for the wildfires were chosen to replicate severe fire events based on 97th percentile historic weather conditions. The burn probabilities were used to calculate wildfire risk profiles for each of the 170 residential structures within the urban interface, and to estimate the expected (probabilistic) wildfire mortality of large trees (>53.3 cm) that are a key indicator of stand restoration objectives. Expected wildfire mortality for large trees was calculated by building flame length mortality functions using the Forest Vegetation Simulator, and subsequently applying these functions to the burn probability outputs. Results suggested that treatments on a relatively minor percentage of the landscape (10%) resulted in a roughly 70% reduction in the expected wildfire loss of large trees for the restoration scenario. Treating stands near residential structures resulted in a higher expected loss of large trees, but relatively lower burn probability and flame length within structure buffers. Substantial reduction in burn probability and flame length around structures was also observed in the restoration scenario where fuel treatments were located 5–10 km distant. These findings quantify off-site fuel treatment effects that are not analyzed in previous landscape fuel management studies. The study highlights tradeoffs between ecological management objectives on wildlands and the protection of residential structures in the urban interface. We also advance the application of quantitative risk analysis to the problem of wildfire threat assessment.

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1. Introduction

Large investments in wildland fuel reduction projects are being made on federal lands in many regions within the United States in an ongoing effort by land management agencies to reduce human and ecological losses from catastrophic wildfire (USDA and USDI, 2001; HFRA, 2003; Sexton, 2006). The implementation of these projects continues to challenge planners as they attempt to reduce fuels over extensive areas while addressing multiple and often conflicting federal planning regulations, management objectives, and public expectations with finite budgets (Agee, 2002a; Johnson et al., 2006; Sexton, 2006; Winter et al., 2004; Dicus and Scott, 2006). Federal lands provide a broad array of ecological benefits including critical habitat for protected species, drinking water, wood products, carbon storage, and scenic and recreational opportunities, to name a few. Large, destructive wildfires are a growing threat to these values, and it is clear that landscape scale changes in forest structure and fuel loadings must be accomplished to significantly alter wildfire behavior, reduce wildfire losses, and achieve longer term fire resiliency in forests (e.g. Agee et al., 2000; Finney, 2001; Peterson et al., 2003; Graham et al., 2004). The most efficient way to achieve these long-term landscape goals remains unclear, and there are different perceptions on the relative role and effectiveness of management activities versus natural and managed wildfire to reduce fuels (cf. Agee, 2002a; Finney and Cohen, 2003; Reinhardt et al., 2008). Management science has generated new concepts and guidelines for developing landscape-tailored spatial designs that leverage mechanical thinning, underburning, and wildfire use to meet wildland fire objectives (Finney,

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2001; Finney et al., 2006; Vaillant, 2008; Schmidt et al., 2008). In particular, there is growing empirical and experimental evidence for optimal spatial and temporal treatment patterns (Finney et al., 2005, 2007; Vaillant, 2008), although it also is becoming clear that policies, constraints, and regulations that restrict treatment location, type, and total area treated, can significantly degrade the performance of these strategies (Finney et al., 2007). For example, policy direction that prioritizes treatments to protect highly valued resources (conservation reserves, wildland urban interface, USDA and USDI, 2001) at the expense of larger scale restoration to create fire resilient forests may well compromise a cohesive strategy to reduce adverse wildfire impacts.

A typical policy paradox exists in the Blue Mountains province in eastern Oregon, US, where extensive fuels build up is being addressed with accelerated fuel reduction treatments. Following guidelines set forth in the 2001 National Fire Plan (USDA and USDI, 2001), planners on national forests have initiated wildland urban interface (WUI) fuel treatment projects adjacent to many of the small towns and dispersed settlements. This focus on WUI fuel treatment projects has been repeated throughout the western US (Schoennagel et al., 2009). Roughly during the same period, a number of policy decisions also directed managers to design and invest in forest restoration projects to preserve and enhance remaining late-old forest structure (USDA and USDI, 1994; HFRA, 2003). Old forests, particularly in the dry ecotypes of the interior Pacific Northwest, have been heavily impacted by a long history of selective logging (Hessburg et al., 2005; Wales et al., 2007). These old forest stands are now highly valued for wildlife habitat, carbon storage, and fire resiliency (Agee, 2002b; Franklin and Agee, 2003; Hessburg et al., 2005; Spies et al., 2005; Thomas et al., 2006; Hurteau et al., 2008), especially those supporting early seral tree species, such as ponderosa pine (*Pinus ponderosa* C. Lawson) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). The steep decline and increasing value of old forest led to a 1994 decision by the USDA Forest Service in the Pacific Northwest region to halt harvesting of live trees with diameter at breast height of 21 in (53.3 cm) or greater on the dry forests east of the Cascade Mountain range. In addition, management activities that reduced late-old forest structure below levels established by the historical range of variability were prohibited. The protection of large trees and late-old forest structure remains a key objective in National Forests and is part of an ongoing restoration strategy to re-create networks of fire resilient forest stands within which natural fire could be re-introduced to manage fuel loadings over time (USDA Forest Service, 2008).

The long-term compatibility of management objectives to protect property values within relatively small WUIs versus meeting landscape restoration goals is not well understood. In fact, there are few case studies that examine tradeoffs among landscape fuel treatment strategies on fire behavior and fire effects. Finney et al. (2007) examined the effect of different spatial patterns on large fire spread, but fire intensity and effects on human and ecological values were not considered. Ager et al. (2007a) examined effects of different treatment intensities (area treated) on northern spotted owl habitat (*Strix occidentalis caurina*), but policy tradeoffs were not considered.

Given the difficulty with implementing landscape studies to analyze alternative treatment strategies, we employed computer simulation to explore alternative fuel treatment strategies on a typical WUI fuels reduction project in Eastern Oregon (Wallace, 2003). We estimated expected wildfire-caused mortality of highly valued large trees when fuel treatments were prioritized based on distance to residential structures. We then studied an alternative scenario that prioritized fuel treatments to overstocked stands on the adjacent wildlands to help achieve stand restoration objectives and preserve large trees. Our methods combined formal risk analyses (Finney, 2005; Scott, 2006; Society for Risk Analysis, 2006) with wildfire simulation methods (Finney et al., 2007) and provided a framework to quantitatively measure performance of the fuel treatments with risk-based measures (GAO, 2004). The findings from this study help understand the tradeoffs between competing fuel treatment investment strategies to mitigate wildfire-caused losses.

2. Materials and methods

2.1. Study area

The study area is 30 km long immediately north of La Grande, OR, where the forested slopes of Mt. Emily and adjoining ridgeline transitions to agricultural lands in the Grande Ronde Valley (Figs. 1 and 2). A project boundary was established as part of the Mt. Emily landscape fuels analysis project on the Wallowa-Whitman National Forest and was based on consideration of major drainages, natural breaks in vegetation and topography, and land ownership boundaries (Fig. 2). The project area enclosed 16,343 ha with 58% as federally managed lands. For this study, the Mt. Emily WUI was defined as the privately owned land on the east side of the project area, and the wildland was considered the federally managed land to the west (Fig. 2). About 12,259 ha of the study area is forested based on a 10% canopy closure definition used in the Wallowa-Whitman National Forest Plan. The forest composition ranges from dry forests of ponderosa pine to cold forests dominated by subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), and a transition zone containing grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), Douglas-fir (*P. menziesii*), and western larch (*Larix occidentalis* Nutt.). Forest Service lands are valued for a number of resources including summer and winter range for Rocky Mountain elk (*Cervus elaphus*), old growth, wood products, recreation, and scenic qualities.

The fire history of the Blue Mountains province points to wildfire as a dominant disturbance agent (Fig. 1). Fire history data were obtained from the Umatilla National Forest GIS library including perimeters greater than 20 ha 1890 to 2007 (http://www.fs.fed.us/r6/data-library/gis/umatilla/, Thompson and Johnson, 1900; Plummer, 1912), and ignitions 1970 to 2007. Although data prior to 1930 are incomplete, the record indicates that at least 1 million ha have burned out of 2.23 million ha total area of federally managed lands (Fig. 2, Umatilla, Wallowa-Whitman, and Malheur National Forests) from 1890 to present. Approximately 64% of the area burned resulted from fires since 1970.

The Mt. Emily area in particular was identified in the National Fire Plan (USDA and USDI, 2001) as high risk due to the density of rural homes and the potential for extreme fire behavior in the surrounding forests. Surface fuel loading exceeded 140 metric t/ha in some areas, and many of the stands were overstocked and contained excessive dead ladder fuel (Wallace, 2003). Fuel accumulations accelerated after the 1980–1986 western spruce budworm (*Choristoneura occidentalis*) epidemic that caused extensive tree mortality. To date, fire suppression in the Mt. Emily project area and surrounding forests has been very effective, with 99% of the fires in the past 30 years contained at less that 5 ha. The last large fire in the vicinity (Rooster Fire, 1973, Fig. 2) burned 2511 ha and reached the outskirts of La Grande, OR, where several residences were destroyed.

2.2. Vegetation and fuel data

A stand polygon map was obtained from the GIS library at the La Grande Ranger District. Stand boundaries outside of Forest Service lands were delineated on digital orthophotos from 2000. Data on

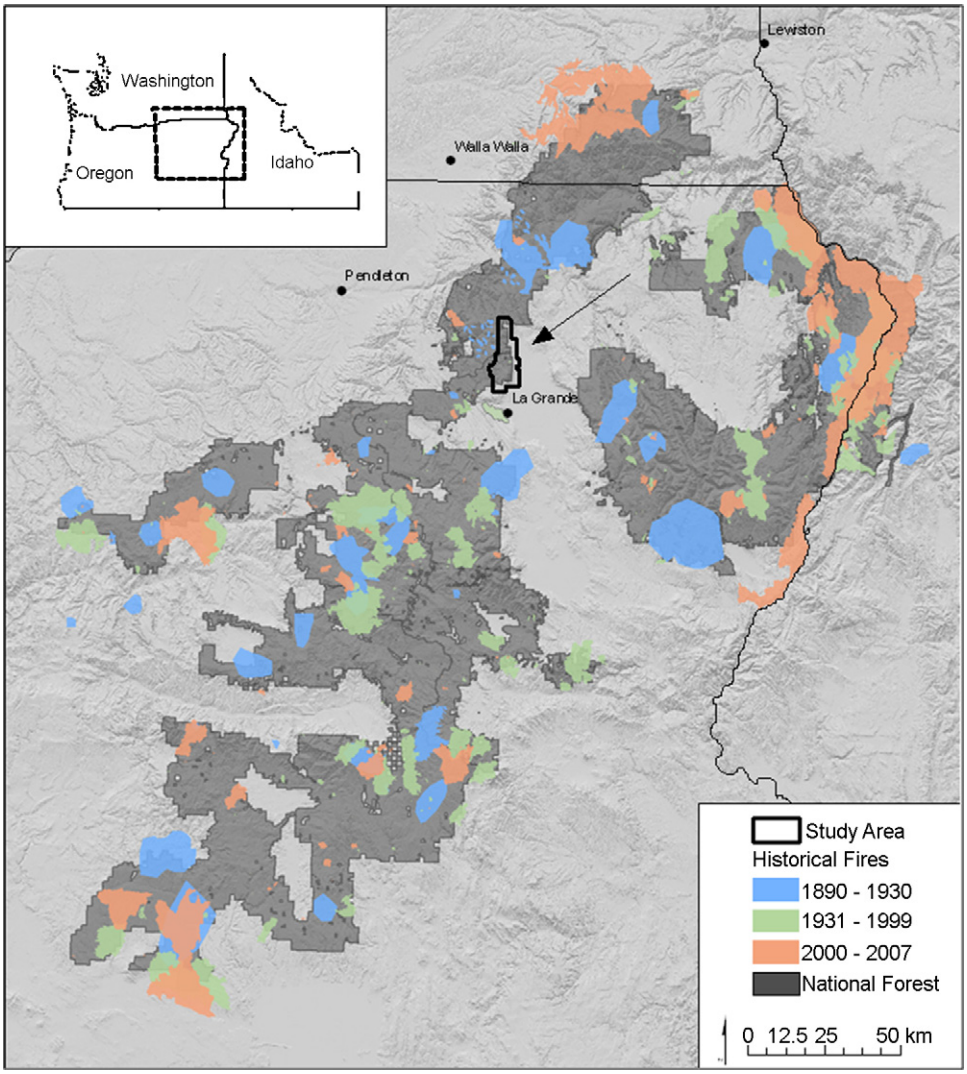

Fig. 1. Map of the Blue Mountains province showing national forest land (Umatilla, Wallowa-Whitman, and Malheur), historic fire perimeters (1890–2007), and Mt. Emily study area outline (arrow).

stand density by tree species and 2.5 cm diameter class were obtained from stand exams and photo-interpretation of 1:12,000 aerial color photos taken in 1998. Stand-specific data on surface fuel loadings were derived in several ways. The fuel loadings on about 1500 ha of the study area were estimated in the field as part of prescription development for a fuel reduction project. In this process, stand surface fuel conditions were matched to the photo series of Fischer (1981). These fuel loadings were then extrapolated to the remaining stands by using aerial photo-interpretation, stand exam data on plant association and stand structure, and local knowledge of stand conditions. Line transect sampling (Hilbruner and Wordell, 1992) was used to calibrate the photo series for extremely high fuel loadings found within many of the old forest stands.

2.3. Simulating management scenarios and prescriptions

We modeled forest vegetation and fuels using the Blue Mountains variant of the Forest Vegetation Simulator (FVS, Dixon, 2003), and the Fire and Fuels Extension to FVS (FVS-FFE, Reinhardt and Crookston, 2003). FVS is an individual-tree, distance-independent growth and yield model that is widely used to model fuel treatments and other stand management activities (Havis and Crookston, 2008). The Parallel Processing Extension to FVS (FVS-PPE, Crookston and Stage, 1991) was employed to model spatially explicit treatment constraints and treatment priorities as described below. FVS simulations and processing of outputs were completed within ArcFuels (Ager et al., 2006).

We simulated six treatment intensities by constraining the total treatment area to 0, 10, 20, 30, 40, and 66% of the forested lands. The 66% area treated all stands that qualified for treatment based on stand stocking as described below. We then applied two spatial treatment priorities, one based on stand density index (SDEN, Cochran et al., 1994), the other based on residential density (RDEN). The RDEN scenario prioritized stands based on the spatial density of residential structures generated from an interpolated point map using inverse distance-weighting. The SDEN scenario assigned the highest priority for treatment to the most overstocked stands as measured by the current stand density index (SDI) relative to the site potential (Cochran et al., 1994). Stand density index is a broad index of stand health and the potential for crown fire behavior (Keyes and O'Hara, 2002) and is widely used in the Blue Mountains and elsewhere to prioritize stands for restoration and fuels reduction treatment. Stands with high SDI in the study area also contained the highest density of large trees. Stands prioritized in the RDEN alternative were also required to exceed SDI thresholds.

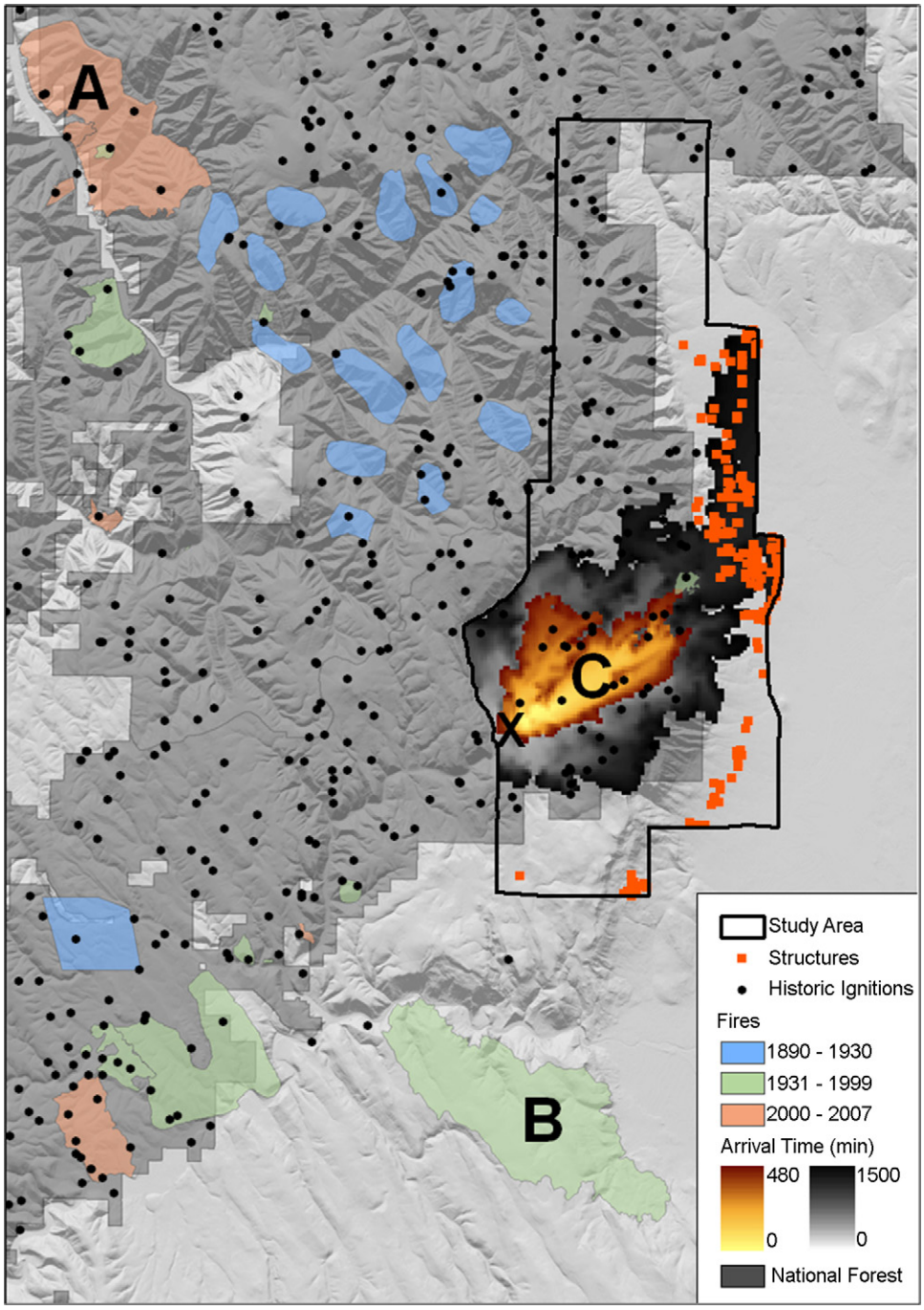

Fig. 2. Vicinity map of the Mt. Emily study area showing residential structures, mapped large fire perimeters (>20 ha) from ca. 1890–2007, historic lightning ignitions (1970–present), and two examples of a simulated fire (C) within the project area from a common ignition point (X). The simulated fires represent a burn period of 480 min and 1500 min (see legend). Spread of the 1500 min simulated fires was terminated at the project boundary on east edge where the study area borders agricultural lands. Fire perimeters prior to 1930 are approximate and partially complete and based on Plummer (1912) and Thompson and Johnson (1900). The 1973 Rooster fire (A) and the 2005 Milepost 244 fire (B) were both ignited by railroad equipment.

The resulting combination of 6 treatment intensities and 2 spatial priorities yielded outputs for 12 simulation runs. However, because the 0% and 66% treatment levels used identical simulation parameters for both SDEN and RDEN (the 66% treated all eligible stands), we chose the SDEN outputs to report here. Simulation outputs for the duplicate runs were found to be within 1.5% or less for all outputs analyzed.

The specific parameters for the fuel reduction prescription were chosen based on operational guidelines from the Mt. Emily fuels reduction project (Wallace, 2003) and elsewhere on the Umatilla and Wallowa-Whitman National Forests. The treatments were simulated with FVS and consisted of a 3-year sequence of thinning from below, site removal of surface fuels, and underburning. Underburning and mechanical treatment of surface fuels was simulated with the FVS-FFE keywords SIMFIRE and FUELMOVE (Reinhardt and Crookston, 2003). Fuel treatment prescriptions for thinning from below had a 21 in (53.3 cm) diameter limit and specified retention of fire tolerant tree species (ponderosa pine, western larch, and Douglas-fir). Stand density index thinning from below improves vigor for large trees, reduces crown fire potential

Table 1
Weather and fuel moisture parameters used in wildfire and underburn simulations. The latter were implemented as part of thinning treatments. Underburn conditions were obtained from fuel specialists in the Blue Mountains. Values for wildfire are 97th percentile weather conditions and were calculated from local weather station data as described in the methods. NA, not applicable.

Variable	Underburn	Wildfire
Temperature (°C)	21.1	32.2
1-h fuel (0–0.64 cm) (%)	6	3
10-h fuel (0.64–2.54 cm) (%)	8	4
100-h fuel (2.54–7.62 cm) (%)	10	6
1000-h fuel (>7.62 cm) (%)	20	7
Live woody fuel (%)	125	64
Duff (%)	20	20
10-min average wind speed at 6.1 m above ground (km/h)	<4	4
Maximum wind gusts (km/h)	NA	32

(Keyes and O'Hara, 2002), and promotes the development of fire resilient single story old forest structure. Stands were thinned to a target SDI of 35% of the maximum for the stand. Surface fuel treatments simulated the removal of 90% of the 7.6–14.8 cm and 40% of the 2.5–7.6 cm surface fuels (Wallace, 2003) using mechanical methods. Underburning was then simulated using weather conditions and fuel moisture guidelines provided by fuels specialists on the La Grande Ranger District (Ager et al., 2007b; Table 1). The prescription was well supported by empirical studies as effective for reducing potential fire behavior (van Wagtendonk, 1996; Peterson et al., 2003; Stephens and Moghaddas, 2005; Stephens et al., 2009; Vaillant et al., 2009).

Each treatment alternative was simulated with FVS and the post-treatment outputs for canopy bulk density (kg/m^3), height to live crown (ft), total stand height (ft), canopy cover (%), and fuel model were then used to build 30 m × 30 m raster input files for fire simulations described below. We overrode FFE fuel model selection on treated stands and assumed a post-treatment fire spread rate and behavior equal to fuel model TL1 (Scott and Burgan, 2005) based on local experience and input from fire managers (personal communication, Burry, T., La Grande Ranger District, La Grande, OR, November 2007). As found in previous studies (Ager et al., 2007b), the Blue Mountains variant of the FFE is not well calibrated to predict post-treatment fuel models.

2.4. Wildfire simulations

We simulated wildfires using the minimum travel time (MTT) fire spread algorithm of Finney (2002), as implemented in a command line version of FlamMap called "Randig" (Finney, 2006). The MTT algorithm replicates fire growth by Huygens' principle where the growth and behavior of the fire edge is a vector or wave front (Richards, 1990; Finney, 2002). This method results in less distortion of fire shape and response to temporally varying conditions than techniques that model fire growth from cell-to-cell on a gridded landscape (Finney, 2002). Extensive testing over the years has demonstrated that the Huygens' principle as originally incorporated into Farsite (Finney, 1998) and later approximated in the more efficient MTT algorithm can accurately predict fire spread and replicate large fire boundaries on heterogeneous landscapes (Sanderlin and Van Gelder, 1977; Anderson et al., 1982; Knight and Coleman, 1993; Finney, 1994; Miller and Yool, 2002; LaCroix et al., 2006; Ager et al., 2007a; Arca et al., 2007; Krasnow et al., 2009; Carmel et al., 2009; Massada et al., 2009). The MTT algorithm in Randig is multi-threaded (computations for a given fire are performed on multiple processors), making it feasible to perform Monte Carlo simulations of many fires (>50,000) to generate burn probability surfaces for very large (>2 million ha) landscapes (Ager and Finney, 2009). The MTT algorithm is now being applied daily for operational wildfire problems throughout the US (http://www.fpa.nifc.gov, http://wfdss.usgs.gov/wfdss/WFDSS_About.shtml). In contrast to Farsite (Finney, 1998), the MTT algorithm assumes constant weather and is used to model individual burn periods within a wildfire rather than continuous spread of a wildfire over many days and weather scenarios. Relatively few burn periods generally account for the majority of the total area burned in large (e.g. >5000 ha) wildfires in the western U.S., and wildfire suppression efforts have little influence of fire perimeters during these extreme events.

For each treatment alternative we simulated 10,000 burn periods assuming random ignition locations within the study area. The number of fires was adequate to ensure that >99% of 30 m × 30 m pixels with burnable fuels in the study area were burned at least once (average = 102 fires per pixel). The simulations for the present study were performed on a desktop computer equipped with 8 quad-core AMD Opteron™ processors (64 bit, 2.41 GHz) with 64GB RAM and required 2 h of processing per scenario.

Simulation parameters were developed to reflect likely future scenarios for escaped wildfires within the study area based on historical fire data on the surrounding National Forests as well as personal communication with fire specialists on the Wallowa-Whitman National Forest (Fig. 2). We assumed a constant fuel moisture, wind speed, wind direction, and varied burn periods by sampling a frequency distribution as described below. Fuel moistures were derived for 97th percentile weather scenario from local remote weather stations (Wallace, 2003; Ager et al., 2007b; Table 1). The wind scenario was developed with input from local fire managers to build a likely extreme wind scenario that called for of 32 km/h winds at 235° azimuth, reflecting a July–August cold-front weather system that typically generates the vast majority (>99%) of lightning and ignitions. Although local automated weather stations in the area indicated northwesterly winds during peak fire season (July–August), lightning ignitions are rare under these conditions. Two recorded fires in the immediate area (Fig. 2, Rooster and Milepost 244) exhibited a dominant spread from northwest winds, but both of these fires were ignited by railroad equipment.

Burn period durations were initially developed with Randig simulations to generate fire size distributions consistent with expectations of an extreme event from local managers and historic fire sizes (Fig. 3). These outputs were then compared to available (1999–2007) burn period data from fires in the Blue Mountains

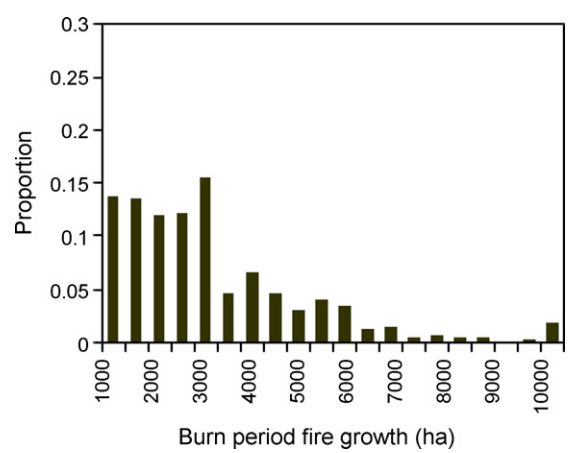

Fig. 3. Frequency distribution of area burned obtained with Randig by sampling burn period duration from the distribution in Table 2 for the Mt. Emily study area. The distribution of burn periods in Table 2 was initially developed based on input from fire specialists on the Wallowa-Whitman National Forest and from qualitative observations on historic fire events in the area. Burn periods represent a daily spread event under extreme weather for a large fire.

Table 2
Distribution of burn periods (min) used for the wildfire simulations. The distribution was created to generate fire size distributions that replicated expected fire sizes in the Mt. Emily area.

Burn period (min)	Proportion
500	0.45
1000	0.27
1500	0.18
2000	0.05
2500	0.03
5000	0.01

province (http://famtest.nwcg.gov/fam-web/). Specifically, burn period (i.e. daily spread) data were obtained for five large fires (6500–64,000 ha) for a total of 40 days (Fig. 4). Daily spread events with less than 1000 ha were excluded from the comparison based on our concept of a severe spread event. Smaller spread events are generally associated with periods of moderate weather and fire perimeters can be strongly influenced by suppression activities and burnout operations. The resulting fire size distribution from the 5 large fires was compared to the size distribution from Randig and found similar (cf. Figs. 3 and 4) although a higher proportion (0.15 versus 0.3) of smaller spread events were observed in the latter. It should be noted that there are many uncertainties with the observed fire spread data and the sample fires represent a range of fuels and topographies, some quite different than in the study area. We concluded that the initial burn period distribution initially developed in Randig based on local knowledge was representative of likely wildfire events under the conditions modeled, and further refinements were not warranted given the small sample size and data limitations noted above.

Outputs from the wildfire simulations included the burn probability (BP) for each pixel:

$$\mathrm{BP} = \frac{F}{n} \tag{1}$$

where F is the number of times a pixel burns and n is the number of simulated fires (10,000). The burn probability for a given pixel is an estimate of the likelihood that a pixel will burn given a single random ignition within the study area and burn conditions as represented in the simulation. Randig also generates a vector of marginal burn probabilities (BP_i) for each pixel which estimate the probability of a fire at the ith 0.5 m flame length category. Different flame lengths are predicted by the MTT fire spread algorithm depending on the direction the fire encounters a pixel relative to the major direction of spread (i.e. heading, flanking, or backing fire; Finney, 2002). The BP_i outputs were used to calculate the conditional flame length (CFL):

$$\mathrm{CFL} = \sum_{i=1}^{20} \left(\frac{\mathrm{BP}_i}{\mathrm{BP}}\right)(F_i) \tag{2}$$

where F_i is the flame length midpoint of the ith category and BP is the burn probability. Conditional flame length is the probability weighted flame length given a fire occurs and is a measure of wildfire hazard (Scott, 2006). In other terms, CFL is the average flame length among the simulated fires that burned a given pixel.

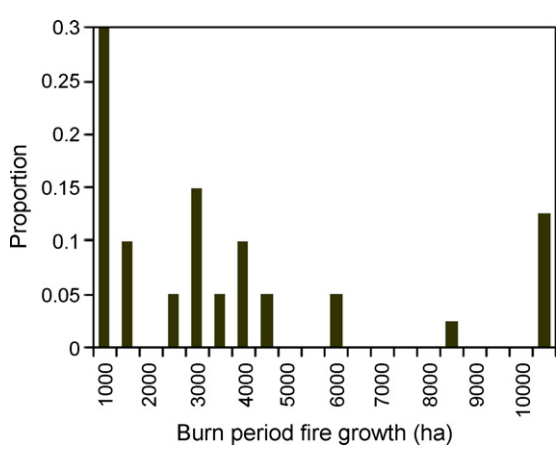

Fig. 4. Frequency distribution of burn period area obtained from daily fire progression data on 6 large wildfires in the Blue Mountains in 2003, 2005, and 2006. Fires included School, Columbia, Jim Creek, Burnt Cabin, Mule and Lightning. Only days with spread events greater than 1000 ha were included to reduce errors from backburns and coarse mapping of fire perimeters.

2.5. Estimating the probability of large tree loss

The probability of large tree mortality from simulated wildfires was determined by processing stand inventory data through FVS at a range of fire intensities to develop species- and size-specific loss functions. Each stand in the study area was burned within FVS-FFE under a pre-defined surface fire flame length ranging from 0.5 to 15 m in 0.5 m increments (SIMFIRE and FLAMEADJ keywords). FVS-FFE incorporates several fire behavior models as described in Andrews (1986), Van Wagner (1977), and Scott and Reinhardt (2001) to predict rate of spread, intensity, and crown fire initiation. Tree mortality from fire is predicted according to the methods implemented in FOFEM (Reinhardt et al., 1997). The post-wildfire stand tree list was then examined to determine the mortality of large trees by species at each flame length category. We note that the current configurations of FVS and Randig do not allow for exact matching of fire behaviors. Specifically, FVS simulated surface fires, and Randig reported total flame length of the combined crown and surface fire. For stands where a crown fire would initiate at lower flame length values than required for tree mortality, we have underestimated mortality. Work is needed to build a better linkage between FVS and wildfire spread models so this approach can be improved and applied to other stand metrics of interest.

Wildfire risk to large trees for each scenario was quantified as the expected loss (Finney, 2005) considering the probability of different flame lengths and the mortality at each flame length. Expected loss was calculated as:

$$\mathrm{E(L)} = \sum_{i=1}^{j} (\mathrm{BP}_i)(L_i) \tag{3}$$

where BP_i is the probability of a fire and L_i is the mortality (trees/ha) of large trees at the ith flame length category. Expected loss of large trees was then summed for individual species and for all trees by flame length category for each treatment scenario.

We also calculated a conditional expected loss of large trees as:

$$\mathrm{CE(L)} = \sum_{i=1}^{j} \left(\frac{\mathrm{BP}_i}{\mathrm{BP}}\right)(L_i) \tag{4}$$

Conditional expected loss is the observed mortality given that the distribution of fires occurs (BP = 1).

2.6. Wildfire burn probability profiles for residential structures

Because structure ignition models have not been incorporated into landscape fire simulators (Finney and Cohen, 2003), we used graphical methods to describe the potential fire impacts on structures and the effect of treatments. The Oregon Department of Forestry has outlined clearance rules for all structures in the WUI based on Senate Bill 360 and the Oregon Forestland-Urban Interface Fire Protection Act of 1997 (ODF, 2006). We chose to

Table 3
Wildfire simulation outputs for the Mt. Emily study area for simulated fire size (ha). Each scenario represented a spatial treatment priority (SDEN, stand density index; RDEN, residential density) and a treatment area (0, 10, 20, 30, 40, and 66% of forested area treated). TRT-66 represents treating all eligible stands in the study area.

Scenario	Area treated (ha)	Wildfire size (ha)	
		Average	Maximum
TRT-0	0	1014	6065
TRT-66	8513	395	4041
SDEN-10	1563	812	5795
SDEN-20	3267	700	5577
SDEN-30	4896	617	5150
SDEN-40	6542	514	4832
RDEN-10	1640	846	5821
RDEN-20	3261	725	5590
RDEN-30	4895	572	5036
RDEN-40	6526	576	5041

use the largest fuel break mandated, 100 ft (30.5 m), for all of the structures in the Mt. Emily WUI. To quantify wildfire risk to residential structures, we calculated the average burn probability by flame length category for pixels within a 45.7 m radius of the individual structures. The 45.7 m radius represents an assumed 15.2 m radius for the structure itself, and a 30.5 m radius fuel break around each structure.

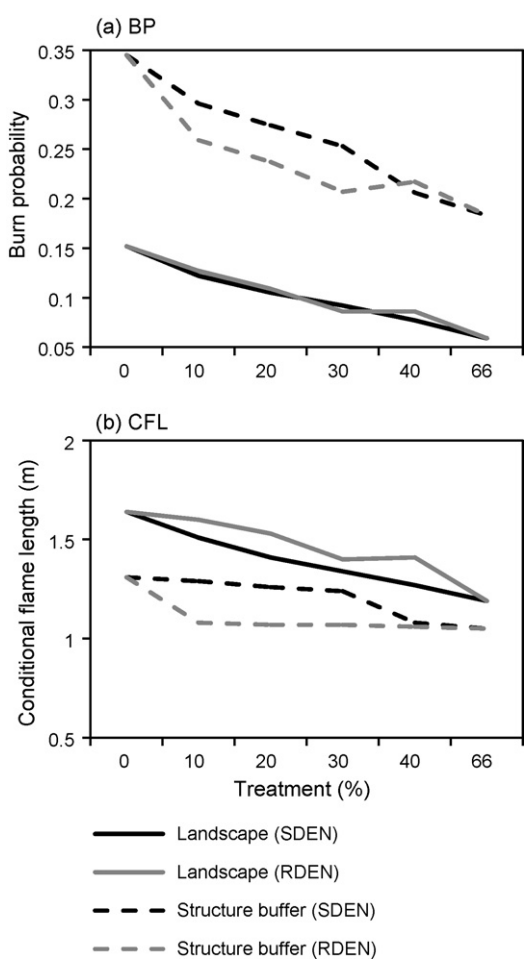

Fig. 5. Wildfire simulation outputs for the Mt. Emily study area for simulated burn probability and conditional flame length (m) for six treatment intensities and two spatial priorities. The lowest (0%) and highest (66%) treatment rates generated nearly identical results for the two spatial priorities and only one of the simulation scenarios was retained. TRT-66 represents treating all eligible stands in the study area. (a) Burn probability and (b) conditional flame length reported for structures (dashed line) represent average values for all pixels within a 45.7 m circular buffer around each of the 170 residential structures (Figs. 2 and 6).

3. Results

3.1. Simulated wildfire size, burn probability, and conditional flame length

Increasing fuel treatment area decreased average wildfire size, BP, and conditional flame length (CFL) for both the SDEN and RDEN treatment priorities (Table 3 and Fig. 5). Average BP among the scenarios and treatment levels ranged from a low of 0.059 (TRT-66) to a high of 0.154 (TRT-0, Fig. 5). The maximum BP for individual pixels was observed for untreated scenarios SDEN-0 (0.462) and RDEN-0 (0.473). The highest BP values were located in the WUI on the eastern edge of the study area (Fig. 6). This result was caused in part by timber-grass fuel models with high spread rates at the lower elevations within the WUI. Treating all overstocked stands in the study area (TRT-66) reduced the average odds of a pixel burning from 1 in 65 (BP = 0.153) to about 1 in 168 (BP = 0.059) from a randomly located ignition in the study area. To compare the

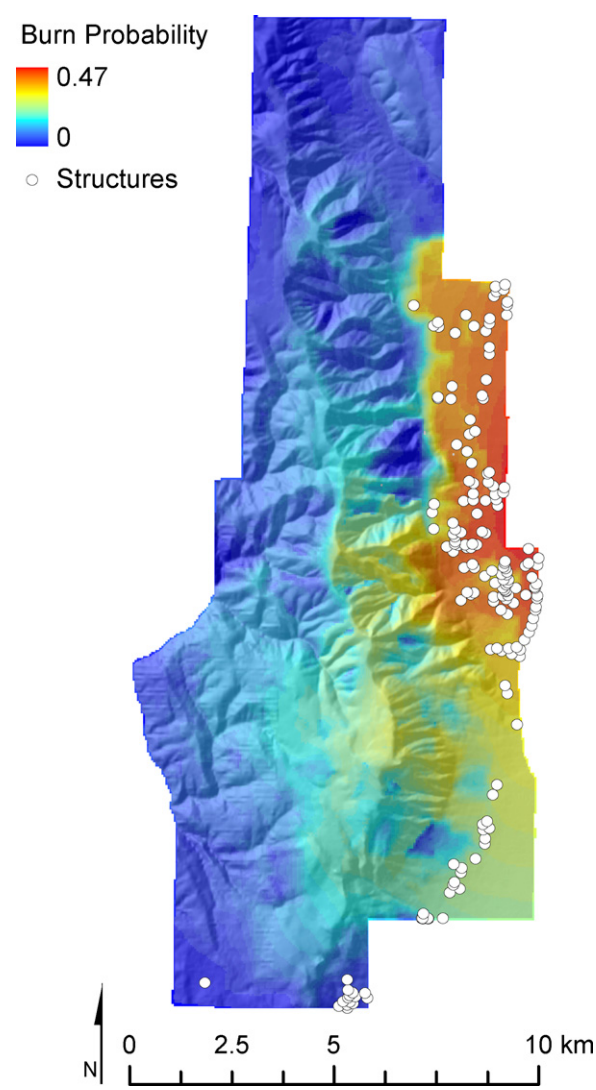

Fig. 6. Burn probability map for the Mt. Emily study area generated from simulating 10,000 wildfires. Simulation outputs are for the no treatment scenario. Burn probability is the likelihood of a point burning given a random ignition in the study area and specific burn conditions as described in Section 2.

Table 4
Ratio of average and maximum burn probabilities between treated scenarios compared to untreated (TRT-0). The ratio indicates the relative likelihood of a fire between no treatment and treatment scenarios. Landscape heading indicates average and max values for the entire study area. Structure buffer columns indicate values for 45.7 m buffer around the 170 residential structures. TRT-66 represents treating all eligible stands in the study area.

Scenario	Landscape		Structure buffer	
	Average BP	Maximum BP	Average BP	Maximum BP
TRT-0	1.0	1.0	1.0	1.0
TRT-66	2.6	1.7	1.9	1.7
SDEN-10	1.3	1.2	1.2	1.2
SDEN-20	1.4	1.3	1.3	1.3
SDEN-30	1.6	1.4	1.4	1.4
SDEN-40	2.0	1.5	1.7	1.5
RDEN-10	1.2	1.2	1.4	1.2
RDEN-20	1.4	1.3	1.5	1.3
RDEN-30	1.8	1.5	1.7	1.5
RDEN-40	1.8	1.4	1.6	1.4

relative wildfire likelihood among scenarios (i.e. relative risk, Zhang and Yu, 1998), we calculated the ratio of the treated to untreated burn probability for both the average and maximum BP values (Table 4). The ratios indicated that on average a pixel was 2.6 times as likely to burn in the untreated landscape compared to the TRT-66 scenario. Pixels within the structure buffers were 1.9 times more likely to burn in an untreated landscape compared to the TRT-66 scenario.

Average and maximum fire size generally decreased with increasing treatment area for both the SDEN and RDEN scenarios (Table 3). For every 100 ha treated, the average and maximum wildfire size decreased 9 ha and 18 ha, respectively. On a proportional basis, treating 20% of the landscape reduced average (and maximum) wildfire size by 31% and 29% (8% and 7%) for SDEN and RDEN, respectively. The SDEN and RDEN treatment priorities resulted in different spatial patterns of fire size and BP (Fig. 7).

Simulation outputs for conditional flame length (CFL) showed relatively high values in the south-central portion of the study area, primarily in the overstocked mixed conifer stands at the higher elevations west of the WUI. Average CFL ranged between 1.64 and 1.19 m for the no treatment and highest treatment area (Fig. 5). Maximum CFL was observed for the TRT-0 (9.71 m) scenario.

3.2. Expected loss of large trees

A total of 143,161 large trees existed in the inventory data for the study area. The SDEN scenario generally targeted stands containing the largest population of large trees since these stands had the highest SDI (Table 5 and Fig. 7a and d). For instance, the stands identified for treatment under the 10% treatment level in the SDEN scenario contained about 10 times the number of large trees compared to RDEN (Table 5). Large trees were concentrated in the south-central portion of the project area about 5–10 km southeast of the main concentration of residential structures, although large trees were observed throughout the project area.

Table 5
Number of large trees (>53.3 cm) inside of treatment units for the stand density (SDEN) and residential density (RDEN) spatial treatment priorities. The upper diameter limit on the thinning from below prescription excluded harvesting large trees as part of simulated treatments. TRT-66 represents treating all eligible stands in the study area.

Treatment priority	Treatment area (% of forested landscape)	Number of large trees
SDEN	10	67,573
	20	81,909
	30	94,917
	40	112,558
RDEN	10	6,926
	20	19,454
	30	63,599
	40	87,899
TRT-66	66	123,507

Mortality functions for large trees generated from FVS simulations were non-linear with increasing flame length and showed interspecific differences (Fig. 8) consistent with fire effects literature (Ryan and Reinhardt, 1988; Miller, 2000). For instance, mortality for subalpine fir and Engelmann spruce at a surface fire flame length of 0.75 m was 3.5 and 4.8 times the mortality of ponderosa pine. Mortality functions for treated stands showed lower mortality at all flame lengths when compared to non-treated stands (Fig. 8). Treated stands also showed a maximum mortality of 37–91% depending on the species (78% for all trees) versus 100% for the non-treated stands.

The expected loss of large trees (Eq. (3)) decreased with increasing treatment area, although the SDEN treatment priority was more effective than RDEN at reducing simulated mortality (cf. Fig. 9a and b). For example, treating 10% of the landscape reduced the proportion of total expected loss by 73% for the SDEN priority, compared to 5% for RDEN. Treating all overstocked stands in the study area reduced the proportion of total expected loss of large trees by 94% for both scenarios. These mortality differences between the scenarios were expected since there are fewer large trees in the WUI compared to the larger study area. Large interspecific differences in expected loss were observed between the treatment priorities (Fig. 9). For example, at the 20% treatment area, the expected loss for western larch in the SDEN scenario was reduced by 88% compared to the no treatment, versus a mere 8% reduction in the RDEN scenario. Similarly, the expected loss for Douglas-fir was reduced by 74% versus 54% for SDEN and RDEN scenarios at the 20% treatment area, respectively. Comparing the expected loss (Fig. 9a and b) with the conditional expected loss (Fig. 9c and d) shows the relative contribution of BP versus CFL in the mortality of trees. Conditional expected loss measures the mortality given that a fire occurs (BP = 1, Eq. (4)), and exhibited similar trends with increasing treatment area compared to expected loss (Fig. 9c and d). However, the conditional expected loss of large trees was roughly 10–20% higher for all tree species for the treatment scenarios. Thus, if stands were burned at the flame lengths generated by the simulated fires, the mortality would be 15–30% higher compared to the expected mortality from a single random ignition within the study area. This difference represents the effect of treatments on burn probability, i.e. the reduced likelihood of fire encountering large trees.

The effect of treatments on expected loss of large trees both inside and outside the treatment units was examined to better understand landscape scale effects of the treatment scenarios (Fig. 10). We first quantified the expected loss of large trees inside the polygons that were selected as treatment units. Expected loss inside these polygons before treatment ranged from about 10,000 to 13,000 trees among the five treatment levels (Fig. 10a, solid bars, SDEN scenario). The simulated fires resulted in only minor mortality of large trees inside the treatment units after treatments were implemented (Fig. 10a, open bars). For instance, comparing the expected loss within treatment units for the SDEN-20 scenario (20% treatment area) versus no treatment, the treatment prescription reduced expected loss by about 97% (Fig. 10a, SDEN-20, solid versus open bars). A similar reduction in the expected loss of large trees was observed within the treatment units selected in the other SDEN scenarios. The reduction in expected loss within treatment units was slightly more variable for the RDEN scenarios, ranging from 75% to 99% (Fig. 10a). In contrast,

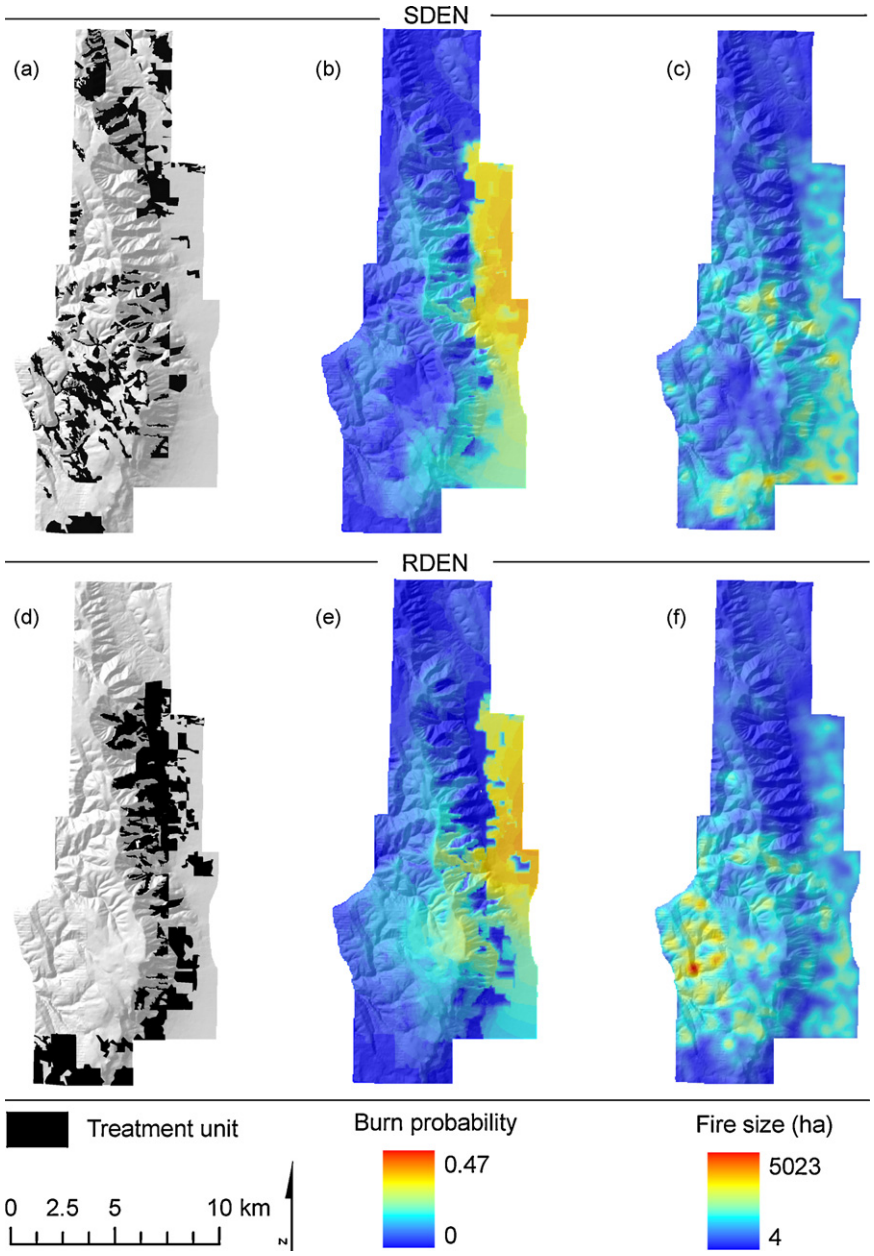

Fig. 7. Maps of treatment units (a and d), burn probabilities (b and e), and kernel smoothed fire size (c and f) for the 20% treatment area for the stand density (SDEN) and the residential density (RDEN) treatment priorities.

the reduction in expected loss of large trees outside the treatment units when treatments were implemented was markedly less, but nevertheless substantial (Fig. 10b, black versus open bars). Outside the treatment units (Fig. 10b), expected loss of large trees was reduced between 30% and 75% for the SDEN scenario, and 2–75% for RDEN.

3.3. Burn probability profiles for residential structures

Average and maximum BP for all pixels within the 45.7 m circular buffer around each residential structure decreased with increasing treatment area, except for a slight increase for RDEN-40 (Fig. 5). In general the RDEN treatment priority reduced BP within the buffers more than SDEN for a given treatment area. At the 10% treatment area, RDEN reduced BP at almost twice the rate of SDEN (27% and 14%, respectively; Fig. 5).

Burn probability profiles for the structure buffers (Fig. 11) showed that treatments reduced BP over all flame length categories, although the reduction was larger for the higher flame length values. The BP profiles also showed that the RDEN treatment scenario was more effective at reducing BP (Fig. 11). Considerable variation in BP and conditional flame length was observed for individual residential structures (Fig. 12). For instance, average BP for individual structure buffers varied from 0.0187 to 0.4726 for the no treatment scenario, while conditional flame length varied between 0.50 and 3.49 m (Fig. 12). These outputs suggested that the odds vary from 1 in 53 to about 1 in 2 in the likelihood that a random ignition in the study area will encounter an individual structure. The effect of the treatments can be observed in the shifting of the BP for individual structures to the left (lower BP) with increasing treatment area (Fig. 12). The difference between the RDEN and SDEN treatment strategies is also evident in the plots

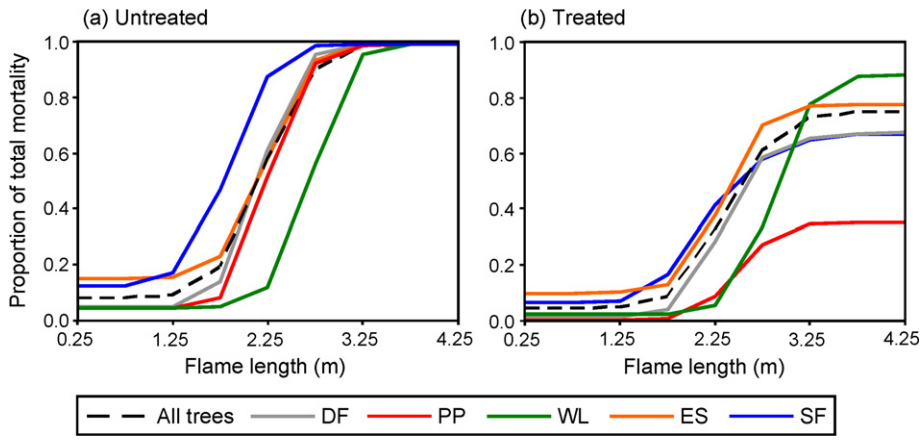

Fig. 8. Comparison of (a) untreated and (b) treated modeled mortality of large trees by flame length for all trees and select species for treated stands within the maximum treatment area (66% of forested lands). DF: Douglas-fir; PP: ponderosa pine; WL: western larch; ES: Engelmann spruce; SF: subalpine fir.

as a larger shift to the left (lower BP) and to a lesser extent down (lower conditional flame length) for the former compared to the latter. The relatively small reduction in conditional flame length compared to burn probability resulted of few treatments being placed in actual structure buffers. Stands containing structure buffers infrequently met the criteria to receive a thinning treatment due to past management on the private lands in the WUI.

4. Discussion

We employed complex simulation models to understand interactions among fuel treatment designs and wildfire risk as quantified by large tree mortality and burn probability profiles for residential structures. Many sources of error in the models and data are possible, and the results should be viewed in general terms. The Mt. Emily study area represents a specific spatial configuration of fuels, potential weather, residential structures, and large trees, making the results specific to a WUI with a similar configuration. Additional case studies are needed on other landscapes containing varying spatial arrangements of wildlands and urban development to better understand topological relationships between fuel treatment strategies and fire risk to human and ecological values. The results suggest that fuel treatments well outside of WUIs can significantly reduce wildfire threats to property values, a finding that helps inform the debate over the effectiveness of Federal fuel treatment programs on wildlands proximal to urban development (Schoennagel et al., 2009). While there are tradeoffs between managing landscapes to address long-term restoration goals versus protecting residential structures (e.g. Fig. 7), both objectives can be addressed with spatial treatment designs that factor likely wildfire spread directions and the juxtaposition of values at risk. For instance, the SDEN scenario, which selected stands based on level of overstocking (SDI),

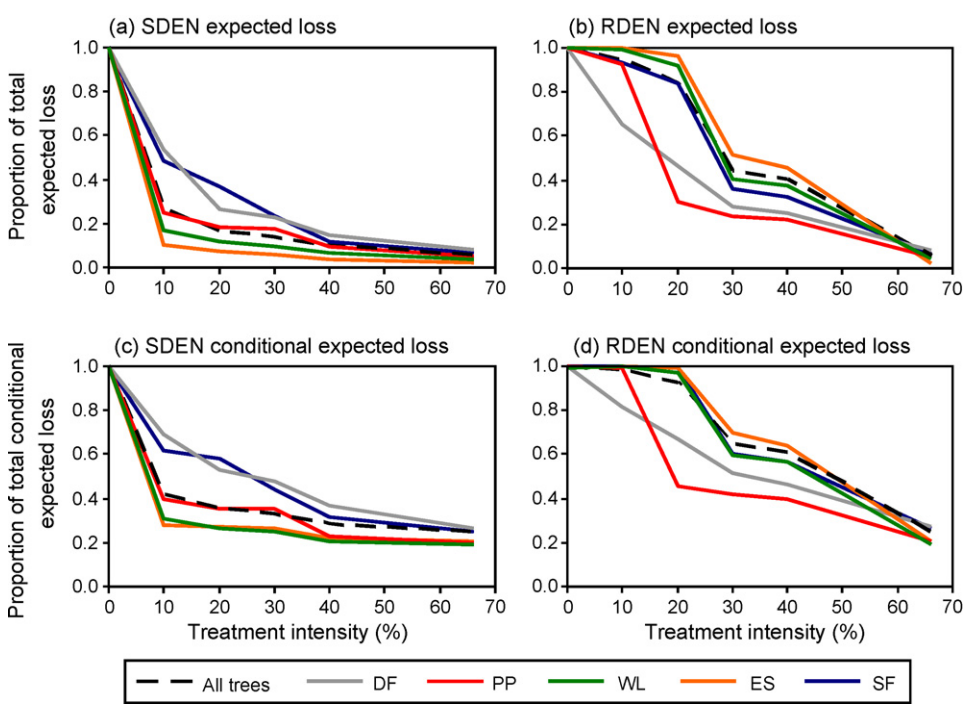

Fig. 9. Proportion of total expected loss (a and b) and total conditional expected loss (c and d) for the stand density (SDEN) and the residential density (RDEN) priorities and six treatment intensities for all trees and select species. DF: Douglas-fir; PP: ponderosa pine; WL: western larch; ES: Engelmann spruce; SF: subalpine fir. Expected loss and conditional expected loss is shown in Eqs. (3) and (4), respectively.

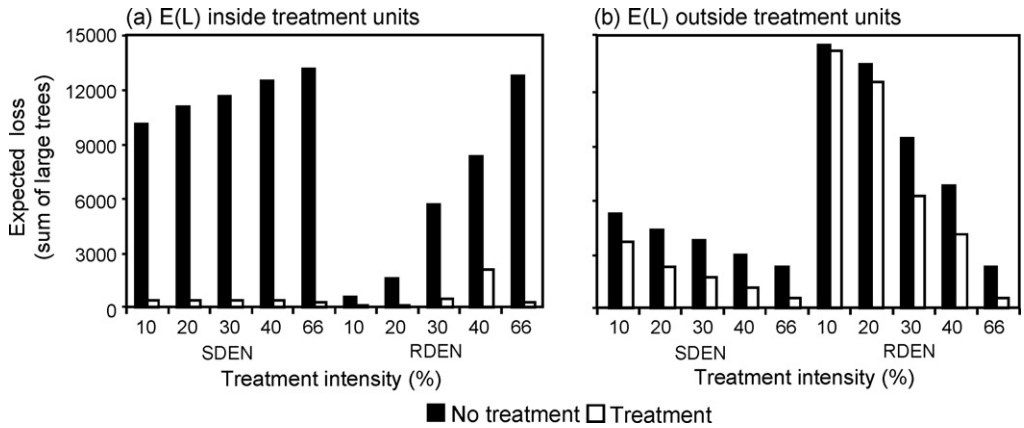

Fig. 10. Comparison of treatment affect on expected loss (Eq. (3)) of large trees (a) inside and (b) outside of treatment units for both the stand density (SDEN) and residential density (RDEN) treatment scenarios for six treatment intensities.

dramatically reduced the expected loss of large trees from a randomly located ignition and subsequent severe fire event in the study area. More importantly, the simulations also predicted a marked reduction in wildfire likelihood and, to a lesser extent, intensity to the buffer around residential structures as quantified in burn probability profiles (Fig. 11). However, when fuel treatments were prioritized based on residential structure density (RDEN), treatments were more effective at reducing BP and intensities in the structure buffers, but were less effective at protecting large trees. The latter result can be attributed to higher BP and CFL in untreated stands located in the wildland portion of the study where the bulk of the large trees are found. Nevertheless, the experiment shows that focusing treatment in and around WUIs with constrained fuel treatment budgets could prevent significant restoration and forest health activities in surrounding wildlands, indirectly contributing to future fuels build up and larger fires that may overwhelm the localized protection offered by WUI treatments.

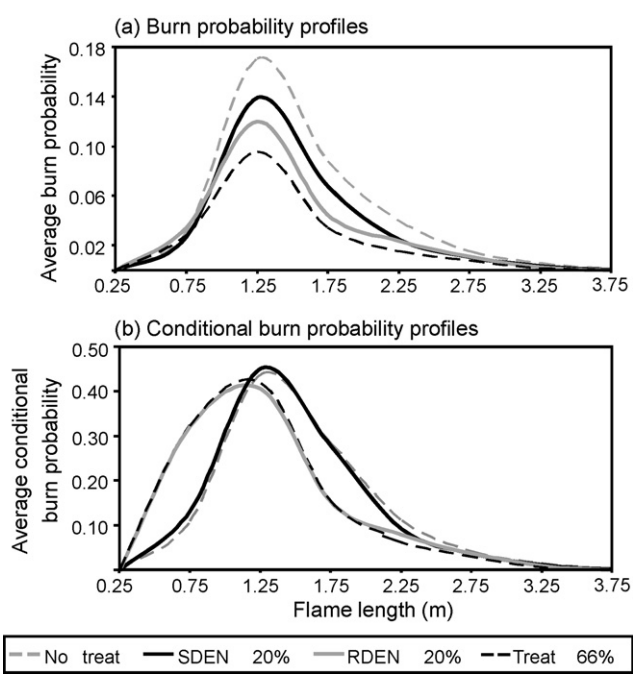

Fig. 11. Burn probability profiles (average burn probability by flame length category) for the pixels within a 45.7 m radius around each structure for three treatment intensities: no treatment, 20% area treated, and 66% area treated for the stand density (SDEN) and residential density (RDEN) scenarios.

As in previous simulation studies (Ager et al., 2007a; Finney et al., 2007) and field experiments (Gil Dustin, Bureau of Land Management, personal communication) a non-linear response to area treated was observed for one or more of the fire modeling outputs, including spread rate, burn probability and fire size. A steep non-linear change was also observed here for the expected loss of large trees (Fig. 9) for one (SDEN) of the two scenarios. The SDEN scenario was more effective at blocking and slowing the spread of large fires at the lower treatment intensities. One reason is that these stands tended to be located in the middle of the study area in the dominant fire travel routes through the landscape. Another reason is that these stands, in general, contained the highest surface and canopy fuel loadings (fuel model 10) in the project area, and the treatments resulted in a dramatic reduction in spread rates. The findings underscore the importance of strategic placement of fuel treatments, and also add to the growing body of evidence that there are diminishing returns with investments in fuel treatments after 10–20% of landscapes are treated (Ager et al., 2007a; Finney et al., 2007).

One exception to the overall result of decreased BP, CFL, fire size, and loss of large trees with increasing treatment area was the trend between the RDEN-30 and RDEN-40 scenarios (Figs. 5 and 10; Table 3). It is not clear why these response variables did not follow the overall pattern, and errors in processing the outputs have been ruled out. Two explanations are: (1) sampling error caused by too few ignitions and (2) the MTT algorithm essentially found a faster path through the landscape after increasing the treatment area. We favor the former explanation but cannot eliminate the latter as a possibility. We have noted significant variation in duplicate simulations when relatively small numbers of simulations are used, but that variation was not observed here. Additional application of these models will help us understand the factors that affect burn probability (Parisien et al., 2010) and the performance of the MTT algorithm.

Our management prescriptions and priorities were patterned after operational programs in the Blue Mountains that are designed to reduce wildfire impacts, facilitate wildfire control near key values and, in the long run, allow natural ignitions to generate beneficial fires. The SDEN restoration scenario resulted in a significant reduction in wildfire intensity, as measured by the expected loss of large trees, thus reducing the potential for future adverse wildfire impacts and perhaps allowing for natural ignitions to play an increased role in future fuel management in the area. The prescriptions were also intended to reduce the impact of density-dependent forest insect mortality, which has been a major contributor to the current surface fuel build up in the study area and much of the Blue Mountains province (Quigley et al.,

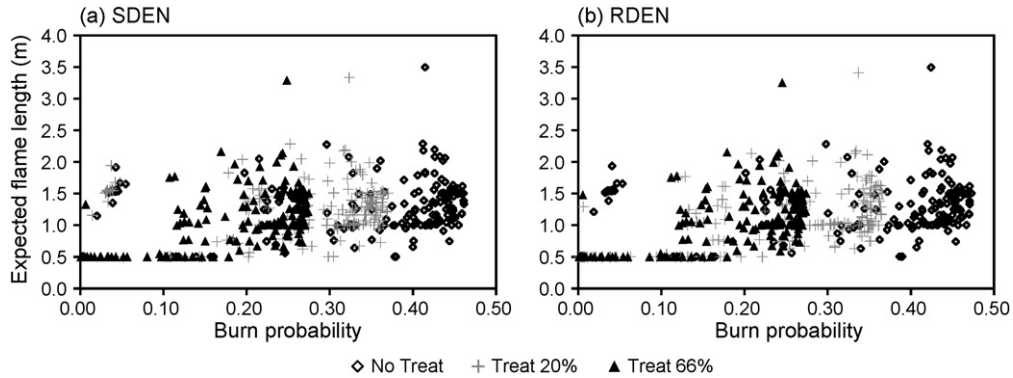

Fig. 12. Conditional flame length and burn probability scatter plots for showing values for individual structures. The stand density (SDEN) and residential density (RDEN) priorities for the no treatment, 20% area treated, and 66% area treated are plotted. Points are averages of all pixels within a 45.7 m radius around each structure.

2001). More detailed modeling of stand prescriptions would allocate thinning, mechanical fuels removal, and underburning in different combinations based on existing stand conditions and potential vegetation types (Schmidt et al., 2008). For instance, additional treatments in the residential structure buffers might have reduced conditional flame lengths if we simulated surface fuels removal in stands that did not qualify for thinning based on stocking (SDI) guidelines. However, the inventory data lacked the detail required for more specific prescription decisions.

The modeling assumes that wildfires burning under severe conditions (97th percentile August weather) will have lower spread rates and intensities within the fuel treatment units and outside to the leeward (Finney, 2001) and thus, given finite periods of extreme burn events, average fire sizes will be reduced. There is ample empirical evidence that fuel treatments including wildland fire use activities can reduce spread rates and intensity (Finney et al., 2005; Collins et al., 2007, 2009; Ritchie et al., 2007; Safford et al., 2009), although there are instances where landscape treatment area and intensity were insufficient to significantly alter extreme fire events (Agee and Skinner, 2005; Graham, 2003).

We recognize that burn probability profiles (Fig. 11a and b) are only a general indicator of wildfire risk to structures and that structure ignition is a complex process dependent on surrounding forest fuels, flammability of the structure, and vegetation in the home ignition zone (Cohen, 2008). More specifically, the probability of structure loss is dependent on the interaction of fire behavior (flame radiation, flame impingement (convection), and burning embers, Cohen and Butler, 1996), structure characteristics (ignitability of the materials, Cohen, 1991; Cohen and Savelend, 1997), and suppression actions. It would be exceedingly difficult to acquire the data and create structure-specific loss functions for landscape wildfire studies to interpret pixel-based burn probabilities in terms of structure loss (Massada et al., 2009). While home ignitability rather than fire behavior is the principal cause of home losses during wildland urban interface fires (Cohen, 2000, 2008; Finney and Cohen, 2003), it is important to note that fuel treatments can moderate fire in the vicinity of structures allowing suppression crews to engage in direct structure protection (Safford et al., 2009). Given these considerations, BP and conditional flame length plots for pixels within the structure buffer are useful for identifying relative wildfire risk among structures, prioritizing treatments, and quantifying potential treatment effects. In a recent study on wildfire risk in a Wisconsin WUI, Massada et al. (2009) calculated burn probabilities using the MTT algorithm in FlamMap and assumed WUI structures burned at all flame lengths. The burn probability profiles used here allows the assessment of both the probability and intensity components of risk.

Burn probability modeling offers more robust measures of wildfire likelihood compared to methods employed previously where fire likelihood was quantified with relatively few (<10) predetermined ignition locations (Stratton, 2004; Roloff et al., 2005; LaCroix et al., 2006; Loureiro et al., 2006; Ryu et al., 2007; Schmidt et al., 2008). The development of the MTT algorithm in Randig and implementation in FlamMap and other wildfire simulation systems (Andrews et al., 2007) makes it feasible to rapidly generate BP surfaces for large landscapes (Massada et al., 2009) and for different management scenarios, eliminating potential bias from assuming specific ignition locations. As discussed previously (Ager et al., 2007a), it is important to note the difference between BP as estimated here versus probabilistic models built with historical fire occurrence and size data (Martell et al., 1989; Preisler et al., 2004, 2009; Mercer and Prestemon, 2005; Brillinger et al., 2006). The BP outputs measure the probability of a pixel burning given a single random ignition in the study area with weather conditions as simulated. In the case of a fuel treatment project like Mt. Emily (Wallace, 2003), qualitative assessments of future wildfire risk are rendered by local fire managers when project areas are chosen, and BP is used to measure relative risk among treatment alternatives. Newer models that use the MTT algorithm include spatio-temporal probabilities for ignition, escape, and burn conditions, and yield estimates of annual burn probabilities (Finney, 2007, see also Miller, 2003; Davis and Miller, 2004; Parisien et al., 2005). Burn probability modeling is now being applied across the US as part of wildfire risk monitoring (Calkin et al., in press), strategic budgeting efforts in the Fire Program Analysis project (http://www.fpa.nifc.gov/), and wildland fire decision support systems (http://wfdss.usgs.gov/wfdss/WFDSS_Home.shtml), collectively mainstreaming BP modeling via the MTT algorithm for a wide range of wildfire threat problems. Studies are underway to better understand the topology and control of BP on large forested landscapes (e.g. Parisien et al., 2010; Ager and Finney, 2009).

Burn probability modeling coupled with loss or benefit functions (Finney, 2005) provides a risk-based framework to quantify wildfire impacts that are highly uncertain in space and time. Risk analysis as implemented here incorporated important interactions among wildfire likelihood, intensity (flame length), and effects (large tree mortality). Moreover, the approach makes it possible to measure the effects of fuel treatments on fire behavior and resource values both within and outside the treatment units. Few studies have attempted to quantify off-site fuel treatment effects (Finney et al., 2005) although their importance is often discussed (Reinhardt et al., 2008), especially in the context of WUI protection (Safford et al., 2009; Schoennagel et al., 2009). In our study, the results suggested substantial reduction in risk to highly valued large trees and structures well outside of treatment areas (5–10 km) and thus the restoration strategy in the wildlands reduced wildfire probability and intensity to structures in the WUI,

even at the low range of area treated. These landscape scale effects could be leveraged in landscape fuel treatment designs to maximize benefits from fuel treatment programs.

The advantages of risk analysis for wildfire threat assessment and mitigation were recognized in reports by oversight agencies like the Government Accountability Office (GAO) report (2004) that stated "Without a risk-based approach at the project level, the [United States Forest Service and Bureau of Land Management] cannot make fully informed decisions about which effects and projects alternatives are more desirable." Our methods also address concerns in the Office of Inspector General (OIG) report (2004) that states "Our audit found that FS lacks a consistent analytical process for assessing the level of risk that communities face from wildland fire and determining if a hazardous fuels project is cost beneficial." Burn probability modeling and risk analyses appear to hold the most promise to answer a range of management questions that continue to be debated in the literature, including analyzing potential carbon offsets from fuel treatments (Hurteau et al., 2008; Mitchell et al., 2009; Ager et al., in review), tradeoff between short-term resource impacts of fuel treatments versus long-term benefits of wildfire mitigation (O'Laughlin, 2005; Irwin and Wigley, 2005; Finney et al., 2006); cost-benefit analysis of fuel treatment programs in terms of avoided suppression costs (Calkin and Hyde, 2004), and wildfire impacts to critical habitat and conservation reserves (Agee, 2002a; Ager et al., 2007a; Hummel and Calkin, 2005). In a broader context, risk-based ecological metrics should be incorporated into forest ecosystem monitoring frameworks (Tierney et al., 2009) to capture uncertain wildfire impacts on ecological structure and function.

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