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Modeling hazardous fire potential within a completed fuel treatment network in the northern Sierra Nevada

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ABSTRACT

We built on previous work by performing a more in-depth examination of a completed landscape fuel treatment network. Our specific objectives were: (1) model hazardous fire potential with and without the treatment network, (2) project hazardous fire potential over several decades to assess fuel treatment network longevity, and (3) assess fuel treatment effectiveness and longevity over a range of two critical fire modeling inputs: surface fuel models and canopy base height. Modeling results demonstrate reductions in the hazardous fire potential across much of the treated landscape, relative to the untreated condition. These reductions persist throughout our modeling duration, 2010–2050. However, there was a strong effect of varying ingrowth levels, which were manipulated to generate different estimates of canopy base height over time, on hazardous fire potential over time. Under the low ingrowth level, which resulted in the highest predictions of canopy base height, hazardous fire potential steadily declined over time for the untreated landscape condition. The effect of varying fuel models in treated areas had much less impact on hazardous fire potential, indicating a robust treatment effect. Our results demonstrate a coordinated fuel treatment network that incorporates local knowledge of fire weather and likely fire behavior patterns can have a substantial impact on reducing hazardous fire potential. However, even with planned maintenance of the treatment network, hazard grows in untreated areas over time, resulting in an increase in overall fire hazard. This suggests additional treatments, including fire use, would be necessary to maintain low hazardous fire potential.

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1. Introduction

Fire exclusion policies coupled with past management practices have rendered many western US forests susceptible to uncharacteristically large and intense wildfires (Hessburg et al., 2005; Stephens et al., 2009). Large-scale fuel reduction efforts are needed to mitigate the potential for extensive losses of many productive and mature forests from such wildfires. However, there are a number of operational constraints (e.g., access, limited operating periods, funding) and land designations (e.g., protected habitat, riparian buffers, wilderness) that limit the extent that fuel reduction treatments can be applied across many landscapes (Collins et al., 2010). As a result, forest managers are tasked with devising strategic and innovative treatments across a landscape to achieve multiple objectives, including mitigating the potential for extensive fire-caused tree mortality throughout the area. Simulation studies have demonstrated reductions in both potential fire behavior (Finney et al., 2007; Collins et al., 2011) and expected tree mortality (Ager et al., 2007, 2010, 2013; Syphard et al., 2011) with as little as 10–25% of the landscape treated by employing strategic placement. While there have been significant advancements in both the tools and approaches available to guide effective landscape fuel treatment design (Ager et al., 2006, 2012; Finney, 2007) few examples of completed landscape fuel treatment networks exist “on the ground” (but see Moghaddas et al., 2010). This is likely due to the complexities associated with planning across large landscapes, as well as acquiring necessary data and executing models (Collins et al., 2010).

Fuel treatments have a finite duration of effectiveness, or longevity. The longevity is a function of several factors: species composition/fuel structure, site productivity, topography, treatment

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type, treatment intensity, and maintenance of treated areas (Chiono et al., 2012). Few studies have explicitly examined fuel treatment longevity, either at the stand-scale or landscape-scale. Empirical studies from wildfires (Finney et al., 2005; Collins et al., 2009; Martinson and Omi, 2013) and studies based on modeled fire (Finney et al., 2007; Collins et al., 2011; Chiono et al., 2012; Stephens et al., 2012) suggest that treatments can be expected to reduce fire behavior and effects within individual stands and across landscapes for 10–20 years. It is unclear how maintenance of treated areas, on a 10- to 20-year schedule, within an existing landscape fuel treatment network would affect fire spread and intensity across a landscape over time.

One of the main limitations in evaluating the effectiveness of landscape fuel treatments is the reliance on simulated fire behavior. Recent studies have been critical of commonly used fire behavior modeling techniques (Cruz and Alexander, 2010; Alexander and Cruz, 2013). In particular, these and other studies (Fulé et al., 2001; Hall and Burke, 2006) have noted a general underprediction of crown fire. Characterization of surface and ladder fuels, represented as surface fuel models and canopy base height in commonly used modeling software, are the most influential inputs determining predicted fire behavior (Hall and Burke, 2006). With regard to modeling fire behavior across landscapes appropriately characterizing these two inputs is critical for obtaining realistic evaluations of fuel treatment effectiveness (Collins and Stephens, 2012). In addition to their importance in capturing static assessments of altered fuel conditions in treated areas (e.g., Moghaddas et al., 2010), surface fuel models and canopy base height are essential for dynamic characterizations of changing surface and ladder fuels over time as well (e.g., Seli et al., 2008; Collins et al., 2011). Despite the importance of these two input variables, little work has been done to analyze the sensitivity of landscape fire behavior predictions, thus assessments of landscape fuel treatment effectiveness, to changes in these two variables.

In this study we analyzed a completed landscape fuel treatment project in the northern Sierra Nevada. This consists of a coordinated network of treated areas that were intended to reduce fuel continuity across the landscape and provide defensible zones for fire suppression resources (Omi, 1996; Weatherspoon and Skinner, 1996; Agee et al., 2000). The network is a product of coordination among multiple fuel treatment projects implemented by the Mt. Hough Ranger District of the Plumas National Forest between 2003 and 2008 (USDA, 2004a). In addition to the fuel treatments, small openings (0.25–1.0 ha), based on group selection silviculture, were created throughout the study area to achieve an all-aged mosaic of forest stands across the landscape. Previous work by Moghaddas et al. (2010) investigated the effectiveness of this same fuel treatment and group selection network at reducing modeled landscape-level fire behavior and found that overall conditional burn probability was lower for the post-treatment landscape relative to pre-treatment. They also demonstrated that fire size under a modeled "problem fire" scenario was reduced by one-third for the post-treatment landscape. We built on this previous work by performing more in-depth analyses examining the impacts of the fuel treatment network in Meadow Valley. Our specific objectives were: (1) model hazardous fire potential across the landscape with and without the treatment network, (2) project hazardous fire potential over several decades to assess fuel treatment network longevity, and (3) assess fuel treatment effectiveness and longevity over a range of two critical fire modeling inputs: surface fuel models and canopy base height. The intent of the third objective was to capture a range of these two variables that bracketed the uncertainty in our ability to quantify surface and ladder fuels (Menning and Stephens, 2007; Cruz and Alexander, 2010; Keane, 2012).

2. Material and methods

2.1. Study area

Our study area is located in the Plumas National Forest and is situated in the northern Sierra Nevada at 39°56′N, 121°3′W (Fig. 1). The study area boundary is defined by three adjacent Hydrologic Unit Code sixth-level watersheds, with a slight modification to the southern-most watershed based on the extent of the Meadow Valley Project area (USDA, 2004a). The climate is Mediterranean with a predominance of winter precipitation totaling about 1050 mm per year (Ansley and Battles, 1998). The core study area is 19,236 ha, with elevations ranging from 850 to 2100 m (Fig. 1). Vegetation on this landscape is primarily mixed conifer forest (Schoenherr, 1992), consisting of white fir (*Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), sugar pine (*Pinus lambertiana*), ponderosa pine (*P. ponderosa*), Jeffrey pine (*P. jeffreyi*), incense-cedar (*Calocedrus decurrens*), California black oak (*Quercus kelloggii*), and other less common hardwood species. Red fir (*A. magnifica*) is common at higher elevations, where it mixes with white fir. In addition, a number of species are found occasionally in or on the edge of the mixed conifer forest includes western white pine (*P. monticola*) at higher elevations, lodgepole pine (*P. contorta*) in cold air pockets and riparian zones, western juniper (*Juniperus occidentalis*) on dry sites, California hazelnut (*Corylus cornuta*), dogwood (*Cornus spp.*) and willow (*Salix spp.*) in moister riparian sites. Montane chaparral and some meadows are interspersed in the landscape. Tree density varies as a result of recent fire and timber management history, elevation, slope, aspect, and edaphic conditions. Historical fire occurrence, inferred from fire scars recorded in tree rings, suggests a historical fire regime with predominantly frequent, low- to moderate-severity fires occurring at intervals ranging from 7 to 19 years (Moody et al., 2006).

The projects that contributed to the fuel treatment and group selection network in our study area are part of the larger Herger Feinstein Quincy Library Group Pilot Project (USDA, 1999). This project was directed by the US Congress to enable local community involvement in forest management. The project objectives included improving forest health, reducing uncharacteristic high severity fire, conserving wildlife habitats, and stabilizing economic conditions in the local community. Projects in Meadow Valley encompassed a range of treatment types and intensities reflecting changes in regional management directions (USDA, 2001, 2004b) and differing land management constraints across a complex landscape (Collins et al., 2010; Moghaddas et al., 2010). We classified treatments into five categories: (1) chainsaw-thinned & pile-burned: trees up to 30 cm dbh were cut and burned in piles; (2) masticated: primarily shrubs and some small trees were shredded and chipped in place with the material left on-site; (3) prescription-burned: stands were burned under moderate relative humidity and fuel moisture conditions, (4) mechanically thinned & prescription-burned: trees up to 51 cm or 76 cm dbh, depending on whether or not stands were considered in the wildland-urban interface, were thinned from below, using a whole tree harvest system, to a residual canopy cover of approximately 40%, and then underburned; and (5) group selection harvested: removal of all conifers up to 76 cm dbh, followed by slash removal, then either natural regeneration or re-planting to a density of 270 trees ha^{-1} with a mix of sugar pine, ponderosa pine, and Douglas-fir (USDA, 2004a). These treatments collectively covered 3692 ha, or 19% of our study area (Table 1) and were implemented between 2003 and 2008.

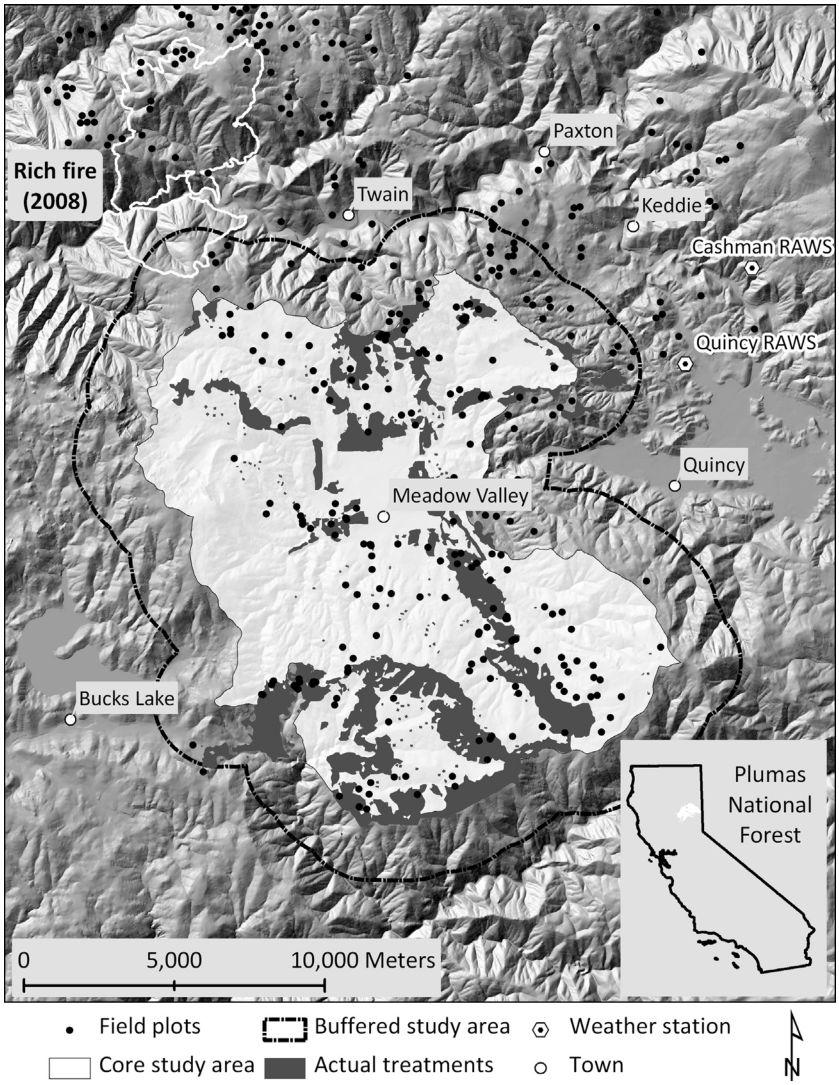

Fig. 1. Map of the Meadow Valley area, Plumas National Forest, California, USA. Areas treated as part of the landscape fuel treatment network implemented between 2003 and 2008 are also identified, along with the 2008 Rich fire, which was used for fuel model calibration.

2.2. Field data collection

We used data from two field sampling efforts. The first effort was aimed at a landscape-scale characterization using stratified-random approach to establish plots, referred to hereafter as landscape plots. This approach used four strata: slope (3 levels: <15%, 15–30%, >30%), elevation (3 levels: <1400 m, 1400–1600 m, and >1600 m), aspect (4 levels: N, E, S, and W), and dominant vegetation (6 classes based on air photo-interpreted California Wildlife Habitat Relationship vegetation classes) (VESTRA, 2003). Plot locations were randomly assigned within each strata combination, with emphases on covering the full extent of the study and greater sampling intensity in more frequently occurring vegetation types. Initial study design and plot sampling was based on a much larger study area, which included three "replicate" landscapes. Difficulties implementing fuel treatment projects led to only one of the three study landscapes actually receiving the full fuel treatment network. As such, the landscape plots extend well beyond our current study area boundary, particularly to the north (Fig. 1). In total there were 604 landscape plots that were sampled between 2004 and 2006.

The second field sampling effort was focused on treated areas, referred to hereafter as treatment plots. Plot locations were selected to capture the range in both treatment types and geographic locations of treatments throughout the study area. There were 72 treatment plots. These plots were initially sampled 1–3 years prior to treatments, and re-sampled 1 year following treatments. Pre-treatment sampling was conducted between 2002 and 2007, with post-treatment re-sampling between 2004 and 2009.

The two sampling efforts followed different plot sampling methodology based on the different objectives of the two efforts. Landscape plots were smaller and less intensive than the treatment plots to allow for a greater number of plots that covered a much larger spatial extent. Landscape plots were circular with a fixed radius of 12.6 m, resulting in a plot area of 0.05 ha. Trees >1.37 m tall and <10 cm diameter-at-breast-height (dbh) were tallied by species with no individual height measurements. Trees $\geqslant$10 cm dbh were binned into 10-cm dbh classes with their heights measured to the nearest 1 m for trees <10 m, while heights for trees $\geqslant$10 m were estimated to nearest 10-m category (e.g., 10–20 m, 20–30 m). Crown base height for all trees $\geqslant$10 cm dbh was measured to the nearest 1 m. Within each plot downed woody fuels, litter, duff, and shrubs were sampled on two 12 m transects, with random directions radiating from plot center. Downed woody fuels were sampled using the planar intercept method (Brown, 1974) with different lengths of the transect for different fuel particle size classes: 1-h (0–0.64 cm) and 10-h (0.65–2.54 cm) fuels were sampled from 10 to 12 m, 100-h (2.55–7.62 cm) fuels from 9 to 12 m, and 1000-h fuels from 2 to 12 m with diameters and sound/rotten recorded for each piece. Fuel height and litter and duff depths were recorded at 5 m and 12 m. Shrub cover was measured as intersections along each transect with species and average height recorded for each individual.

Treatment plots were rectangular: 50 m × 20 m, resulting in a plot area of 0.1 ha. Tree sampling was as follows: ⩾76.2 cm dbh sampled on the entire plot, 40.6–76.2 cm dbh on the center half of the plot, 12.7–40.5 cm dbh on the center quarter of the plot, 2.5–12.7 cm dbh on five 16 m^2 sub-plots established along the main plot centerline. Total height, crown base height, and dbh were measured on all trees. Shrub cover by species was recorded at each of the five 16 m^2 sub-plots. The original protocol for treatment plots used the Blonski and Schramel (1981) photo series to assess surface fuel loads pre- and post-treatment. Post-treatment surface fuel sampling was augmented to provide more detail and better capture variability among treatment types. In 2011 an additional 100 plots were added (41 in prescription burned, 25 in mechanically thinned & prescription burned, 17 in hand-thinned & pile-burned, and 17 in masticated stands) using the same surface fuel sampling protocol as the landscape plots. In each of these additional plots surface and ground fuels were measured on 50-m transects, with repeat measurements every 10 m. Group selection stands were not included in the augmented sampling because they covered only 1% of the study area (Table 1). The surface fuel information collected from these plots was used to determine surface fuel models for the different treatment types based on expert opinion from local fuels specialists (J. Moghaddas, R. Bauer, personal communication, Plumas National Forest).

2.3. Data integration

We used tree data from both field sampling efforts to generate tree lists to 'populate' distinct forested stands across the landscape. Stand boundaries corresponded with mapped vegetation polygons (VESTRA, 2003). There were 1505 forested stands that were either within or intersected our core study area boundary, ranging in size from 0.5 to 490 ha. Forested stands accounted for approximately 92% of the core study area. Non-forest vegetation, barren areas, and lakes accounted for the remaining 8% of the study area. Tree lists from plots were assigned to forested stands based on the same four strata that were used in sampling landscape plots: slope, elevation, aspect, and dominant vegetation type. The rules for tree list assignment were: if multiple plots matched the four strata for a given stand, a random plot was chosen; if no plots matched exactly, strata were eliminated in the following order until a match was made: slope, then aspect, then elevation.

Treatment polygons, compiled by the Herger-Feinstein Quincy Library Group monitoring team (http://www.fs.fed.us/r5/hfqlg/monitoring/), were overlaid on our stand/vegetation polygon layer. In cases where stands were partially overlapped by treatment polygons, stands were sub-divided such that stand boundaries corresponded with treatment boundaries. Treatment plots were assigned to stands within treatment polygons based on the five treatment types described in Table 1. When multiple plots existed for a given treatment type, a random plot was chosen for a given treated stand. In the specific cases where a treatment plot fell in a given treated stand that plot was assigned to the stand.

We created two databases with tree lists for every forested stand across our buffered study area (Fig. 1): (1) a pre-treatment database, in which stands that fell within treatment boundaries were 'populated' by tree lists from pre-treatment sampling efforts for the treatment plots and stands outside treatment polygons were 'populated' using the stratified approach described previously; and (2) a post-treatment database, in which tree lists for treated stands were generated from the post-treatment plot measurements of the same plots used in the pre-treatment database, while tree lists for stands outside of treatment polygons were the same as those in the pre-treatment database. The two tree list databases were entered into the Forest Vegetation Simulator (FVS) (Dixon, 2002) to simulate forest dynamics within the treated and untreated landscapes for four 10 year cycles beyond our starting year of 2010. Modeling was performed using the integrated platform ArcFuels (Ager et al., 2006; Vaillant et al., 2011), which executes FVS and the fire and fuels extension (FFE) to generate inputs needed to execute spatially explicit fire behavior models: canopy bulk density, canopy cover, canopy height, and canopy base height.

We used the western Sierra variant of FVS, which does not explicitly simulate establishment of new trees in the absence of disturbance, or ingrowth. To simulate ingrowth users must input the number, species, and frequency of establishment events. We used a random number generator to choose the actual number of seedlings, within species-specific bounds, that established for a given stand in a given FVS cycle. Additionally, we regulated seedling height growth to simulate more realistic conditions under an intact canopy. This approach of stochastically modeling ingrowth and then regulating height growth has been used in a previous simulation study within the Sierra Nevada (Collins et al., 2011) and is practiced by Forest Service silviculture personnel within the region (R. Tompkins, pers. comm., Plumas National Forest). In the absence of simulated ingrowth, stand canopy base heights increased

Table 1
Summary of treated area making up the fuel treatment and group selection network within the Meadow Valley study area, Plumas National Forest, CA, USA. Three different surface fuel models were used to capture the range of potential post-treatment surface fuel conditions and to assess the sensitivity of fire behavior output to residual surface fuel assumptions.

Treatment category	Treated area within study area (ha)	Proportion of treated area	Proportion of total study area	Post-treatment surface fuel model[a]		
				Low	Moderate	High
Mechanically thinned & prescription-burned	1588	0.43	0.08	181	183	201
Prescription-burned	1071	0.29	0.06	181	183	185
Hand-thinned & pile-burned	480	0.13	0.02	183	184	185
Masticated	309	0.08	0.02	201	142	202
Group selection harvested	240	0.07	0.01	184	201	202
Total	3688	1.0	0.19			

[a] Based on Scott and Burgan (2005) fuel models.

considerably over time in untreated stands, which occurred at a rate that is difficult to justify ecologically, especially given the large proportion of shade-tolerant species present in many stands (Collins et al., 2011). We incorporated three levels (low, moderate, high) of simulated ingrowth to assess its influence on predicted fire behavior over time (Table 2). The values used for the moderate ingrowth level were based on previous work in the Sierra Nevada (Collins et al., 2011), with the low and high levels being approximately one-third and two-times that, respectively (Table 2).

Rather than rely on FFE to model dead and downed fuels over time and choose surface fuel models needed for fire behavior modeling, we developed our own surface fuel model selection logic (Seli et al., 2008; Miller and Davis, 2009; Collins et al., 2011). This logic was different for treated and untreated stands. The logic for untreated stands followed the approach performed by Collins et al. (2011), in which fuel model assignment was based on a decision tree involving multiple forest stand structural metrics. These metrics and the break values associated with them were identified through regression tree analysis on three plot-derived fuel variables: (1) dead and downed fine surface fuel load (includes litter and 1-, 10-, 100-h fuels), (2) shrub cover, and (3) coarse fuel load (1000-h fuels). The proportion of variance explained by the three regression trees was low, ranging 0.13–0.15. However, for the purposes of binning stands into one of six Scott and Burgan (2005) surface fire behavior fuel models the low explanatory power was considered acceptable. Fig. 2 demonstrates the compiled logic that was performed for each stand, at each 10-yr time step. Fuel models for treated stands were based on separate logic involving treatment type, planned maintenance treatments, and time since treatment. As with simulated ingrowth, we chose three scenarios to represent surface fuel conditions in treated stands: low, moderate, and high (Table 1). Based on information collected in the augmented post-treatment surface fuel plots, fuel model assignments were as follows: stands that were prescription burned, either burned alone or thinned then burned, and stands that were hand-thinned/pile-burned were assigned timber-litter or light slash fuel models, which were fixed for the study duration based on the planned maintenance of these treated areas outlined by the forest managers; group selection stands were assumed to progress from timber litter or slash, to shrub, and then to timber-shrub; masticated stands progressed from slash or shrub, to timber-shrub, to timber litter (Table 1). Note that as of 2013 burns were not completed on all units that were mechanically thinned & prescription burned. As such, the range of fuel models for this treatment type captures recently burned and unburned surface fuel conditions (Table 1).

2.4. Surface fuel model comparison: FFE default vs. override

To compare fuel model selection for untreated stands we took advantage of an actual wildfire (Rich Fire) that occurred adjacent to the Meadow Valley study area (Fig. 1). We modeled the Rich Fire using the fire spread simulator FARSITE (Finney, 1998) with canopy fuel inputs derived from FVE-FFE and two different fuel model layers: one output from FFE and the other based on our logic informed by regression tree analysis, referred to subsequently as override. This fire occurred in 2008, 2–4 years following establishment of our landscape field plots. This allowed for unique comparison of modeled fire behavior under the two fuel model scenarios, both informed by pre-fire field data, to actual wildfire effects. Although there is not a robust way to equate fire behavior to actual fire effects, we chose to bin fire behavior into three classes based on flame length: 0–1.2, 1.3–2.4, and >2.4 m. The classes were based on fire behavior likely to result in torching and crowning (NWCG, 2004) and correspond with scorch heights of: 0–3.9, 4.0–15.7, and >15.7 m (based on an average wind speed of 10 km h^{-1}, Van Wagner, 1973). We compared the proportion of area in each flame length class to that in low, moderate, and high fire severity classes, based on the relativized differenced normalized burn ratio (RdNBR) (Miller and Thode, 2007). While this is an imperfect comparison we submit it is a reasonable approximation for the purposes of investigating general patterns between the two fuel model inputs. Weather inputs for FARSITE were obtained from the Cashman Remote Automated Weather Stations (RAWS) (Fig. 1), and were the actual observations during the Rich Fire.

2.5. Fire modeling

We obtained weather information from the Quincy and Cashman RAWS, restricting the analysis period to the dominant fire season for the area (June 1–September 30) (Fig. 1). Observations were available from 2002 to 2009. We used 90th percentile and above wind speeds, based on hourly observations, to generate multiple wind scenarios under which fires were simulated. We identified the dominant directions and average wind speeds of all observations at or above the 90th percentile value. This resulted in three different dominant wind directions, each with its own wind speed and relative frequency (based on the proportion of observations recorded at or above the 90th percentile value for each dominant direction) (Table 3). The modeled wind speeds were similar to those recorded during large spread events in the nearby 2008 Rich Fire. We used 97th percentile fuel moistures, as these are the conditions associated with large fire growth and difficulty in control (Table 3).

Topographic inputs including slope, aspect, and elevation were derived from 10-m digital elevation model obtained from the National Elevation Dataset (http://ned.usgs.gov/). Stand structure and fuels layers were derived from FVS outputs. For each stand, at each time step, FVS outputs for canopy cover, canopy bulk density, canopy base height, and dominant tree height, along with a fuel model assignment (computed outside of FVS), were compiled to develop continuous layers for each of these five variables across the buffered Meadow Valley study area using ArcFuels (Vaillant et al., 2011). The buffered area included a 2 km buffer around the core study area. We compiled these variables at five different time steps (2010, 2020, 2030, 2040, and 2050), for treated scenarios under three different treatment surface fuel model levels, and untreated scenarios under three different ingrowth levels, resulting in 30 different simulated landscapes. We did not do a full cross of the three ingrowth and three treatment fuel model levels due to long computational times associated with executing the fire behavior model (next paragraph) and processing output. To avoid potential edge effects, we extracted fire modeling output from only the core Meadow Valley study area, i.e., not including the area within the 2 km buffer (Fig. 1).

We employed a command-line version of FlamMap (Finney, 2006) called RANDIG to model fires across the Meadow Valley

Table 2
Three levels of simulated ingrowth for dominant conifer species in forested stands. The actual number of seedlings simulated in each 10-year cycle was stochastically assigned within the ranges reported for each stand. These three levels were applied to the 40-year simulations under the untreated scenario.

Species	Scenario (seedlings per ha)		
	Low	Moderate	High
Abies concolor	7.4–18.5	25.9–64.8	48.2–120.4
Pinus lambertiana	0.6–2.5	1.9–7.4	4.9–19.8
P. ponderosa	1.2–3.7	3.7–11.1	9.9–29.6
Pseudotsuga menziesii	2.5–9.9	8.6–34.6	16.1–64.2
Calocedrus decurrens	1.2–11.1	4.3–38.9	8.0–72.2
Overall average	29.3	100.7	196.7

landscape (Fig. 1). RANDIG uses the minimum travel time method (Finney, 2002) to simulate fire spread based on user-inputs for: number/pattern of ignitions, fire duration, wind speed and direction, fuel moistures, topography, stand structure, and fuels. For each scenario and time step we simulated 10,000 randomly placed ignitions, burning for 240 min (one 4-h burn period). This burn period duration was selected such that simulated fire sizes (for one burn period) approximated large-spread events (daily) observed in nearby recent wildfires, including 2007 Wheeler, 2007 Moonlight, and 2008 Rich (Ager et al., 2010). Large daily spread events in these fires ranged between 1000 and 6000 ha (Fites et al., 2007; Dailey et al., 2008), which is consistent with the interquartile range (25th–75th percentile) for the untreated landscape (Fig. 3).

For each simulated landscape RANDIG outputs conditional burn probabilities and marginal conditional burn probabilities for 20 flame length classes (0–10 m in 0.5 m increments) for individual 60 m pixels, spanning the entire buffered study area. Conditional burn probabilities are computed by dividing the total number of times a pixel is burned by the total number of simulated fires (n = 10,000). In an attempt to focus on more problematic simulated fire occurrence, both from a fire effects and a fire suppression standpoint, we only performed analysis on the burn probabilities where modeled flame lengths were greater than 2 m. Flame lengths above 2 m typically correspond with crown fire initiation and present substantial challenges for suppression efforts (NWCG, 2004). We chose to use a fixed flame length cutoff rather than one based on individual stand conditions (e.g., Ager et al., 2007, 2010) because of the influence of canopy base height on individual stand flame length calculations. The calculation of canopy base height can often be problematic and exhibit unrealistic ranges in calculated values given small changes in stand conditions (Rebain, 2010). We imported conditional burn probability surfaces, for modeled flame lengths greater than 2 m, into ArcGIS software for further data analysis. For each of the 30 simulated landscapes we computed overall mean conditional burn probability, only using those pixels within the Meadow Valley core study area.

3. Results

3.1. Untreated surface fuel model comparison

The two methods for assigning surface fuel models, FFE-derived and custom override, resulted in very different fire behavior predictions for the 2008 Rich Fire. Based on the FFE fuel models 10% of the Rich Fire area was predicted to burn with flame lengths exceeding 2.4 m, whereas 27% was predicted based on the override fuel models (Table 4). Seventy-four percent of the area was predicted to burn with flame lengths less than 1.2 m based on the FFE fuel models compared to 45% based on the override fuel models. Under the assumption that three flame length classes analyzed, <1.2, 1.2–2.4, and >2.4 m, in this forest type would generally result in low, moderate, and high severity fire effects, respectively, predicted fire behavior using the override fuel models better approximated the observed proportion of high severity, which was 28% (Table 4). The FFE fuel models considerably overpredicted the proportion of area in the low severity or <1.2 m flame lengths (74% vs. 28%). The override fuel models also overpredicted low severity, but to lesser extent (Table 4). Both fuel model scenarios had lower proportions of area in the 1.2–2.4 m flame length class relative to the area classified as moderate fire severity (Table 4).

Table 3
Weather parameters for fire simulations. Parameters were obtained from the Cashman Remote Automated Weather Station. Wind speeds and directions represent the dominant values from all 90th percentile and above observations. Fuel moistures represent the 97th percentile values for the predominant fire season in the area (June 1–September 30).

Weather input	Specific parameter/value				
	Speed (km h^{-1})	Direction (az.)	Relative frequency		
Wind	31	225	0.77		
	32	45	0.15		
	31	180	0.08		
Fuel moisture (%)	1 h	10 h	100 h	Live herbaceous	Live woody
	2	2	5	35	60

3.2. Effects of varying ingrowth and treatment fuel model levels

Despite the considerable range in simulated ingrowth among the three levels we tested (Table 2), average live tree density in forested stands was similar across levels, even for projections out to 2050 (Table 5). This was not the case for canopy base height (CBH). Under the low ingrowth level, average CBH increased throughout the simulation duration (Table 5). Under the high ingrowth level, average CBH initially increased in 2020, but by 2040 and 2050 it decreased to below the 2010 value. The moderate ingrowth level resulted in increased average CBH in 2020 and 2030, but declined to near the 2010 average by 2040 and 2050.

There was a strong effect of varying ingrowth levels on hazardous fire potential (conditional probabilities of fire occurring with flame lengths >2 m) over time. For simplification we only report burn probabilities for the different ingrowth and treated fuel model levels in 2010, 2030, and 2050 (Fig. 4). Under the low ingrowth level (untreated) hazardous fire potential declined over time and by 2050 was well below treated scenarios at any of the time steps (Fig. 4). The moderate ingrowth level resulted in a decline in hazardous fire potential through 2030, but then increased by 2050, exceeding the 2010 value (Fig. 4). There was no initial decline in hazardous fire potential under the high ingrowth level, which increased steadily throughout the simulation duration (Fig. 4).

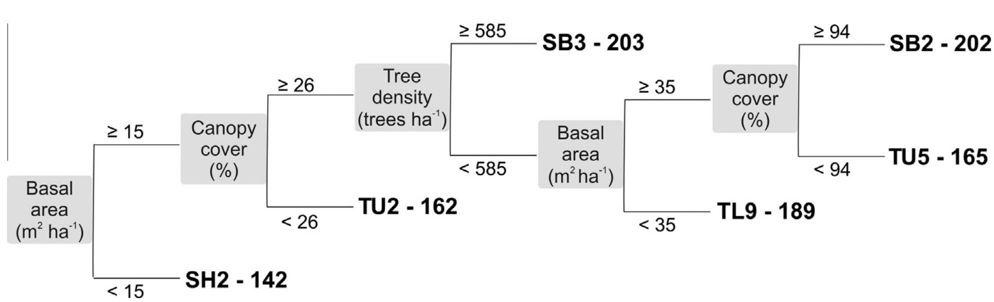

Fig. 2. Surface fuel model selection logic for Meadow Valley untreated forested stands. Surface fuel models were selected from Scott and Burgan (2005) and are identified in bold by code and number.

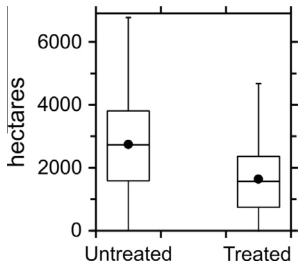

Fig. 3. Burn period fire size box and whisker plots based on 10,000 randomly placed ignitions for the Meadow Valley landscape in 2010. Filled circles represent mean fire size.

Table 4
Proportion of area within the 2008 Rich Fire perimeter by fire severity and predicted flame length classes. Fire behavior predictions were performed using two different surface fuel model layers, one based on fuel models output from the Fire and Fuels Extension (FFE) and the other based on our custom logic informed by regression tree analysis (override). Fire severity classes are based on relative differenced normalized burn ratio (RdNBR) thresholds established by Miller and Thode (2007). Flame length classes are based on fire behavior likely to result in torching and crowing (NWCG, 2004).

Fire severity – predicted flame length class	Observed	Predicted	
	RdNBR based	FFE fuel models	Override fuel models
Low – <1.2 m	0.28	0.74	0.45
Moderate – 1.2–2.4 m	0.44	0.16	0.28
High – >2.4 m	0.28	0.10	0.27

The effect of varying fuel models in treated areas was much less noticeable than that associated with varying ingrowth levels. Note that because we did not perform a full cross of ingrowth and treatment fuel model levels, we held ingrowth at the moderate level for the treated scenarios. Not surprisingly the high fuel model level in treated areas resulted in greater hazardous fire potential than the moderate or low fuel treatment levels (Fig. 4). The difference in hazardous fire potential between low and moderate treatment fuel model levels was minimal (Fig. 4). Hazardous fire potential increased in 2050 for all three fuel model levels, but was well below the untreated scenarios under moderate and high ingrowth levels (Fig. 4). In 2030, hazardous fire potential for the three treated scenarios was well below the untreated/high ingrowth level and marginally below the untreated/moderate ingrowth level (Fig. 4).

3.3. Spatial patterns of treatment impacts over time

Across much of the Meadow Valley landscape, hazardous fire potential for the initial untreated condition (2010) noticeably exceeded those for the treated condition in 2010, 2020, and 2030 (Fig. 5). Note that the conditional burn probabilities in Fig. 5 are based on the moderate level fuel models in treated areas (Table 1) and the moderate ingrowth level (Table 2). The considerable reduction in fire sizes for the treated landscape condition reflects a similar treatment effect (Fig. 3). In the 2010 treated condition, areas

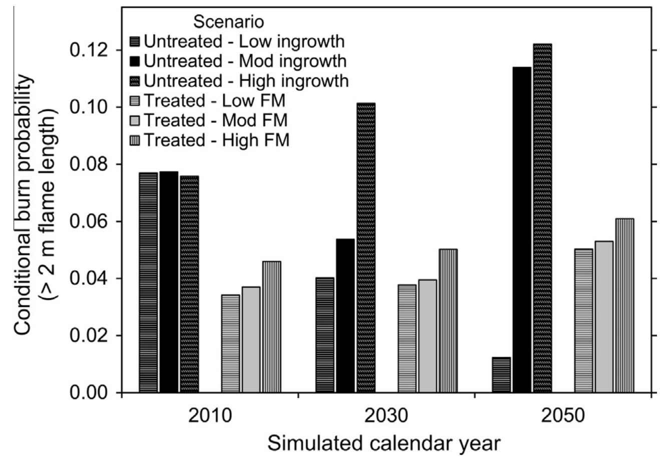

Fig. 4. Conditional burn probabilities across the Meadow Valley landscape for which simulated flame lengths are greater than 2 m. Burn probabilities are reported for the treated and untreated landscape conditions, varying treatment fuel models (FM) and ingrowth levels, respectively. Three levels of each were included in the analyses: low, moderate (mod), and high.

within and on the leeside of treatments had the most obvious reductions relative to the untreated condition, which, for many areas resulted in very low (<0.02) conditional burn probabilities (Fig. 5). These treatment effects existed fairly consistently across the landscape through 2030. By 2040 and 2050, burn probabilities in the central and northern portion of the treated landscape appear similar to the initial untreated landscape. This was not the case in the southern portion of the landscape, which has reduced burn probabilities relative to the initial untreated condition continuing through 2050 (Fig. 5).

4. Discussion

4.1. Influence of critical modeling inputs on hazardous fire potential

There is great need for spatially explicit modeling of fire behavior and effects to provide support for land management planning in fire-prone forests. This is particularly true when analyzing immediate and longer-term impacts of different vegetation management strategies. To assess these short- and long-term impacts, land management agencies often rely on both forest dynamics models (e.g., FVS-FFE – Dixon, 2002) and fire behavior models (e.g., FARSITE – Finney, 1998, FlamMap – Finney, 2006) to produce stand- and landscape-level metrics required in planning documents. While the coupling of these models allows for detailed quantitative output, uninformed use of the models can lead to inaccurate predictions and possibly inappropriate management decisions (Collins et al., 2010). Our findings demonstrate that fire behavior predictions are strongly influenced by surface fuel model selection, primarily in untreated stands, and predictions of CBH. While this has been demonstrated before in stand-scale fire modeling studies (Hall and Burke, 2006), little has been done at the landscape-scale, especially with respect to evaluating fuel treatment networks.

Table 5
Average modeled live tree density and canopy base height for all forested stands across the Meadow Valley study area over time under the three ingrowth scenarios outlined in Table 2.

Ingrowth scenario	Tree density (ha^{-1})					Canopy base height (m)				
	2010	2020	2030	2040	2050	2010	2020	2030	2040	2050
High	537	623	687	736	762	2.4	3.6	2.2	1.1	1.2
Moderate	537	623	686	734	759	2.4	3.6	4.4	2.4	2.0
Low	537	623	686	733	758	2.4	3.6	5.0	6.5	8.0

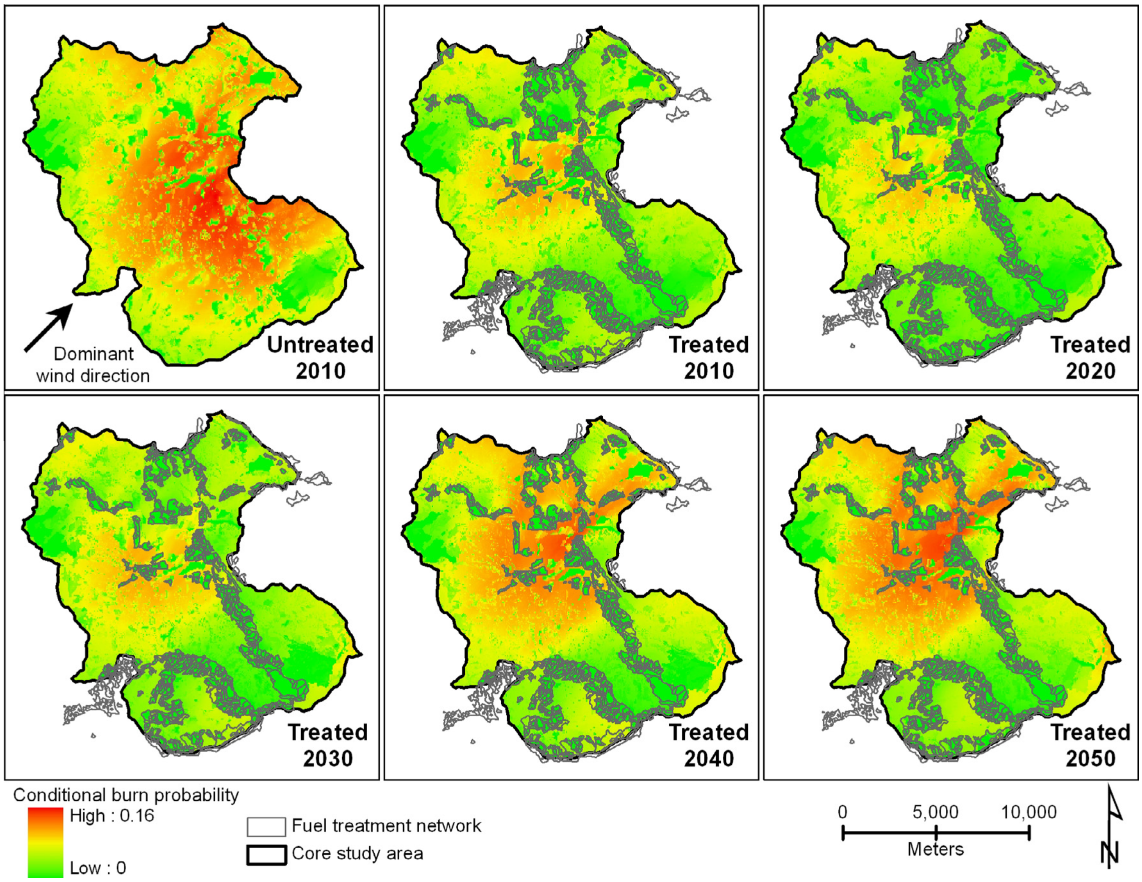

Fig. 5. Conditional burn probabilities across the Meadow Valley landscape for which simulated flame lengths are greater than 2 m. Burn probabilities are reported for the untreated condition, as well as for the treated condition modeled 40 years beyond 2010. Probabilities are based on 10,000 randomly placed ignitions simulated using RANDIG. Treated areas and the predominant modeled wind direction are also displayed.

When modeling forest stands across a landscape through time, both surface fuel model selection and CBH predictions are computed within forest dynamics models. Based on our modeling of the 2008 Rich Fire, it appears that some adjustment to the default surface fuel model selection in FVS-FFE is necessary to approximate more realistic fire behavior/effects (Table 4). Predictions based on our fuel model selection logic resulted in poor differentiation between area burned at low and moderate fire severity. However, predictions for area burned at high severity were good (Table 4). With regard to CBH, the main concern is how it is predicted over time, which, in our modeling approach, was manipulated by varying ingrowth levels over time (Table 2). The impact of the range in the evaluated ingrowth levels on predications of hazardous fire potential over time was dramatic, with large discrepancies in predictions between the different levels in 2030 and 2050 (Fig. 4). The finding that hazardous fire potential declines steadily over time with the low ingrowth level, and even the decline with the moderate ingrowth level in 2030, is counter to what was reported in an empirical study in a similar forest type (Stephens et al., 2012). This study reported both a decrease in canopy base and an increase in torching potential for untreated stands over time, which, based on our results, is only captured in the high ingrowth level (Table 5, Fig. 4). This suggests a potentially significant limitation of FVS-FFE when growing forest stands over time. In the absence of fairly high modeled ingrowth rates (~200 seedlings per ha, every 10 years) canopy base height increases over time, and coincident with that, hazardous fire potential declines. It is unclear whether this ingrowth rate is consistent with empirical observations or the rate is simply an adjustment that compensates for potential deficiencies within FVS-FFE (e.g., overestimated crown rescission, or self-pruning). More long-term data on change in these forests over time, both in untreated and treated stands, is needed to validate/adjust model predictions.

The three treated surface fuel model suites we evaluated capture the range of conditions actually observed, which was driven by both variability within completed treatments and incomplete treatments. For example, the high level treated fuel model assigned to mechanically thinned & prescription burned stands represents an unburned condition with additional activity fuel from thinning, while the low level fuel model represents a recently burned condition, as was planned for this treatment type (Table 1). Given the range in surface fuel conditions modeled among the different levels, we expected to observe a wider range in hazardous fire potential. While the high level treated fuel models consistently resulted in greater hazardous fire potential, the magnitude of differences among treated fuel model levels was much less than that among ingrowth levels for untreated scenarios (Fig. 4). This suggests a fair degree of robustness in the modeled reduction in hazardous fire potential resulting from the fuel treatment network in the Meadow Valley landscape.

4.2. Spatiotemporal patterns of hazardous fire potential

There are several potential sources of error associated with simulating forest dynamics and fire behavior at the landscape scale. One source of error in our study is the use of a stand-level model (FVS-FFE) to generate fire behavior modeling inputs across our study landscape. Our approach used a base vegetation map to delineate stands, which are then simulated independently for the study duration. Aggregating stands to create the continuous vegetation structure and fuel inputs needed to execute RANDIG potentially leads to unrealistic fire behavior predictions across the landscape due to possible abrupt transitions at stand boundaries. Another potential source of error is the two disparate field data collection efforts, 'landscape' and 'treatment' plots. The two efforts had different objectives and different protocols, which led to some inconsistencies in the resolution of the data. For example, in 'landscape' plots we did not obtain detailed tree height and height-to-crown-base measurements, which could result in coarser estimates of CBH in stands outside of the treatment units. In 'treatment' plots we did not obtain robust measurements of surface and ground fuels pre- and post-treatment (we relied on photo-series), which could lead to uncertainty in surface fuel model selection for treated stands. (However, given that we explicitly incorporated multiple levels of surface fuel models for treated stands, this uncertainty is likely not a great concern.) Despite these potential sources of error, and the uncertainties associated with FVS-FFE projections, we submit that our analyses capture the effects of the fuel treatment network in Meadow Valley reasonably well. Our approach incorporated vegetation and fuel data from over 670 field plots in an attempt to capture the diverse vegetation conditions across our large study area. This allowed for more detailed quantification of vegetation structure and fuels compared to efforts based on remotely sensed imagery (e.g., LANDFIRE – see Krasnow et al., 2009). Furthermore, the coupling of forest dynamics models with landscape-scale fire behavior models is being implemented operationally in forest planning (e.g., Collins et al., 2010; Ager et al., 2013). Our findings provide guidance in the use of these models, which potentially improve planning outcomes and management on-the-ground.

Based on our findings, the implemented fuel treatment network in the Meadow Valley study area is effective at reducing both fire size and the occurrence of more hazardous fire. This was the case, not only within the treated areas, but across much of the landscape (Figs. 3 and 5). This landscape-scale effect was achieved with treatment across less than 20% of the approximately 20,000 ha study area. Several previous studies have demonstrated similar landscape-scale reductions in modeled fire behavior with relatively low proportions of treated area (Ager et al., 2007, 2010; Finney et al., 2007; Moghaddas et al., 2010; Collins et al., 2011; Syphard et al., 2011), and in general this is indicative of a successful landscape fuel treatment (Finney, 2001; Collins et al., 2010). What is unique about the results presented here is that, with the exception of follow-up underburning in several mechanically thinned stands, the fuel treatment network analyzed actually exists on the ground, as opposed to a theoretical fuel treatment design. Furthermore, we characterize forested stands throughout our study landscape with data from intensive field sampling efforts, including pre- and post-treatment measurement of many treated stands. Much of the previous work on landscape fuel treatment effectiveness is based on modeled treatment locations and simulated treatment effects on forest structure and fuels. Differences between planned landscape fuel treatments and actually implemented fuel treatment projects in terms of treatment locations, extents, and intensities can be substantial (Collins et al., 2010). Analysis associated with planning often assumes treatments will be fully implemented in all planned treatment units, when in reality there are untreated portions within treatment units or in some cases individual treatment units are entirely untreated (R. Tompkins, personal communication, Plumas National Forest). The reasons for these include slope gradient (i.e., too steep to operate certain equipment), watercourse protection, limited operating periods for wildlife species of concern, and access.

While reductions in hazardous fire potential are evident throughout the study area following treatment implementation, the effects are more pronounced on the leeward side of the relatively linear treatment blocks (Fig. 5). The predominant wind-direction under which fires were modeled was from the south-west, which is consistent with the more problematic fire season winds in this area. Previous studies have reported similar lee-side effect of treatments in actual wildfires (Weatherspoon and Skinner, 1995; Finney et al., 2003, 2005). The lee-side effects are particularly noticeable in the southern and north-eastern portions of the Meadow Valley study area. We hypothesize there are two reasons why the effects are so prominent in these areas: (1) treatment blocks are generally oriented orthogonal to the dominant wind directions, which maximizes the potential for the wind-driven modeled fires to intersect treatments, and (2) the treatment blocks are somewhat layered, which limits the potential for fires to regain spread and intensity after encountering initial treatment blocks. The treatment effects are less pronounced in the central-west portion of the landscape, where treatments are largely absent, particularly towards the windward side (Fig. 5). Based on these results, it appears that the Meadow Valley study area could have benefited from another treatment block in the central-western portion of the landscape. However, there are considerable land management constraints on forest management activities in this portion of the landscape (Moghaddas et al., 2010).

Despite the increasing hazardous fire potential in the central portion of the study area from 2030 through 2050 for the treated landscape (Fig. 5), average burn probabilities for the treated landscape were well below that for the untreated landscape (moderate and high ingrowth levels), particularly in 2050 (Fig. 4). These large differences between the treated and untreated landscapes in 2050, along with the fact that burn probabilities for the treated landscape are lower in 2050 than the initial untreated condition (2010), suggest considerable longevity for the fuel treatment network in Meadow Valley. This is much longer than the 20-yr period of reduced hazardous fire potential reported for a landscape fuel treatment project in the north-central Sierra Nevada (Collins et al., 2011). This discrepancy is likely due to differences in planned maintenance of treated areas over time. Maintaining existing treatments with underburning was planned as part of the Meadow Valley project (R. Bauer, personal communication, Plumas National Forest), but this was not the case for the project analyzed by Collins et al. (2011).

5. Summary and implications for landscape fuel management

The coordinated network of fuel treatments in the Meadow Valley area is a "real-world" example of a landscape treatment design that took into account local knowledge of fire weather and likely fire behavior patterns, but at the same time was fairly constrained by various land designations limiting or restricting treatment area (Moghaddas et al., 2010). Our results demonstrate that this fuel treatment network, which covers approximately 20% of the landscape, can effectively reduce hazardous fire potential across much of the landscape, relative to the untreated condition. These reductions persist throughout our modeling duration (2010–2050), suggesting that a treatment network that is maintained on a 10–20 yr cycle can have long-lasting effect (Chiono et al., 2012). However, our predictions of hazardous fire potential were very sensitive to assumptions on how canopy base height in untreated stands changed over time, which was manipulated by varying ingrowth levels. Under the low ingrowth level hazardous fire potential steadily declined over time for the untreated landscape condition, which is not supported by previous studies that investigated stand structure changes over time in similar forest types (e.g., Stephens et al., 2012). The effect of varying fuel models in treated areas had much less impact on hazardous fire potential.

Although these results clearly support the notion of maintaining an existing fuel treatment network, there are two important considerations related to fuel treatment longevity that should be noted. The first is the feasibility of the maintenance treatments. In the Meadow Valley area the planned maintenance treatments rely heavily on the use of fire, primarily through prescribed burning. In areas where risk of escape to nearby communities or where smoke impacts severely restrict burning, accomplishing maintenance using prescribed fire may have a lower likelihood of occurring. The second consideration is that simply maintaining the existing fuel treatment network may not be enough to maintain low hazardous fire potential. Based on our results it appears that hazard in untreated areas continues to increase, which is also demonstrated empirically at the stand-level by Stephens et al. (2012). It appears that this increased hazard in untreated areas over time ultimately leads to a reduction in overall effectiveness of the fuel treatment network. This is most evident in the central and northern portions of the Meadow Valley study area in 2040 and 2050 (Fig. 5). This suggest that for long-term reduction of hazardous fire potential across landscapes, both the maintenance of an initial fuel treatment network and the establishment of new fuel treatments are needed. If the implementation of an initial fuel treatment network can improve willingness and likelihood for expanded use of managed wildfire (North et al., 2012), perhaps subsequent "treatments" can be augmented by managing lightning-ignitions under less-than-extreme fire weather conditions (Collins et al., 2009).

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