CSIRO PUBLISHING

International Journal of Wildland Fire **2011**, *20*, 364–383 www.publish.csiro.au/journals/ijwf

Simulating landscape-scale effects of fuels treatments in the Sierra Nevada, California, USA

Alexandra D. Syphard[A,E], *Robert M. Scheller*[B,C], *Brendan C. Ward*[B], *Wayne D. Spencer*[D] *and James R. Strittholt*[B]

[A]Conservation Biology Institute, 10423 Sierra Vista Avenue, La Mesa, CA 91941, USA.
[B]Conservation Biology Institute, 136 SW Washington Avenue, Suite 202, Corvallis, OR 97333, USA.
[C]Present address: Department of Environmental Sciences and Management, Portland State University, PO Box 751, Portland, OR 97207, USA.
[D]Conservation Biology Institute, 815 Madison Avenue, San Diego, CA 92116, USA.
[E]Corresponding author. Email: asyphard@consbio.org

Abstract. In many coniferous forests of the western United States, wildland fuel accumulation and projected climate conditions increase the likelihood that fires will become larger and more intense. Fuels treatments and prescribed fire are widely recommended, but there is uncertainty regarding their ability to reduce the severity of subsequent fires at a landscape scale. Our objective was to investigate the interactions among landscape-scale fire regimes, fuels treatments and fire weather in the southern Sierra Nevada, California. We used a spatially dynamic model of wildfire, succession and fuels management to simulate long-term (50 years), broad-scale (across 2.2×10^6 ha) effects of fuels treatments. We simulated thin-from-below treatments followed by prescribed fire under current weather conditions and under more severe weather. Simulated fuels management minimised the mortality of large, old trees, maintained total landscape plant biomass and extended fire rotation, but effects varied based on elevation, type of treatment and fire regime. The simulated area treated had a greater effect than treatment intensity, and effects were strongest where more fires intersected treatments and when simulated weather conditions were more severe. In conclusion, fuels treatments in conifer forests potentially minimise the ecological effects of high-severity fire at a landscape scale provided that 8% of the landscape is treated every 5 years, especially if future fire weather conditions are more severe than those in recent years.

Additional keywords: climate change, LANDIS-II, prescribed fire, wildfire.

Introduction

Wildfire has played a crucial role in shaping the structure and ecological function of coniferous forests throughout the world, including those in the Sierra Nevada of the western United States (Bond and van Wilgen 1996; van Wagtendonk and Fites-Kaufman 2006). Prior to widespread Euro-American settlement, fire was the predominant disturbance in the Sierra Nevada, and fire regimes were characterised by frequent, low- to mixed-intensity surface fires that created a fine-scaled mosaic of vegetation across the landscape (Kilgore and Taylor 1979; Collins and Stephens 2007; Beaty and Taylor 2008). Although the timing and extent of fires naturally varied over the last several millennia (Gavin *et al*. 2007; Beaty and Taylor 2008), the 20th-century policy of fire exclusion reduced fire activity to levels that were far below historical estimates (Keeley and Stephenson 2000), although extensive burning and structural changes did occur owing to logging.

The unforeseen consequence of fire exclusion was that many western forests became denser, with a greater abundance of surface and canopy fuel (Keane *et al*. 2002). Structural changes due to extensive harvesting also occurred (Laudenslayer and Darr 1990). Because the fire–fuel relationship is typically self-limiting in forested ecosystems (i.e. recently burned areas limit subsequent fire size) (e.g. Collins *et al*. 2009), large areas of dense, continuous fuels increase the likelihood that fires will become larger and more severe, to the point that they are considered outside the historical range of variability (Stephens 1998; Keeley and Stephenson 2000; Sugihara *et al*. 2006). As a result, the extent and frequency of high-severity, stand-replacing wildfire has increased substantially since the mid-1980s (Miller *et al*. 2009).

Compounding the difficulties posed by fuel accumulation, projected changes in climate may also favour increased incidence of fire (Lenihan *et al*. 2003; McKenzie *et al*. 2004; Flannigan *et al*. 2005). Increased spring and summer temperatures and earlier spring snowmelts have resulted in more frequent, larger, longer-duration fires since the 1980s because longer, drier summers generally increase the availability of fuels (Westerling *et al*. 2006). Abnormally large and severe fires can result in dramatic reduction in large trees and aboveground live biomass, leading to cascading ecological effects (DellaSala *et al*. 2004; Lehmkuhl *et al*. 2007; Hurteau *et al*. 2008; Scheller *et al*. 2008; Hurteau and North 2009).

Although there is growing consensus that fire exclusion has had a negative effect on natural communities (Backer *et al*. 2004),

 10.1071/WF09125 1049-8001/11/030364

there is continued debate over how to return fire to forested landscapes while also reducing the extent of high-severity fire (Schoennagel *et al*. 2004; Miller *et al*. 2009; Stephens *et al*. 2009). Some argue that only fire should be used to restore forest structure (Parsons *et al*. 1986), but others fear that forest conditions are now so altered that fires will become unacceptably hazardous in wildland–urban interface areas, particularly under severe weather conditions (Miller and Landres 2004). Therefore, mechanical fuels treatments (i.e. reducing vertical and horizontal continuity of canopy fuels) have become widely accepted as necessary management tools for reducing fuel loads and restoring structure to minimise ecological effects of high fire severity (Agee *et al*. 2000; Peterson *et al*. 2005; Schmidt *et al*. 2008; Stephens *et al*. 2009). The aim of treatments is to maintain large trees through the decrease in surface fire intensity and severity (Agee and Skinner 2005). Whereas the effects of fire exclusion are not confined to the Sierra Nevada (e.g. Covington and Moore 1994; Varner *et al*. 2005), it is important to also point out that the necessity to restore forest structure will vary according to the natural fire regime in a region and the extent to which fire exclusion has actually altered ecosystem qualities (Noss *et al*. 2006).

Although the conceptual basis of fuels treatments is well founded, there is ongoing uncertainty regarding their ability to modify fire regimes across broad landscapes. Estimating fuels treatment effects on the overall fire regime is difficult because natural fire regimes vary widely and the effect of a single treatment depends on treatment type, vegetation composition and structure, the natural fire regime, weather conditions and local topography (Stratton 2004). Another source of uncertainty is how treatments will affect subsequent fire behaviour and decrease fire severity under more severe weather conditions (Reinhardt *et al*. 2008). For example, a review by Schoennagel *et al*. (2004) revealed that fuels treatments were largely ineffective under severe weather conditions in the 2002 Hayman Fire in Colorado; however, this may have been due to the small size of the treated area. Nevertheless, the extreme fire weather on the day of the fire strongly contributed to the fire's severity. However, fuels treatments effectively slowed and reduced the severity of the 2002 Rodeo–Chediski Fire in Arizona under extreme fire weather conditions (Finney *et al*. 2005).

Because fire occurs sporadically over large areas, it is difficult to design landscape-scale experiments to evaluate how fuels treatments affect subsequent fire behaviour. Some studies have taken advantage of natural experiments, and there is empirical evidence of how fires respond to individual fuels treatments (e.g. Schoennagel *et al*. 2004; Agee and Skinner 2005; Raymond and Peterson 2005); but there are insufficient empirical examples to make general conclusions. To overcome this shortcoming, model simulation experiments at the scale of individual fires have been developed to evaluate the effectiveness of different forest management approaches for reducing the size and spread of fires (e.g. Miller and Urban 2000; Finney *et al*. 2006; Schmidt *et al*. 2008). Results of these simulations suggest that treatment effects on individual fires may vary as a function of treatment type, treatment frequency or spatial arrangement on the landscape.

Nevertheless, although individual treatments may be effective at the stand scale, there are many remaining questions at the landscape scale (Agee and Skinner 2005). A fundamental concern is that, given the stochastic nature of fire, if fires rarely or never encounter fuels treatments on the landscape, the ability of treatments to reduce fire severity will be minimised (Rhodes and Baker 2008). Furthermore, fuels treatments are fully effective for a limited time, further decreasing opportunities for intersection. Finally, if climate changes, fuel treatment effectiveness may be either compromised or strengthened.

Because of the spatial and temporal scale of the ultimate processes and interactions, a full assessment of fuels treatment effectiveness cannot be accomplished using model simulations that capture only a small fraction of the landscape or individual fire events. A more complete evaluation requires a model that accounts for the multiple, stochastic interactions between disturbance and successional processes that occur over a wide range of environmental conditions and over sufficiently long durations (at least long enough to capture the duration of fuels treatment effectiveness). However, modelling fires, fuels treatments and landscape change over large landscapes ($>1 \times 10^6$ ha) and over long durations (>30 years) necessitates compromises in model detail and the judicious allocation of complexity. If the questions at hand are motivated by landscape-scale processes and interactions, neither the data available for parameterisation and calibration nor the available computational resources warrant inclusion of fine-scale processes and interactions. For example, although flame length is a critical component for predicting fire effects within the simulation of an individual fire, such mechanistic detail must be subsumed within broader indices of fire-caused mortality when modelling large landscapes. Furthermore, if the system has a large inherent uncertainty (Clark *et al*. 2001) due to stochastic disturbances or the vagaries of forest management and policy, sensitivity to fine-scale processes will be relatively minor. By tuning model complexity to match the primary hypotheses, landscape-scale simulations become tractable and better able to inform landscape-scale management and policy.

Our objective was to use a spatially explicit landscape model of wildfire, succession and forest management to evaluate the relative landscape-scale effects of different management actions designed to reduce the spread rate and severity of fires over a 50-year duration. Note that we were explicitly not testing the efficacy of individual treatments at the stand scale, but rather the effect of treatments from the perspective of the total landscape as it changes through time.

Specifically, the primary objectives were to answer these questions:

1. What are the long-term effects of fuels treatments on the fire regime across a large landscape in the Sierra Nevada?
2. What are the relative effects of treatment rate and intensity on the landscape-scale fire regime?
3. Does the effect of fuels treatments on the overall fire regime vary under more extreme weather conditions?

We evaluated the landscape-scale effects of treatments on fire severity in terms of forest age and total biomass at the end of the simulations. Because large patches of high-severity crown fires are likely to kill large, old trees, older forests with greater total biomass were assumed to reflect lower landscape-scale fire severity. We also calculated fire rotation (i.e. the length of time required to burn an area the size of a specific area) to determine whether fuels treatments affected the overall fire regime on the landscape.

Methods

Study area

Our study area was $\sim 2.2 \times 10^6$ ha of forest in the southern Sierra Nevada, CA, including portions of the Sierra, Sequoia and Stanislaus National Forests and Yosemite and Sequoia–Kings Canyon National Parks (Spencer *et al.* 2011; Fig. 1). The region ranges in elevation from 31 to 4409 m and therefore includes substantial variation in topographic and climatic conditions, and includes diverse vegetation types. The climate is primarily Mediterranean, and although precipitation patterns vary over the region, decreasing from north to south and from high elevation to low elevation, more than half the total precipitation occurs as snow in January, February and March (van Wagtendonk and Fites-Kaufman 2006). The fire season occurs during summer and fall (autumn) when there is little rain

The LANDIS-II model and extensions deployed

To estimate fuels treatment effects on landscape-scale fire regimes, we used LANDIS-II, a spatially dynamic and stochastic, landscape-scale forest succession and disturbance simulation model (Mladenoff 2004; Scheller *et al.* 2007, 2010) that has been applied to many forested and shrubland ecosystems throughout the world (e.g. Ward *et al.* 2005; Syphard *et al.* 2006; Scheller *et al.* 2007; Xu *et al.* 2007; Gustafson *et al.* 2010). LANDIS-II was designed to simulate large (up to and exceeding $>10^7$ ha) landscapes, and model complexity is allocated towards the spatial interactions among the principal processes driving landscape change: succession, natural disturbances and forest management. Therefore, by necessity, each component process is individually of moderate or low complexity and fine-scale inference (e.g. the effects of individual fires) is weak. However, the model is well suited to answer research questions that focus on the interaction between vegetation and wildfire at broad spatial extents and long temporal scales (many decades).

LANDIS-II simulates individual tree and shrub species, and each species is characterised by unique life history characteristics (Roberts 1996) including longevity, age of maturity, fire tolerance, shade tolerance, seed dispersal distances, the ability to resprout and reproduction following fire. LANDIS-II does not represent individual trees; rather, trees are binned into species and age cohorts. Multiple species and age cohorts may be present at a single site. Successional dynamics result from interactions among disturbances, species life history behaviours and site conditions on the landscape. Using estimates from Burns and Honkala (1990), refined by expert opinion, we compiled life history characteristics for 23 individual tree species (Table 1). We also defined two chaparral functional types that represent groups of species that share similar life history traits and responses to disturbance (facultative seeders, such as *Adenostoma fasciculatum*, and obligate resprouters such as *Cercocarpus montanus*) (Keeley and Davis 2007). Finally, we developed a riparian functional type composed of willows (*Salix* spp.), black cottonwood (*Populus trichocarpa*) and alders (*Alnus* spp.) (Table 1).

We simulated fire within LANDIS-II using the Dynamic Fire extension (Sturtevant *et al.* 2009), which was designed with an emphasis on landscape-scale fire regimes and stochastic behaviour occurring over many decades. The Dynamic Fire extension simulates the general characteristics of a fire regime, including fire frequency, fire sizes or durations and fire effects (mortality). Fire-induced cohort mortality is not mechanistically simulated, but is class-based. Mortality depends on both the cohort age and the species' parameterised fire tolerance relative to the potential severity of a fire (Sturtevant *et al.* 2009). Young cohorts with low fire tolerance (Table 1) are most susceptible, but old, fire-tolerant trees can be killed by high-intensity fires. Species-specific post-fire regeneration is simulated by specifying a probability of vegetative reproduction or serotiny. Post-fire succession occurs when species disperse into burned areas after fires, depending on their capacity for dispersal and shade tolerance.

In the Dynamic Fire extension, fire spread rate and direction are a function of fuel type, weather, topography and ignition rate (Sturtevant *et al.* 2009). Fuel types represent fuel bed and ladder fuel conditions with unique spread parameters, ignition probability and the crown base height (CBH: the height above ground that the live crown base begins) (Sturtevant *et al.* 2009). Daily weather records, including temperature, wind speed, wind azimuth, relative humidity and precipitation, are required inputs. Daily weather data determine wind speed velocity and direction, percentage curing of grass and fine fuel and larger fuel moisture (Van Wagner 1987) (Fig. 2). Fine fuel moisture conditions and wind speed velocity determine the Initial Spread Index (ISI), which is combined with larger fuel moisture into the Fire Weather Index (FWI) (Amiro *et al.* 2004; Sturtevant *et al.* 2009).

The Dynamic Fire extension calculates *potential* fire severity as a function of crown-fraction burned and fire rate of spread (Sturtevant *et al.* 2009). Crown-fraction burned is a function of foliar moisture content (FMC), CBH and surface fuel consumption (Sturtevant *et al.* 2009). Due to the complex interactions among weather, fuels and topography, simulated fires generally contained a mixture of potential fire severities. Depending on the cohorts present and their mortality, actual severity is generally also mixed.

We used the Dynamic Biomass Fuels extension (Sturtevant *et al.* 2009) to assign fuel types. During model simulations, the extension assigns a single fuel type to every cell in the study area based on species and cohort ages present. Fuel type assignments are dynamic and change depending on succession, disturbance or management activity. For example, following a fire, the fuel type assignment will no longer represent the prefire conditions, but will instead reflect the cohort species and ages present at this new successional stage.

To determine many of our fuels- and fire-related model parameters and assumptions, we used an expert-knowledge approach through close correspondence with fire scientists and fuels specialists of the USDA Forest Service, Region 5. All data on model design, fuels treatment parameters and treatment efficacy were reviewed and approved by our scientific advisory board.

We simulated fuels treatments using the Biomass Harvest extension. In particular, we simulated the effects of fuels management activities on stand structure by thinning (from below) the cohorts present on a site (a reduction in aboveground live biomass). Immediately following a fuels treatment in the

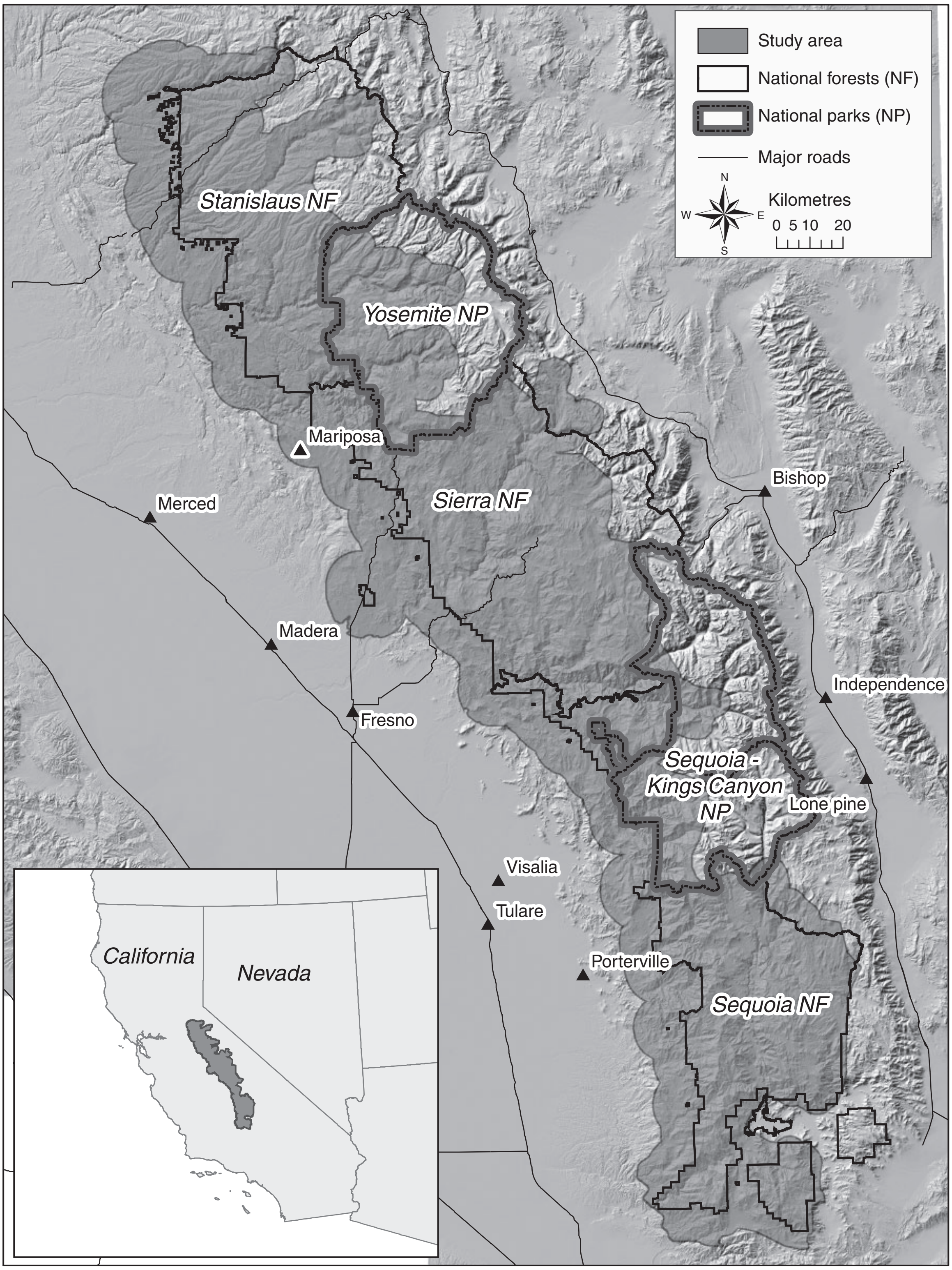

Fig. 1. Study area in the southern Sierra Nevada, CA, USA, showing successional land types.

simulations, we explicitly reassigned fuel types using a set of rules that varied according to the type of treatment, the species and ages present and the assumed efficacy of the treatment. We did not design fuel treatments to stop fires. Rather, the treatments were designed to alter fire behaviour and potential severity based on the suite of characteristics of the treated fuel types (details of fuel treatments are provided under the section *Defining and calibrating fuel types*). Thus, we explicitly

Table 1. Tree and shrub species life history parameters used in LANDIS-II vegetation dynamics model

Shade tolerance ranges from 1 to 5, with 1 requiring full sun to establish and 5 capable of establishment under very low sun. Fire tolerance ranges from 1 to 5, with 1 being least fire tolerant, 5 most fire tolerant

Common name	Species name	Species code	Longevity (years)	Sexual maturity (years)	Shade tolerance	Fire tolerance	Seed dispersal effective distance (m)	Seed dispersal max. distance (m)	Vegetative reproductive probability	Sprout min. age (years)	Sprout max. age (years)	Post-fire regeneration
Lodgepole pine	*Pinus contorta*	Pinucont	150	7	1	1	20	60	0	0	0	None
Ponderosa pine	*P. ponderosa*	Pinupond	350	20	2	4	35	150	0	0	0	None
Sugar pine	*P. lambertiana*	Pinulamb	450	15	3	4	30	150	0	0	0	None
Jeffrey pine	*P. jeffreyi*	Pinujeff	450	18	2	4	50	350	0	0	0	None
Bullpine or gray pine	*P. sabiniana*	Pinusabi	200	18	1	3	30	1000	0	0	0	None
Western white pine	*P. monticola*	Pinumont	400	12	1	4	30	400	0	0	0	None
Giant sequoia	*Sequoiadendron giganteum*	Sequgiga	3000	150	1	5	100	400	0	0	0	Serotinous
White fir	*Abies concolor*	Abieconc	400	40	3	3	30	200	0	0	0	None
Limber pine	*P. flexilis*	Pinuflex	1000	20	2	1	30	5000	0	0	0	None
Red fir	*A. magnifica*	Abiemagn	400	40	3	3	30	200	0	0	0	None
Douglas-fir	*Pseudotsuga menziesii*	Pseumenz	300	15	2	3	30	1000	0	0	0	None
Mountain hemlock	*Tsuga mertensiana*	Tsugmert	600	20	5	1	30	250	0	0	0	None
Incense cedar	*Calocedrus decurrens*	Calodecu	500	40	4	3	30	3000	0	0	0	None
Sierra juniper	*Juniperus occidentalis*	Juniocci	1000	20	2	1	2	500	0	0	0	None
Quaking aspen	*Populus tremuloides*	Poputrem	175	15	1	2	30	7500	0.95	1	175	Resprout
California black oak	*Quercus kelloggii*	Querkelo	300	30	3	2	30	1000	0.8	1	300	Resprout
Canyon live oak	*Q. chrysolepis*	Querchry	250	20	3	1	30	1000	0.95	1	250	Resprout
Blue oak	*Q. douglasii*	Querdoug	250	20	3	1	30	1000	0.8	1	250	Resprout
Interior live oak	*Q. wislizeni*	Querwisl	200	20	4	1	30	1000	0.8	1	200	Resprout
Chamise	*Adenostoma fasciculatum*	Adenfasc	100	10	1	1	5	10	0.7	3	100	Resprout
Mountain mahogany	*Cercocarpus montanus*	Cercmont	150	5	2	2	50	500	0.95	3	150	Resprout
Annual grasses		Anngrass	5	1	1	1	100	10 000	0	0	0	None
Riparian areas[A]		Riparian	150	5	1	1	50	3000	0.95	5	150	None

[A]Riparian areas represent deciduous species common along stream corridors, including willows (*Salix* spp.), black cottonwood (*Populus trichocarpa*) and alders (*Alnus* spp.).

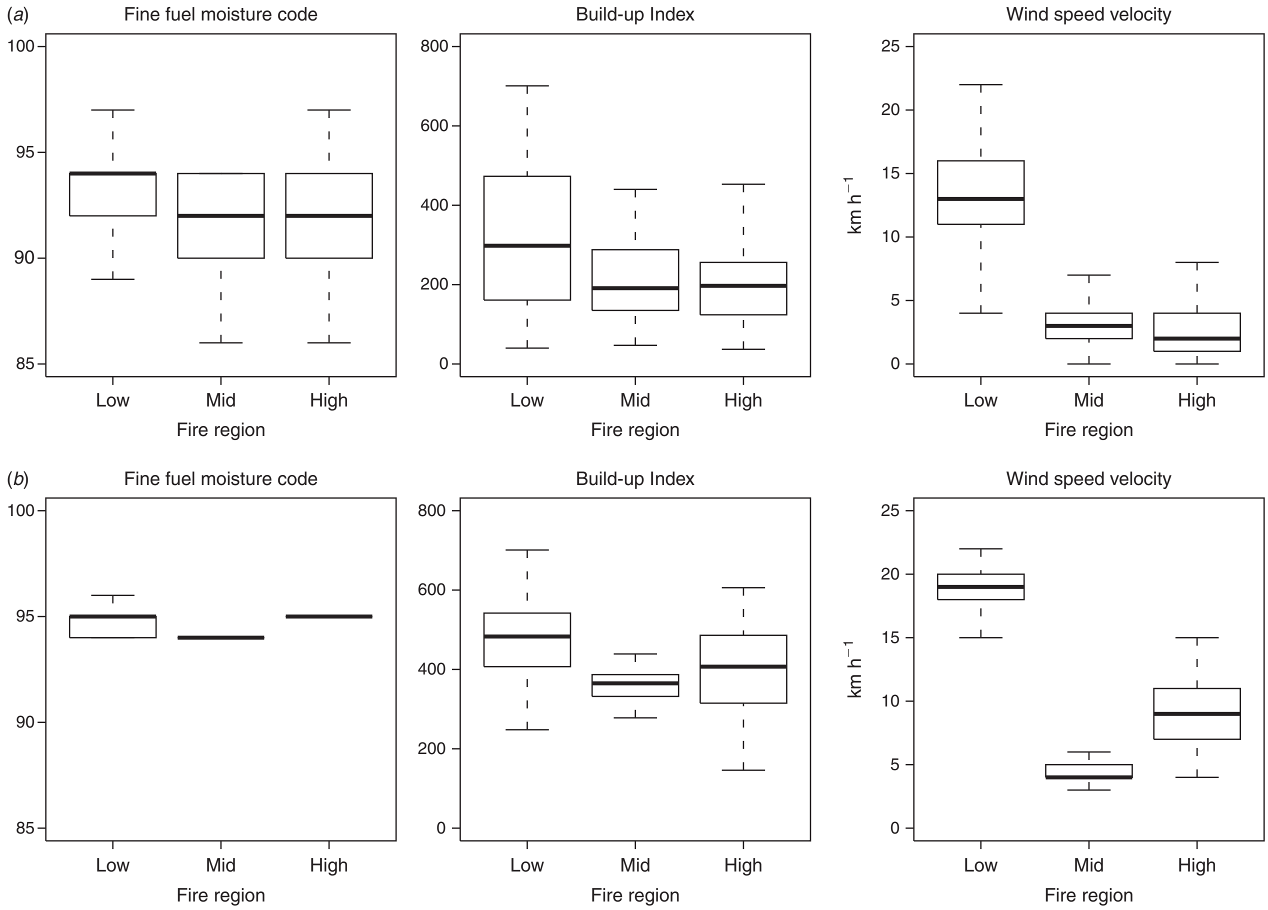

Fig. 2. The fine fuel moisture code (FFMC), Build-up Index (BUI) and wind speed velocity (WSV) used for simulating the baseline (*a*) and high fire (*b*) regimes in the Sierra Nevada, CA.

assumed that individual fuels treatments changed fire behaviour once a fire intersected the treatment, and our objective was to evaluate the cumulative ecological effects of these treatments at a landscape scale where fire and landscape dynamics are stochastic.

Succession parameterisation and calibration

Within LANDIS-II, the landscape is divided into a grid of square cells that are aggregated into land types (or 'eco-regions') that represent relatively homogeneous climatic and soil conditions that capture differences in species' ability to establish and grow. For the southern Sierra Nevada, we used a 100-m cell resolution. We stratified the study area into ecoregions with an unsupervised clustering approach to derive seven land types from six environmental data layers (e.g. Franklin 2003). The initial vegetation for the forested area of the landscape was represented as combinations of species present in different age classes. To estimate the initial community composition of the landscape, we used a combination of California Wildlife Habitat Relationship (CWHR) tree size classes and forest type data (Mayer and Laudenslayer 1988) and Forest Inventory and Analysis (FIA) data (Hansen *et al*. 1992). We regressed the log of the largest-diameter tree against log stand age across all plots within the study area ($n = 608$) to estimate relationships between diameter and age for each species. Next, for each FIA plot, we derived an age estimate for each tree based on these diameter–age relationships. We then crosstabulated the FIA diameter and species data with the CWHR size classes and forest types so that we could assign the more detailed species and age data of the FIA plots to the broadly classified polygons of the CWHR data using a stratified random assignment.

The LANDIS-II Biomass Succession extension (version 2.0) (Scheller and Mladenoff 2004) simulates cohort regeneration, growth, inter- and intra-specific competition and tree mortality. The extension requires estimates of the probability of establishment (P_{EST}) and maximum aboveground net primary productivity ($ANPP_{MAX}$) by species and land type. P_{EST} was estimated through consultation with USDA Forest Service Region 5 silviculturists. $ANPP_{MAX}$ for each species was estimated from Forest Vegetation Simulator (FVS) (Dixon 2002) simulations of the FIA plots within the study area. We subsequently iteratively calibrated LANDIS-II growth estimates by comparing 50-year estimates of aboveground biomass (AGB, Mg ha^{-1}) for 24 FIA plots produced by LANDIS-II and FVS. The final set of LANDIS-II and FVS estimates of AGB had an R^2 of 0.50.

Delineating fire regions

We stratified the study area into three fire regions that broadly reflect the effect of elevation and moisture on regional fire regimes (Agee 1993). The classes included low (up to 1190 m), medium (~1190–2120 m) and high (above 2120 m) elevations that roughly correspond to the foothill shrubland and woodland, lower-montane forest and upper-montane forest ecological zones respectively (van Wagtendonk and Fites-Kaufman 2006). Fire regime parameters that varied by elevation included fire size (or duration), daily weather conditions, Fine Fuel Moisture Code (FFMC) and number of fires (detailed below in calibration section). Seasonal FMC was estimated by USDA Forest Service Region 5 fire scientists. Topographic data were assigned to each cell. Fires can spread across fire region boundaries, and the model adjusts the fire size and spread rate to account for differences among fire regions. Therefore, boundaries between these elevation classes will not be reflected in fire behaviour.

We further stratified our fire regions by wildland–urban interface (WUI) to reflect the potential human influence on ignition rates. We increased ignition probabilities in the WUI regions to reflect the added number of fires caused by humans. Without empirical data to estimate the increase in ignitions, we aimed to adjust the probabilities so that fire rotation was, on average, 25% shorter in the WUI (based on Syphard *et al*. 2007). To simulate the combined effects of more ignitions and better fire suppression capabilities in heavily populated areas, we also calibrated the model to simulate smaller fire sizes in the WUI areas. The net effect was more frequent, smaller fires in the WUI (ranging from 46 to 82% smaller). We delineated WUI areas using data from a map of WUI in the conterminous United States (developed from 2000 US Census data and land-cover data from the US Geological Survey National Land Cover Dataset) (Radeloff *et al*. 2005) and also incorporated roads data (buffered to 100 m) from the 2000 US Topologically Integrated Geographic Encoding and Referencing system TIGER/Line files (US Census 2000).

Our source of daily weather data for the fire regions was the California Climate Data Archive produced by the Western Regional Climate Center of Scripps Institution of Oceanography and the California Energy Commission (http://www.calclim.dri.edu/stationlist.html, accessed 7 April 2011). For all Remote Automated Weather Stations (RAWS) within the fire regions, we downloaded the full available history of daily weather data. Because some weather calculations (e.g. fine fuel moisture code (FFMC) and Build-up Index (BUI)) require all days to be present within the fire season, we evaluated the data to find the combination of stations that provided the longest complete histories.

Study design

To assess the effectiveness of fuels treatments at a landscape scale, we developed a factorial experimental design to examine relationships between fire regime, fuels treatment rate and fuels treatment intensity. Specifically, we developed and evaluated: (1) two fire regimes: Baseline or High Fire; (2) three fuels treatment rates: 2, 4 or 8% of the treatable landscape (every 5 years); and (3) two fuels treatment intensities: Light Thin or Medium Thin.

For all combinations of treatment and fire (12 in total), we ran model simulations for 50 years. We also ran simulations for both fire regimes without treatment to use as reference conditions. Owing to the stochastic nature of the model, we replicated each factorial combination 10 times, which resulted in a total of 120 simulations. Details of the fire regimes, treatment rates and treatment intensities are detailed in following sections.

We evaluated the landscape-scale effects of treatments on fire severity in terms of mean forest age and total AGB, and we also evaluated fire rotation. Total AGB and mean age of forests were calculated at the end of each simulation, at Year 50. Owing to differences in fuel types and fire regimes, we conducted our analyses separately for the three elevation-defined fire regions. We did not evaluate the WUI fire regions separately in the analysis.

Defining and calibrating fire regimes

We developed parameters to simulate two different fire regimes. The 'Baseline Fire Regime' represented fire patterns similar to those observed during the last 20 years (1985–2006), which reflect a recent trend towards increased size and extent of fires (Westerling *et al*. 2006; Miller *et al*. 2009). We estimated baseline fire regime parameters from historic fire perimeter data (http://frap.cdf.ca.gov/data/frapgismaps/download.asp, accessed 7 April 2011). We calculated mean historical fire rotation over the period 1985–2006 by dividing the area of the fire regions by the mean area burned per year. We did not calculate fire rotation separately for the WUI fire regions because the WUI areas were mapped using census boundaries from the year 2000. The number of houses in the WUI was historically much lower, and the extent of the WUI was smaller in the past (Radeloff *et al*. 2005). Therefore, we would not expect the influence of human-caused ignitions to be fully reflected in summary statistics of historic data.

We calibrated the Dynamic Fire extension such that the simulated mean fire rotation, across 10 model replicates, was within 25 years of the historic mean (for the last 20 years) for the low, middle and high fire regions. We also calibrated the extension to simulate fire sizes within 100 ha of empirical means. During our calibration process, we iteratively varied three model parameters: mean fire duration, mean variability of the duration and number of ignitions attempted. We calibrated the Baseline Fire Regime so that the mean landscape potential fire severity was slightly skewed around a mean of ~3.5 on a scale of 1 to 5, where 1 reflects the relative severity of a ground fire and 5 reflects high-severity crown fire (Sturtevant *et al*. 2009). We calibrated the mean potential landscape fire severity to reflect findings of Miller *et al*. (2009), who reported that fire severity in the Sierra Nevada has been increasing in recent years. In LANDIS-II, post-fire mortality is a function of age and species-specific fire tolerance relative to the fire event. Therefore, the calibration of potential fire severity at 3.5 meant that, on average, the mean fire intensity translated to moderate fire severity.

High fire regime

Current trends and climate projections suggest that wildfires are likely to become larger and more intense in the Sierra Nevada, with a longer fire season that may also increase ignitions and fire frequency (Lutz *et al*. 2009). Therefore, we developed a 'High Fire' regime to determine if management effectiveness would vary under heightened fire conditions. We did not try to attribute

Table 2. Fire Weather Indices (FWI) broken into five quintile classes for the baseline calibration and the high fire regime, which reflects more extreme weather conditions than the baseline

The FWI represents a single integration of fire weather

Percentile	Class	FWI baseline middle elevation	FWI high fire middle elevation
97–100	Extreme	35.14–37.17	36.00–37.17
90–96	Very high	34.58–35.13	35.53–35.99
75–89	High	33.27–34.57	35.21–35.52
50–74	Moderate	27.55–33.26	34.85–35.20
0–49	Low	10.01–27.54	12.34–34.84

these weather changes to any particular climatic cause or to project exactly how the fire regime might change. Instead, our goal was to determine the degree to which fuels treatments may affect landscape-scale fire regimes when weather conditions and fires are more severe.

To create the high fire regime, we used the FWI to select a subset of historic weather records that reflected the most severe weather conditions. We selected and used those records with FWIs that were originally scaled as 'Extreme' in the baseline calibrated regime (Table 2). We also specified a higher mean potential fire severity in the high fire regime (~4.5) owing to the projections that fires are likely to become more intense in the future. Lutz *et al*. (2009) projected that the area burned at high severity will increase by ~22%, and our scenario of increased severity is potentially higher than what is likely to occur under climate change.

Defining and calibrating fuel types

We defined fuel types based on characteristic species assemblages and age ranges that together exemplify relatively uniform fire behaviour and rates of spread (Table 3). Fuel types fell into seven basic groups: Mixed Conifer, Red Fir, Pines and White Fir, Sequoia, Lodgepole and Hemlock, Chaparral and Deciduous (predominately oaks). Within each group, fuel types were further divided into age groups: young, mid-aged and old. We also created two fuel types to represent fuel conditions following treatment, depending on the intensity of the treatment (described below).

Each fuel type exhibits characteristic rates of spread. Beginning with the fuel type coefficients defined in the Canadian Forest Fire Behaviour Prediction System Forestry (Forestry Canada Fire Danger Group 1992), we modified some of the coefficients where necessary to reflect rates of spread characteristic of Sierra Nevada fuels based on expert opinion (Sturtevant *et al*. 2009, appendix D) (Fig. 3). Unfortunately, fire behaviour experimental data to confirm our fuel type parameters were not available. Regional fire experts also provided estimates of CBH for each fuel type. For the deciduous fuel type, we derived parameters (from fuel class TL6; broad-leaf deciduous) from the Fire Behaviour Fuel Models (FBFM) developed by Scott and Burgan (2005).

Fuels treatments

In collaboration with the regional fire and fuel experts, we defined fuel treatments (below) that are broadly representative of current and anticipated management activity. Similarly, we used a panel of local managers and fire ecologists (listed in the Acknowledgements) to estimate the effectiveness and duration of each of our fuel treatments. Therefore, both the immediate treatment effects (removal of existing trees) and treatment efficacy (i.e. removal of slash and alteration to fire spread rates) were assumed at the stand scale.

We restricted all simulated fuels treatments to those areas that could potentially be treated by the US Forest Service. This potentially treatable area included lands inside national forests but excluded non-treatable designations, such as existing and recommended Wilderness Areas, existing and recommended Wild and Scenic River areas (*Wild and Scenic Rivers Act* 1968), Research Natural Areas, non-vegetated land and spotted owl (*Strix occidentalis caurina*) Protected Activity Centers (PACs) (USDA Forest Service Region 5). We subdivided the potentially treatable area into two slope categories (>30 or ≤30% slope) because mechanical treatments typically cannot be performed on slopes >30%.

We divided the landscape into management units, which were further divided into stands. Because our stands were not shaped to contain fire or clustered or set into arrays, they are not equivalent to Strategically Placed Area Treatment (SPLAT), e.g. Schmidt *et al*. (2008). Within each stand, the aboveground live biomass of individual cohorts was reduced as prescribed. Note that fuels treatments only directly affect cohort aboveground biomass. Any effect on fire behaviour is assumed when different fuel types with reduced rates of spread and intensity are assigned to a treated area following the management activity.

The highest treatment rate tested (8% per 5 years) (equivalent to ~1/3 of the landscape over 20 years), approximated the proportion of the landscape that fire-spread modelling suggests should be treated to substantially reduce fire incidence on a forested landscape (Finney *et al*. 2006). The lower treatment rate (2% per 5 years) was intended to approximate the current treatment rate in the region's national forest lands (USDA Forest Service Region 5).

Light- and medium-thin treatments represented a combination of mechanical treatment followed by prescribed fire. On slopes ≥30%, a third treatment, Prescribed Fire, was simulated alone. Following a treatment, a stand was assigned a new fuel type (Fig. 3) with an assumed rate of spread and duration (10 or 15 years, details below) (USDA Forest Service Region 5). Following this maximal efficacy period, stand fuel type was assigned based on stand structural characteristics alone. Each treatment included a prescribed fire component; therefore slash would be substantially reduced.

Prescribed fire

The Prescribed Fire treatment, designed to emulate effects of a 1.22-m (4-foot) flame length, was applied as a stand-alone treatment only on slopes >30% because this was considered too steep for mechanical thinning treatments. On these slopes, stand age had to be greater than 50 years, and the stands had to be dominated either by pines, firs or Douglas-fir (aged 40 to 200 years) or by oaks (aged 40 to 200 years). Prescribed fires could only be re-applied if 10 years had passed since the last treatment. The treatment removed the biomass of tree cohorts according to a declining curve, with the largest percentage

Table 3. Fuel type parameters used in the Dynamic Fire extension for LANDIS-II

The default grass fuel type is not shown. Min. age and Max. age reflect the age range for the fuel type assignments. BUI, Build-up Index; BE, Build-up Effect; CBH, Crown Base Height; for equations and definitions, see Sturtevant *et al*. (2009)

Description	Assigned number (for Fig. 3)	Ignition probability	Mean BUI	Max. BE	CBH (m)	Min. age (years)	Max. age (years)	Characteristic species (species codes in Table 1)
Young mixed conifer	FT1	0.01	64	1.321	1	0	40	juniocci abieconc pseumenz pinupond pinulamb calodecu
Mid-aged mixed conifer	FT2	0.01	64	1.321	2	41	80	juniocci abieconc pseumenz pinupond pinulamb calodecu
Old mixed conifer	FT3	0.01	66	1.184	4	81	1000	juniocci abieconc pseumenz pinupond pinulamb calodecu
Young pine–white fir	FT4	0.01	64	1.321	1	0	40	pinulamb pinujeff pinumont pinualbi abieconc pinupond
Mid-aged pine–white fir	FT5	0.01	62	1.261	2	41	80	pinulamb pinujeff pinumont pinualbi abieconc pinupond
Old pine–white fir	FT6	0.01	62	1.261	5	81	1000	pinulamb pinujeff pinumont pinualbi abieconc pinupond
Young red fir	FT7	0.01	64	1.321	1	0	40	abiemagn
Mid-aged red fir	FT8	0.01	72	1.076	2	41	80	abiemagn
Old red fir	FT9	0.01	72	1.076	8	81	1000	abiemagn
Young sequoia	FT10	0.01	64	1.321	1	0	40	sequgiga
Mid-aged sequoia	FT11	0.01	62	1.197	3	41	80	sequgiga
Old sequoia	FT12	0.01	62	1.197	10	81	3000	sequgiga
Young lodgepole–hemlock	FT13	0.01	64	1.321	1	0	40	pinucont tsugmert
Mid-aged lodgepole–hemlock	FT14	0.01	62	1.197	2	41	80	pinucont tsugmert
Old lodgepole–hemlock	FT15	0.01	62	1.197	5	81	1000	pinucont tsugmert
Young shrubs	FT16	0.01	64	1.321	1	0	40	cercmont querchry querwisl adenfasc
Mid-aged shrubs	FT17	0.02	64	1.321	1	41	80	cercmont querchry querwisl adenfasc
Old shrubs	FT18	0.02	64	1.321	1	81	1000	cercmont querchry querwisl adenfasc
Young deciduous	FT19	0.001	32	1.179	1	0	40	querkelo querdoug poputrem riparian
Old deciduous	FT20	0.001	32	1.179	2	41	1000	querkelo querdoug poputrem riparian
Intensity A	FT90	0.0001	500	1.179	2			Effective for 15 years (or 10 years for chaparral) following Prescribed Fire or Light Thinning
Intensity B	FT91	0.0001	500	1.076	4			Effective for 15 years (or 10 years for chaparral) following Medium Thinning

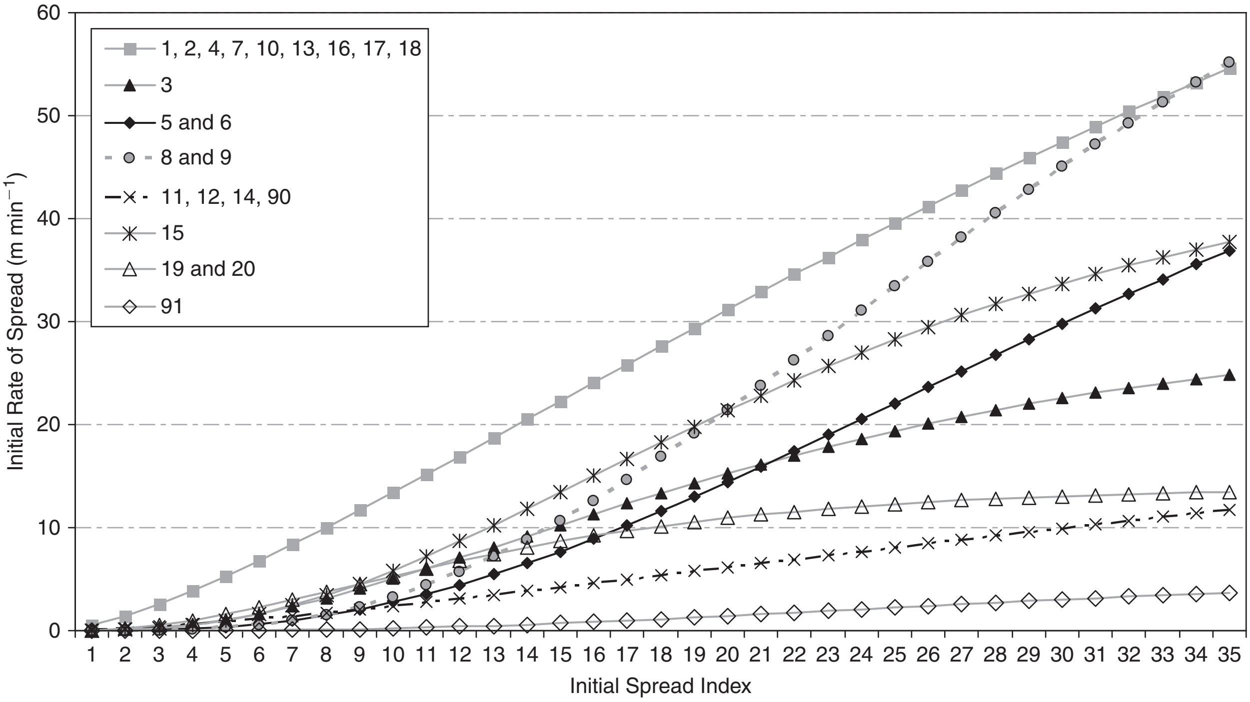

Fig. 3. Initial Rate of Spread (RSI) as a function of the Initial Spread Index (ISI) for 20 fuel types in the Sierra Nevada, CA. The numbers in the legend corresponds to the fuel type numbers in Table 4.

biomass removed for the youngest cohorts, and the lowest percentage biomass removed for older cohorts (Fig. 4). Following a Prescribed Fire treatment, the stand was assigned Fuel Type 90 (FT90) (Table 3, Fig. 3) for 10 years. Fuel Type 90 is consistent with the expectation that fuel loads will be significantly reduced following prescribed fires or mechanical thinning plus prescribed fire (Stephens and Moghaddas 2005).

Light thinning followed by prescribed fire (Light Thin)

The Light Thin prescription was designed to emulate thinning from below with understorey trees up to 12′ (3.66 m) in diameter being removed (Fig. 4) followed by a prescribed fire with a 0.61-m (2-foot) flame length. We assumed that light thinning combined with prescribed fire would not leave any slash on the ground, and therefore stands treated with Light Thin were assigned to FT90 (Table 3, Fig. 3) for 15 years. Stands had to have a minimum age of at least 50 years and treatment could only occur after 20 years since the last treatment.

Moderate thinning followed by prescribed fire (Medium Thin)

The Medium Thin prescription was designed to emulate a more intense thinning from below (relative to Light Thin) with trees up to 30′ (9.14 m) in diameter removed, followed by a prescribed fire with a 0.61-m (2-foot) flame length (Fig. 4). Following Medium Thin, stands were assigned to Fuel Type 91 (FT91) (Table 3, Fig. 3) for 15 years. The stand qualifications for application were identical to the rules for Light Thin.

Analysis

We estimated simple, bivariate linear regression models using the R 2.7 statistical programming environment (R Development Core Team 2004) to estimate the independent contributions of total aboveground live biomass removed, treatment rate and treatment intensity (explanatory variables) on fire rotation period, mean forest age and total AGB (dependent variables) for the baseline and high fire regime. Using total AGB removal as an explanatory variable allowed us to quantify the combined overall effect of treatment rate and treatment intensity at a landscape scale. Biomass removed by treatment and total AGB at Year 50 are mechanistically linked and without fire, we would expect a negative correlation between the two. However, the amount of biomass removed for treatment was a very small percentage of total biomass and induced more vigorous growth of the remaining trees; thus, any significant correlation (below) largely reflected the contravening influence of wildfire.

Results

Simulated fire regime – no treatment

Over the past 20 years, the mid-elevation (~1190–2120 m) fire region had a longer fire rotation than the higher-elevation (>2120 m) or lower-elevation (<1190 m) fire regions (Table 4). The fire rotation in our baseline fire regime simulations followed the same trends and was within ±35 years of the empirical means. The mean simulated fire sizes were also within 100 ha of those calculated for the last 20 years.

Under the high fire regime, fire rotation was shorter than the baseline by 18% (low elevation) to 37% (high elevation), and fire sizes increased by 12% (mid-elevation) to 68% (high elevation) (Table 4), resulting in a substantial increase in fire. The overall distribution of fire frequency on the landscape was similar in both fire regimes, but increased fire frequency in the

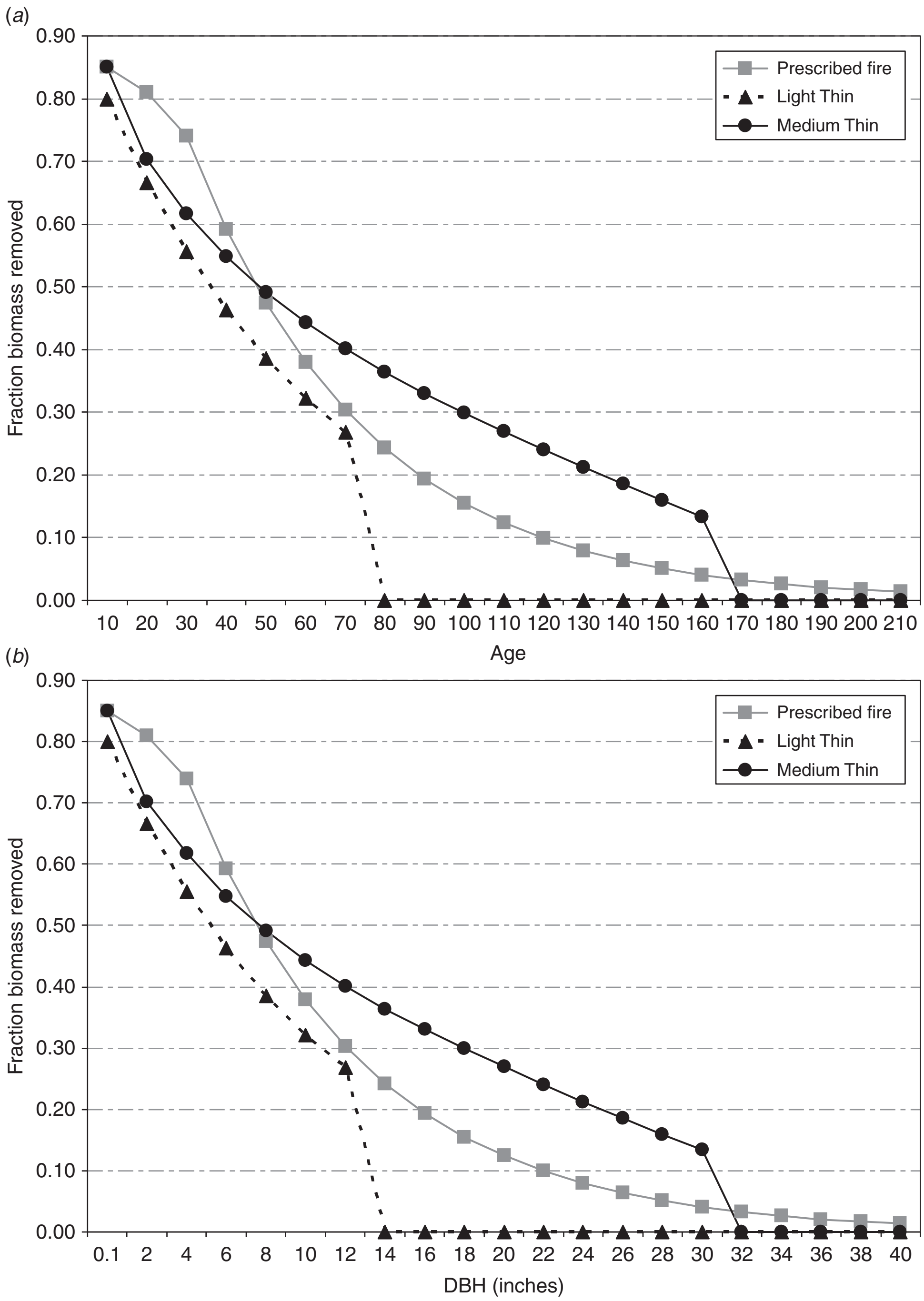

Fig. 4. The proportion biomass removed as a function of (*a*) age (years) and (*b*) diameter at breast height (DBH) for three prescriptions (1 inch = ~2.5 cm). The Light Thin and Medium Thin prescriptions include biomass removed from mechanical thinning followed by prescribed fire. The lines represent the curves for mixed conifers (ponderosa pine, Douglas-fir, Jeffrey pine, sugar pine) as a representative example.

high fire regime was particularly evident in the high elevations and in the southern portion of the landscape (Fig. 5).

Intersection of fires and fuels treatments

Under all fuels treatment combinations, the area of fuels treatments and fire intersections was highest in the mid-elevation region and lowest in the low-elevation region, although there was substantial variability among the replicates (Fig. 6). Also, there was a consistently larger area in which fuels treatments and fires intersected in the high fire regime than the low fire regime. In terms of the proportion of fires that intersected with fuels treatments, the mid-elevation region was slightly

Table 4. Fire regime statistics for the three elevation fire regions in the southern Sierra Nevada, USA

	Fire region		
	Low elevation	Mid elevation	High elevation
Empirical fire rotation period (FRP) (years)	90	140	120
Mean simulated baseline FRP (years)	89	175	141
Mean high fire regime FRP (years)	79	134	90
Empirical fire size (ha)	401	513	577
Mean calibrated baseline fire size (ha)	458	495	544
Mean high fire regime size (ha)	563	592	963
Empirical maximum fire size (ha)	19 460	32 060	60 490
Mean simulated baseline fire size (ha)	25 255	18 211	36 237
Mean high fire regime size (ha)	30 186	22 609	46 690

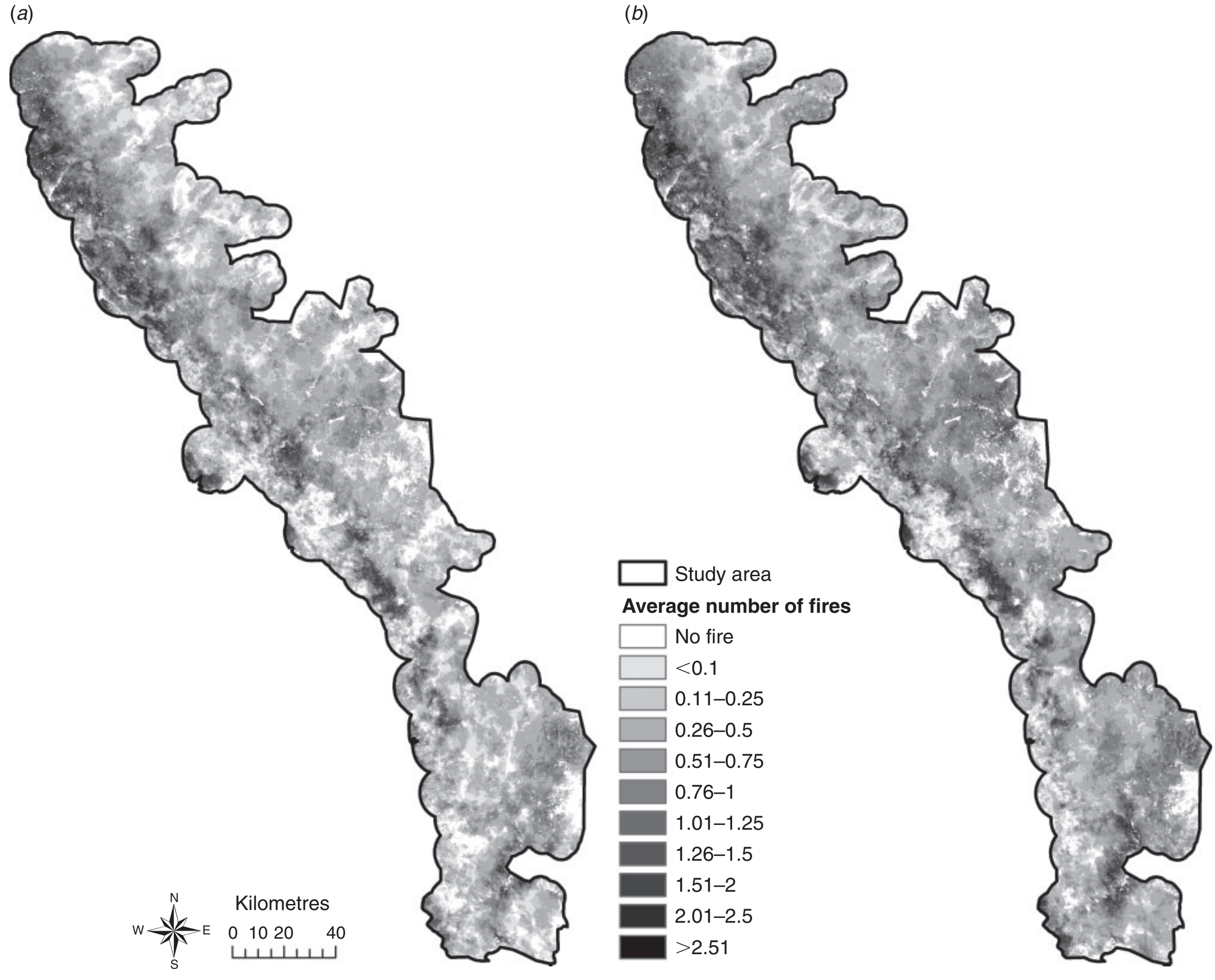

Fig. 5. Distribution of fires on the landscape for (*a*) the baseline fire regime and (*b*) the high fire regime. The range of low to high fire frequency represents the number of fires that occurred in each grid cell for 10 replicates of 50-year simulations.

higher than the high-elevation region, but a much lower proportion of fires (by 98%) intersected treatments in the low-elevation region (Fig. 6). These proportions were similar in the high fire regime.

Fuels treatment effect on mean forest age

For all treatment combinations and under both fire regimes, the simulated mean maximum age of forests was older than the age

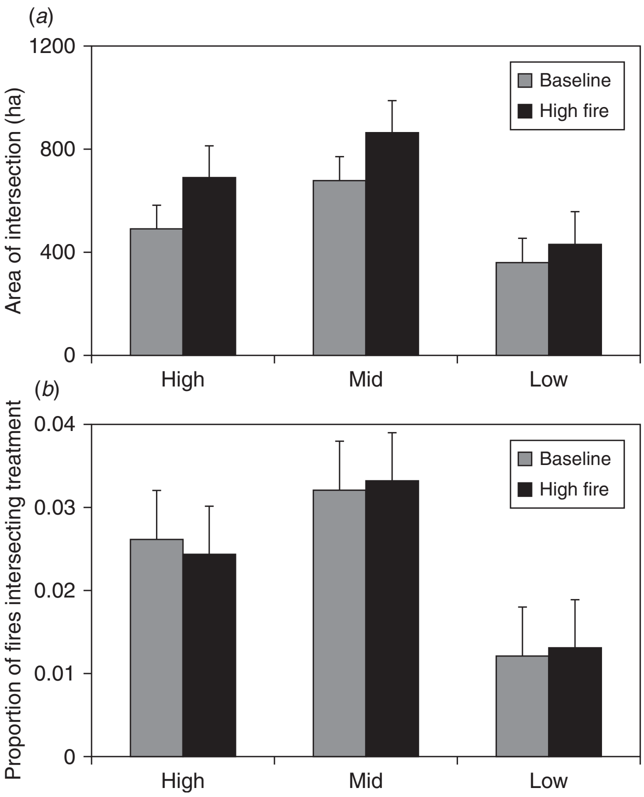

Fig. 6. Area of intersection (*a*) and proportion of fires (*b*) that intersected with fuel treatments for the baseline and high fire regimes in the low-, mid- and high-elevation fire regions. Error bars represent one standard deviation ($n = 10$ replicates).

of forests when no treatment was performed (Fig. 7). The mean forest age differed according to fire region, fire regime and treatment rate and intensity (Figs 7, 8*b*). The mid- and high-elevation regions were more significantly affected by treatment than the low-elevation region ($R^2 = 0.02$–0.15), and treatment intensity was statistically insignificant in all fire regions in the baseline fire regime. Under the high fire regime, the effect of treatment was more substantial compared with the baseline fire regime ($R^2 = 0.11$–0.36). Total biomass removed, treatment rate and treatment intensity all explained significant variation in forest age for the low-elevation region under the high fire regime (Fig. 8*b*).

Fuels treatment effect on total aboveground biomass

The total AGB at the end of the simulations was either the same or higher when there were fuels treatments than if there were no treatments (Fig. 9). This is the opposite of what would be expected if there were no wildfires. In other words, removing some biomass for treatment generally resulted in a greater overall amount of biomass on the landscape after 50 years of simulations. The effect of treatment, again, was stronger in the mid- and high-elevation regions, and in fact, there was almost no effect ($R^2 = 0$–0.003) under both fire regimes in the low-elevation region (Fig. 8*c*). Also, AGB was only significantly higher with treatments than without treatment under the high fire regime. In the high fire regime, the effect was slightly stronger in the high-elevation region, and biomass removal, treatment rate and treatment intensity all explained significant variation in the total AGB.

Fuels treatment effect on fire rotation

The effect of fuels treatments on fire rotation in the baseline and high fire regimes also varied by fire region and according to treatment rate and intensity (Figs 8*a*, 10). In the low-elevation fire region, fire rotation lengthened slightly with fuels treatments (not shown), and was significantly positively related to total AGB removed (Fig. 8*a*). However, the effects of treatment rate and intensity were not significant when examined separately. In both the mid- and high-elevation regions, fire rotation lengthened substantially with fuels treatments (from 11 to 31 years), but was quite variable in the mid-elevation region (Figs 8*a*, 10). Under the baseline fire regime, fire rotation was significantly positively related to total AGB removed and treatment rate, but was not significantly affected by treatment intensity. Under the high fire regime, treatment explained almost twice the variability than was found under the baseline fire regime, and fire rotation was significantly related to total AGB removed, treatment rate and treatment intensity. The relative effect of fuels treatment intensity ($R^2 = 0.05$–0.18) was lower than the effect of fuels treatment rate ($R^2 = 0.05$–0.28).

Discussion

Our simulations showed that a combination of fuels treatments and prescribed fire in southern Sierra Nevada conifer forests may reduce the severity and extent of fire across a large, heterogeneous landscape during a 50-year time span, particularly if weather conditions become more severe. In particular, simulations with fuels treatments resulted in lower mortality of large, old trees (as indicated by forest age; Fig. 7) and greater total landscape AGB compared with simulations with only fire and without treatment. In spite of these benefits, there is some concern about potentially negative ecological effects of fuels treatments, such as increased non-native plant abundance (e.g. Merriam *et al*. 2006) or effects to aquatic resources or wildlife (e.g. Rieman *et al*. 2003; Lehmkuhl *et al*. 2007). Owing to increasing occurrence of large, severe fires in the region's conifer forests (Miller *et al*. 2009), our model results suggest that fuels treatments may provide ecological benefits (e.g. preventing mortality of large, older trees) that offset potential localised loss of aboveground carbon (Hurteau and North 2009) or habitat (Lehmkuhl *et al*. 2007). The magnitude of treatment effects, however, may vary by elevation, type of treatment and fire regime.

The differences among elevations in the simulated fire rotation were consistent with the fire rotation from the last 20 years in the fire-history database, and the differences in all of the results among the fire regions in part reflect how fire regimes and fuel conditions vary by elevation in the Sierra Nevada (van Wagtendonk and Fites-Kaufman 2006). These differences also determined which parts of the landscape were treated, so one of the primary reasons that fuels treatments were most influential in the mid- and high-elevation regions, and not in the low-elevation region, was simply a function of where most of the treatments occurred. The low-elevation region had the largest proportion of chaparral and WUI, which is why the fire regime was characterised by shorter fire rotation

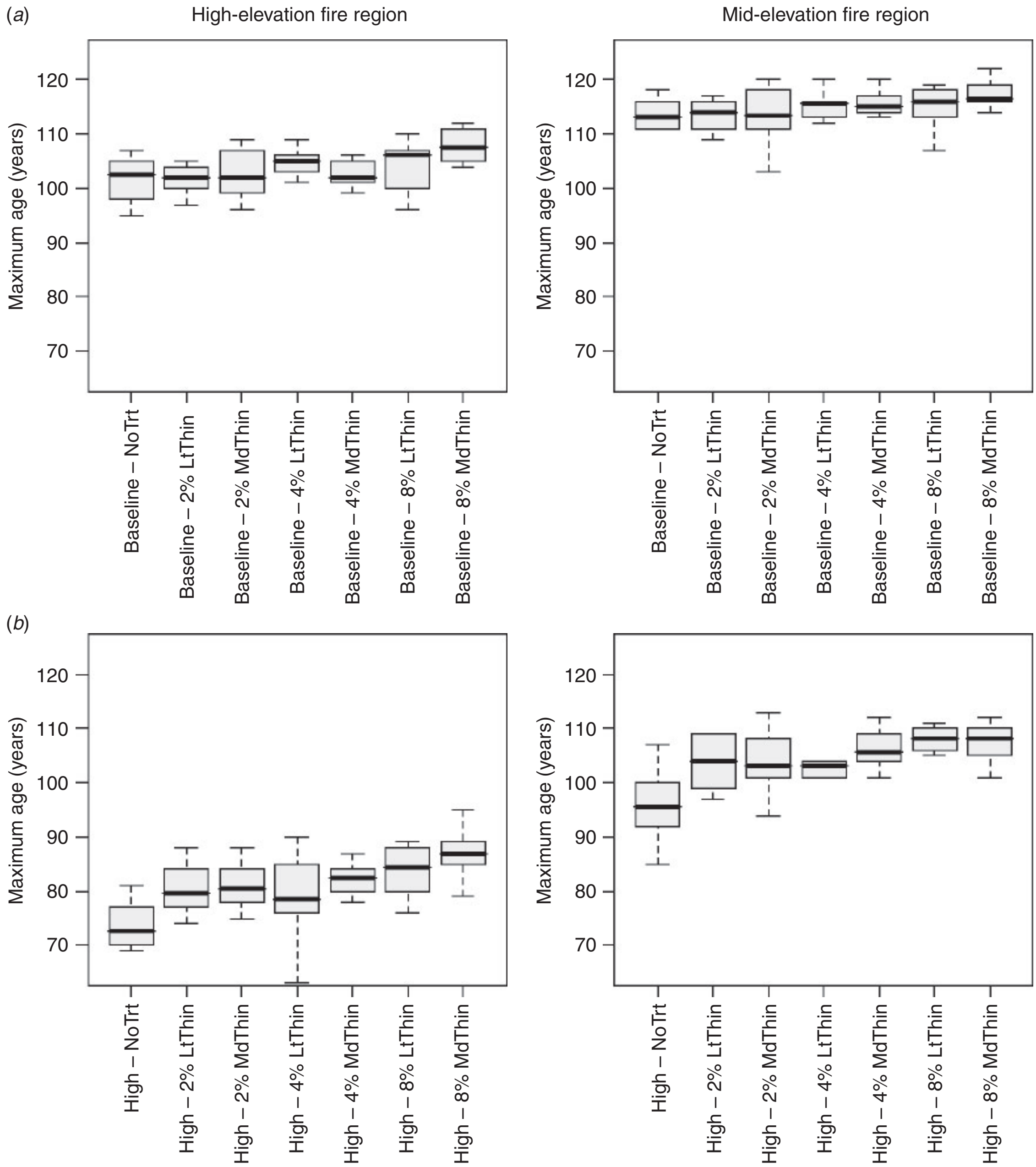

Fig. 7. Simulated mean forest age under the (*a*) baseline and (*b*) high fire regimes for two fire regions (high- and mid-elevation). Wildland–urban interface (WUI) fire regions not shown. Percentages represent the treatment rate (2, 4 and 8%), and LtThin and MdThin represent light and medium thinning intensity respectively. The whisker plots show the median (dark line); the box is defined by the quartiles; the dotted lines extend to the maximum and minimum.

with more frequent, smaller fires (to reflect higher human-caused ignition frequency, but more effective suppression in human-dominated areas) (Cardille *et al*. 2001; Sturtevant *et al*. 2004; Syphard *et al*. 2008). Although the chaparral and oak fuel types in the low elevation region had relatively rapid fire spread rates, there were few treatments in those areas to reduce the spread of fire.

The relationship between treatment effect and area of intersection between fires and treatments speaks to one of the concerns over fuels treatment efficacy: for treatments to be effective, they must intersect with fires that occur stochastically across space and time (Rhodes and Baker 2008). The strength of the LANDIS-II model is that it simulates the stochastic nature of fire and how the probability of fire is conditioned on multiple, interacting, dynamic processes (such as succession, weather, topography, disturbance history and stochastic ignitions) that vary over time across large, heterogeneous landscapes. Therefore, our results suggest that despite the stochastic nature of fire, fires intersect fuels treatments at a sufficient rate across the landscape to significantly alter the fire regime. Not surprisingly, treating a greater area (higher treatment rate) therefore increases the overall effect on the fire regime (Figs 7, 10).

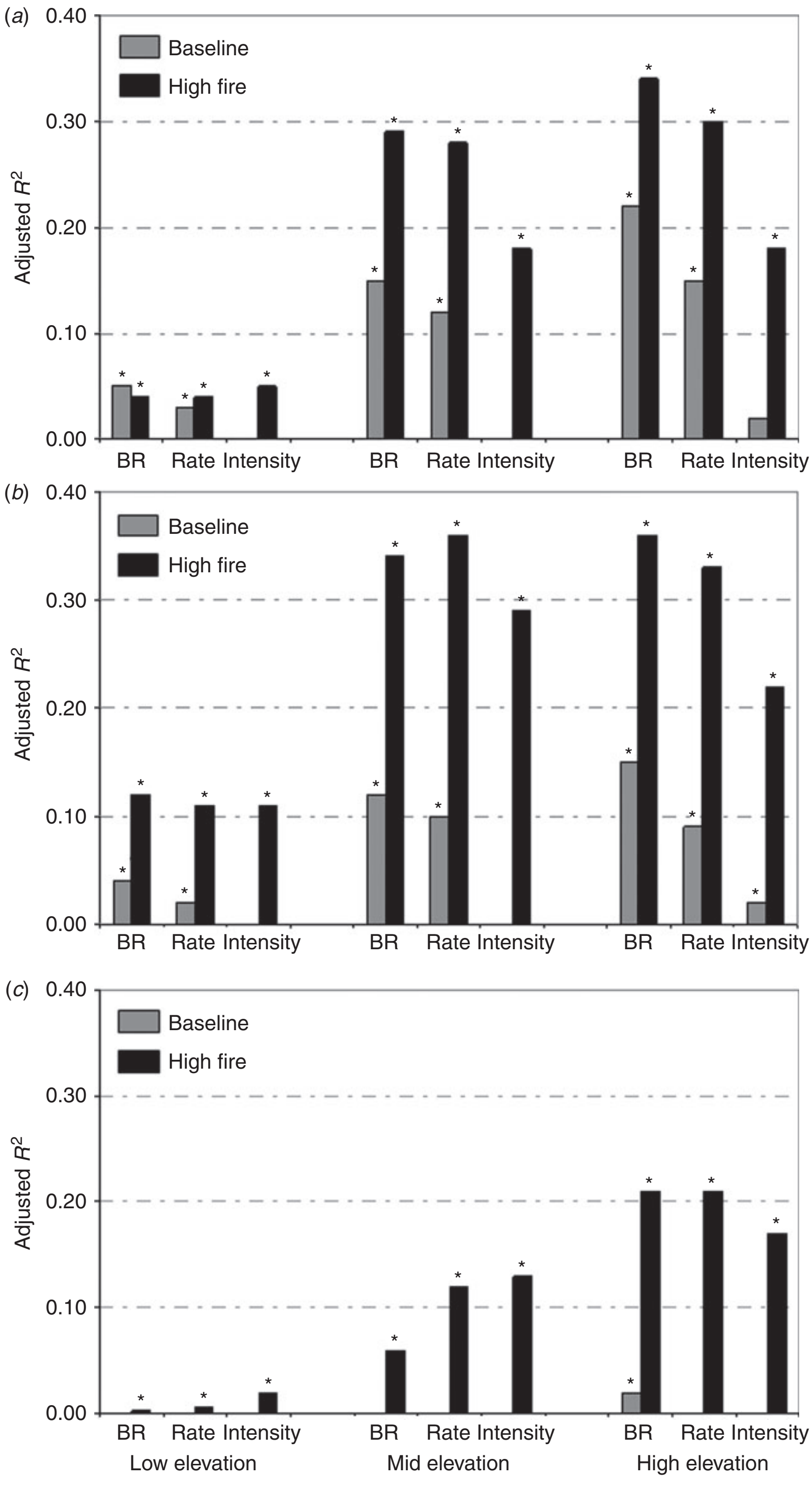

Fig. 8. Amount of variation explained (adjusted R^2) in: (*a*) Fire Rotation Period (FRP); (*b*) mean forest age; and (*c*) total aboveground biomass by total biomass removed (BR), treatment rate (Rate) and treatment intensity (Intensity) for low-, mid- and high-elevation fire regions. Asterisks denote significance of $P < 0.05$.

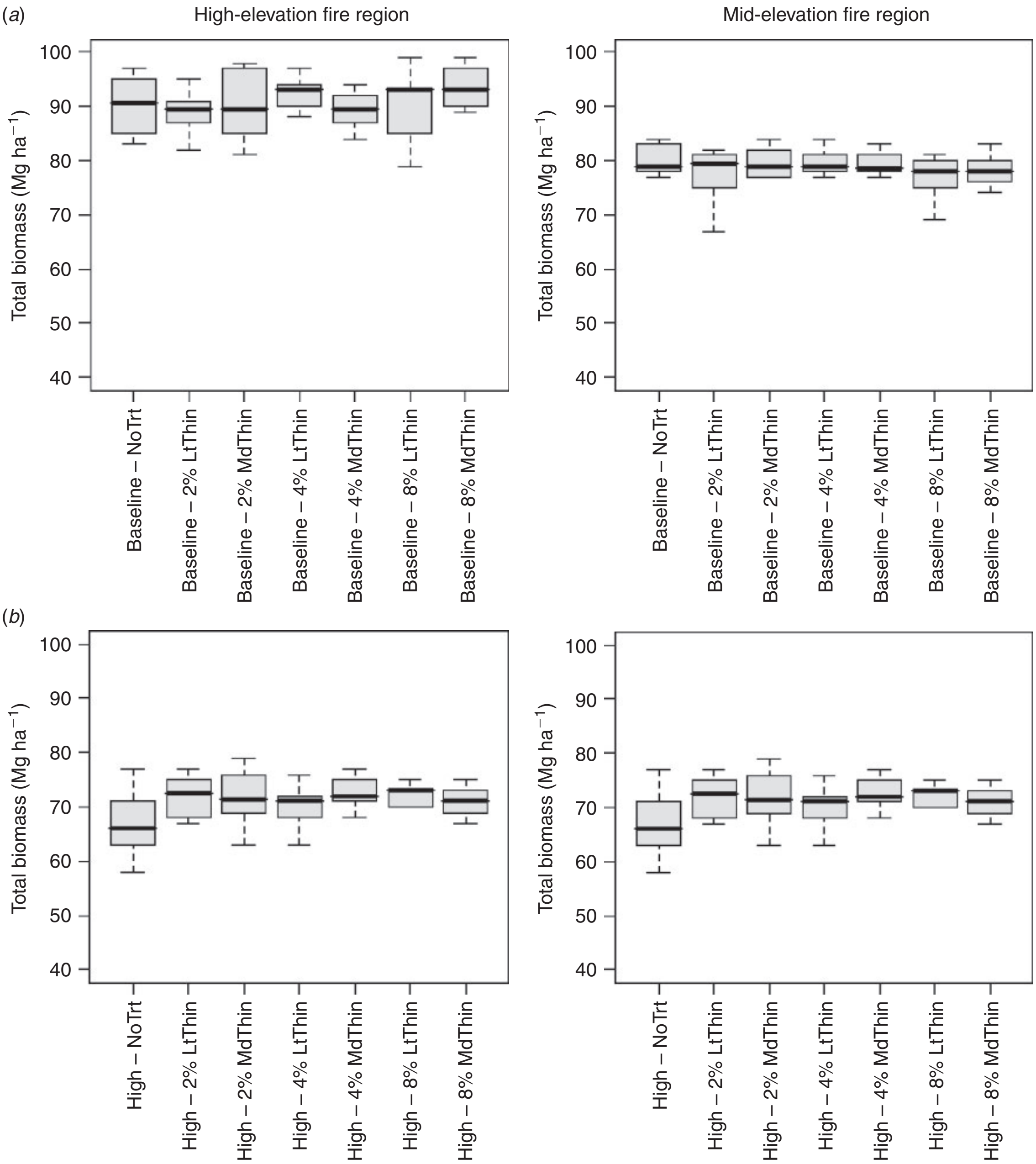

Fig. 9. Simulated total aboveground biomass under the (*a*) baseline and (*b*) high fire regimes for two fire regions (high- and mid-elevation). Wildland–urban interface (WUI) fire regions not shown. Percentages represent the treatment rate (2, 4 and 8%), and LtThin and MdThin represent light and medium thinning intensity respectively. The whisker plots show the median (dark line); the box is defined by the quartiles; the dotted lines extend to the maximum and minimum.

The maximum effect of treatment occurred with the greatest area treated (8% per 5 years) (equivalent to ~1/3 of the landscape over 20 years), consistent with previous modelling efforts (Finney *et al*. 2006). However, we cannot extrapolate beyond 8% to conclude that treatment effects would increase with even greater treatment rates. Although the lowest treatment rate (2% per 5 years) did have some effect in the mid- and high-elevation regions, the overall effect was not significant. Considering that the 2% rate approximates the current rate of treatment implemented by the US Forest Service, the results suggest that it may be important for forest managers to commit additional resources to expand the current scope of treatment.

In addition to the increased effect with treatment rate, the influence of treatment was much stronger under the high fire regime than the baseline because fires were generally larger under the high fire regime, and therefore the probability of fires encountering fuels treatments increased. Under the more severe conditions represented by the high fire regime, the rate of fire spread was generally higher; thus, a relatively greater reduction in fire spread and momentum occurred when these fast-moving fires encountered the fuels treatments. Although we did not

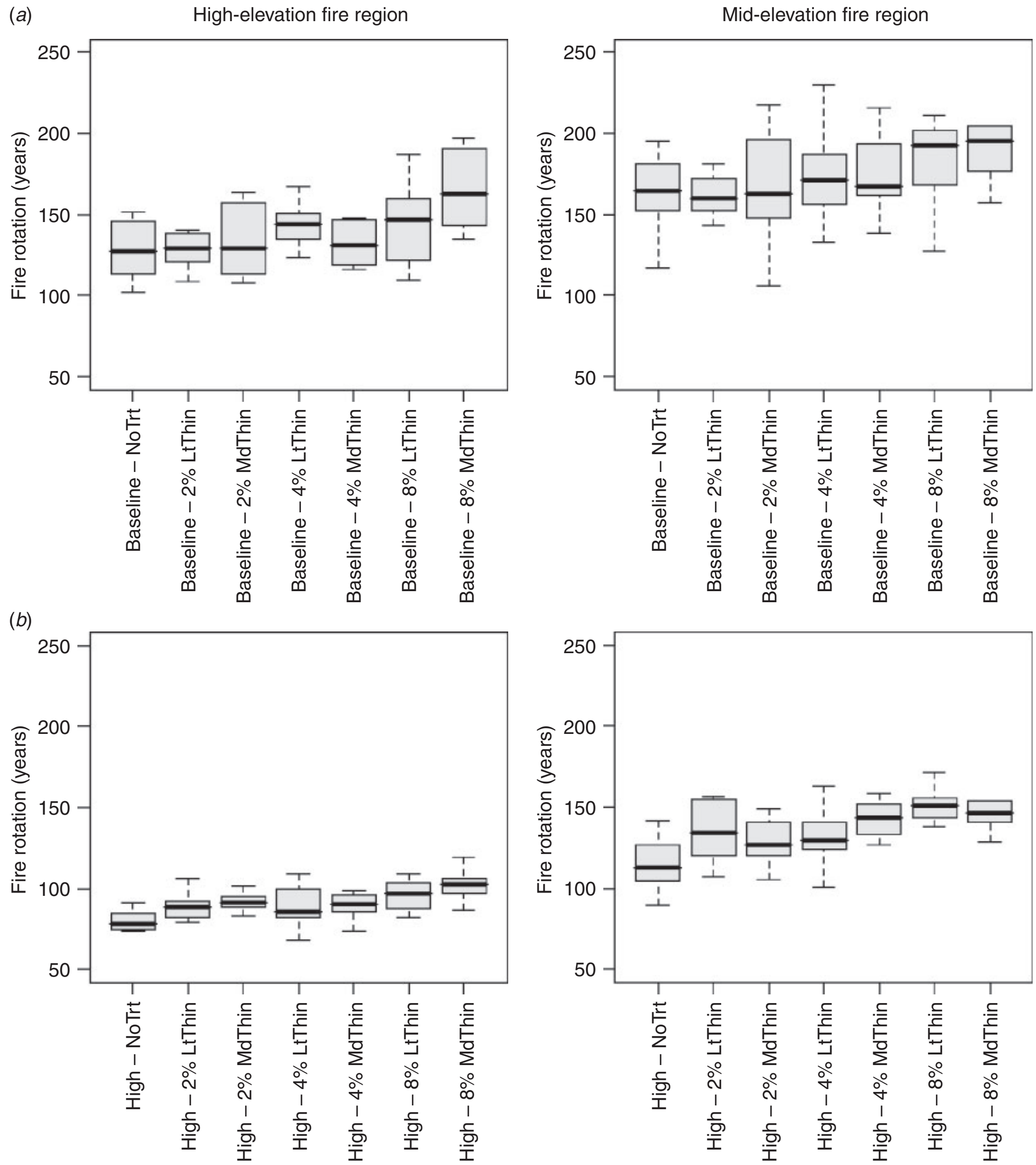

Fig. 10. Simulated fire rotation periods (FRPs) under the (*a*) baseline and (*b*) high fire regimes for two fire regions (high- and mid-elevation). Wildland–urban interface (WUI) fire regions not shown. Percentages represent the treatment rate (2, 4 and 8%), and LtThin and MdThin represent light and medium thinning intensity respectively. The whisker plots show the median (dark line); the box is defined by the quartiles; the dotted lines extend to the maximum and minimum.

intend to predict how weather may change when we restricted the database to reflect more extreme conditions, the response in fire rotation showed that fire was strongly influenced by weather in the simulations. Although there is considerable uncertainty regarding how climate may change in the future, and how fire may respond, the shorter fire rotation in the high fire regime was consistent with recent analyses that show increasing area burned in response to more severe weather (Calkin *et al*. 2005; Westerling *et al*. 2006). In the southern Sierra Nevada, fuels treatment may be more beneficial if weather conditions continue to become more severe and using the fuels treatment rates and intensities that we simulated.

Although treatment effect increased with rate, the overall influence of treatment intensity was insignificant in the baseline conditions, although it did explain significant variation in forest age, total AGB and fire rotation in the high fire regime. Nevertheless, our results do not strongly support the conclusion that higher-intensity treatments are more effective at reducing the extent and severity of fire than low-intensity treatments. Another consideration is that the range of treatment intensities tested was much smaller than the range of rates (medium-intensity treatments removed only ~20% more biomass than the low-intensity treatments, whereas the high treatment rate (8% per 5 years) was four times as large as the low treatment rate (2% per 5 years)).

One of the main objectives for implementing fuels treatments is to reduce fire intensity to improve the chance that older trees can survive fires (Peterson *et al*. 2005), which is why we evaluated mean forest age and total AGB at Year 50 relative to fuels treatments. We evaluated AGB in part because it is correlated with the presence of many large trees. Although fuels treatments actually remove biomass from the landscape, the simulation results showed that removing biomass as prescribed results in greater preservation of biomass over longer time scales. This seemingly counterintuitive result is because the biomass removed for treatments is young, dense understorey vegetation, whereas the bulk of total AGB is in the form of large, old trees (Hurteau and North 2009). The age of the forests was also much younger when no treatments were applied, which also shows that fuels treatments prevented substantial mortality of old trees.

Although our simulation results demonstrate a reduction in the magnitude of fire effects with fuels treatment, it is likely that a significant broad-scale effect would become most apparent over longer time scales than we simulated. The reason for such a potential time lag is that we parameterised the simulations to reflect current conditions, including high fuel accumulation due to fire suppression. Therefore, any fire that would initially occur on this landscape (that does not encounter a fuels treatment) would likely burn at a relatively high intensity and severity as compared with pre-suppression conditions. Assuming that fires in areas that have been treated, or have recently burned, would be relatively less intense owing to the loss of understorey fuels and ladder fuels, it would take many decades for enough treatments and fires to occur to substantially reduce the overall fuel conditions on the landscape to the point that the mean fire intensity was substantially reduced. A large-scale treatment effect would only become apparent after a substantial proportion of fires spread into areas that had been treated or had burned previously. Similarly, Scheller *et al*. (2005) found that fire severity did not substantially decline until after 40 years of simulated fire re-introduction in the Boundary Waters Canoe Area of Minnesota, USA.

Fuels treatments and prescribed fires are not designed to exclude fire from the landscape (which would further the paradigm of fire exclusion), but to slow the spread of fires and reduce their intensity by lowering fuel loads (Agee and Skinner 2005). The objective is to allow reintroduction of fire with moderated fire severity (or effects). Likewise, our treatment simulations were not designed to exclude fire and the simulated reduction in fire rotation from fuels treatments – even at the maximum treatment rate – still resulted in substantially more fire on the landscape than occurred during much of the 20th century. Ultimately, the objective of forest management is to restore the forest structure to conditions similar to those before fire suppression and to safely support a more natural fire regime of frequent, low-intensity fires (Miller and Urban 2000). Our simulations suggest that forest management activities, using a thin-from-below technique followed by prescribed fire, may help to further the goal of restoring forest conditions and fire regimes, and the ecological benefits of reducing the risk of large, severe fires may outweigh localised effects.

Another future consideration with respect to strategic treatment design is the determination of how and why fire patterns vary across the landscape (DellaSala *et al*. 2004). If treatments are placed in areas where there is a greater risk of fire, they will likely be more effective, as opposed to treating areas with lower probabilities of fire. Previous modelling studies have also shown that the spatial arrangement of treatments can increase their efficiency (e.g. Finney *et al*. 2006; Schmidt *et al*. 2008). There have been several approaches developed for mapping fire risk and probability using biophysical and climate variables (e.g. Preisler *et al*. 2004). Human influence on fire (i.e. ignitions and suppression patterns) may also be incorporated into probability maps (Syphard *et al*. 2008). Therefore, although our results support the differential effectiveness of treatment at different elevations and rates and in different fire regions, we also suggest that future management consider how treatment efficiency could be further maximised by focussing on areas that have a relatively high probability of experiencing severe fire.

The applicability of our results is, by design, limited to the landscape scale. Although many models encapsulate greater complexity for individual disturbance events or management activities, the additional resources (data inputs, calibration data, computational) required were not justified given our landscape-scale hypotheses. Furthermore, any additional precision gained through increased model complexity would have been overwhelmed by the inherent uncertainty of a system dominated by highly stochastic wildfire and increasing human activity and intervention (Milne *et al*. 2009).

Acknowledgements

Funding for this project was provided by the Pacific Southwest Region of the US Forest Service (Region 5), and we thank C. Knopp, D. Macfarlane, B. Pendleton and A. Gaffrey for their support. We are also grateful for the expert advice and guidance from many colleagues and regional experts, including: D. Yasuda, B. Bahro, K. Barber, J. Sherlock, H. Safford, S. Anderson, M. Smith, D. Romberger, J. Whitfield, J. Bridges, K. Clement, T. Terrell, J. Quinn, S. Parks, R. Rojas, J. Fites and J. Miller. We also thank our science advisors, W. Zielinski, F. Davis, R. Barrett, C. Skinner, S. Stephens, J. Vankat, J. van Wagtendonk, D. Graber, K. Aubry, B. Heald, M. North and D. Mladenoff, for reviewing our technical work.

References

Agee JK (1993) 'Fire Ecology of Pacific Northwest Forests.' (Island Press: Washington, DC)

Agee JK, Skinner CN (2005) Basic principles of forest fuel reduction treatments. *Forest Ecology and Management* **211**, 83–96. doi:10.1016/J.FORECO.2005.01.034

Agee JK, Bahro B, Finney MA, Om PN, Sapsis DB, Skinner CN, van Wagtendonk JW, Weatherspoon CP (2000) The use of shaded fuelbreaks in landscape fire management. *Forest Ecology and Management* **127**, 55–66. doi:10.1016/S0378-1127(99)00116-4

Amiro BD, Logan KA, Wotton BM, Flannigan MD, Todd JB, Stocks BJ, Martell DL (2004) Fire Weather Index System components for large fires in the Canadian boreal forest. *International Journal of Wildland Fire* **13**, 391–400. doi:10.1071/WF03066

Backer DM, Jensen SE, McPherson GR (2004) Impacts of fire suppression activities on natural communities. *Conservation Biology* **18**, 937–946. doi:10.1111/J.1523-1739.2004.494_1.X

Beaty RM, Taylor AH (2008) Fire history and the structure and dynamics of a mixed conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. *Forest Ecology and Management* **255**, 707–719. doi:10.1016/J.FORECO.2007.09.044

Bond WJ, van Wilgen BW (1996) 'Fire and Plants.' (Chapman and Hall: London)

Burns RM, Honkala BH (1990) Silvics of North America: 1. Conifers; 2. Hardwoods. USDA Forest Service, Agriculture Handbook 654. (Washington, DC)

Calkin DE, Gebert KM, Jones JG, Neilson RP (2005) Forest Service large fire area burned and suppression expenditure trends, 1970–2002. *Journal of Forestry* **103**, 179–183.

Cardille JA, Ventura SJ, Turner MG (2001) Environmental and social factors influencing wildfires in the Upper Midwest, United States. *Ecological Applications* **11**, 111–127. doi:10.1890/1051-0761(2001)011[0111:EASFIW]2.0.CO;2

Clark JS, Carpenter SR, Barber M, Collins S, Dobson A, Foley JA, Lodge DM, Pascual M, Pielke R, Jr, Pizer W, Pringle C, Reid WV, Rose KA, Sala O, Schlesinger WH, Wall DH, Wear D (2001) Ecological forecasts: an emerging imperative. *Science* **293**, 657–660. doi:10.1126/SCIENCE.293.5530.657

Collins BM, Stephens SL (2007) Managing natural wildfires in Sierra Nevada wilderness areas. *Frontiers in Ecology and the Environment* **5**, 523–527. doi:10.1890/070007

Collins BM, Miller JD, Thode AE, Kelly M, van Wagtendonk JW, Stephens SL (2009) Interactions among wildland fires in a long-established Sierra Nevada natural fire area. *Ecosystems* **12**, 114–128. doi:10.1007/S10021-008-9211-7

Covington WW, Moore MM (1994) South-western ponderosa forest structure: changes since Euro-American settlement. *Journal of Forestry* **95**, 39–47.

DellaSala DA, Williams JE, Williams CD, Franklin JF (2004) Beyond smoke and mirrors: a synthesis of fire policy and science. *Conservation Biology* **18**, 976–986. doi:10.1111/J.1523-1739.2004.00529.X

Dixon GE (2002) Essential FVS: a user's guide to the Forest Vegetation Simulator. USDA Forest Service, Forest Management Service Center, Internal Report. (Fort Collins, CO)

Finney MA, McHugh CW, Grenfell IC (2005) Stand and landscape effects of prescribed burning on two Arizona wildfires. *Canadian Journal of Forest Research* **35**, 1714–1722. doi:10.1139/X05-090

Finney MA, Seli RC, McHugh CW, Ager AA, Bahro B, Agee JK (2006) Simulation of long-term landscape-level fuel treatment effects on large wildfires. USDA Forest Service, Rocky Mountain Research Station, Proceedings RMRS-P-41. (Fort Collins, CO)

Flannigan MD, Logan KA, Amiro BD, Skinner WR, Stocks BJ (2005) Future area burned in Canada. *Climatic Change* **72**, 1–16. doi:10.1007/S10584-005-5935-Y

Forestry Canada Fire Danger Group (1992) Development and structure of the Canadian Forest Fire Behavior Prediction System. Forestry Canada, Science and Sustainable Development Directorate, Information Report ST-X-3. (Ottawa, ON)

Franklin J (2003) Clustering versus regression trees for determining ecological land units in the southern California mountains and foothills. *Forest Science* **49**, 354–368.

Gavin DG, Hallett DJ, Hu FS, Lertzman KP, Prichard SJ, Brown KJ, Lynch JA, Bartlein P, Peterson DL (2007) Forest fire and climate change in western North America: insights from sediment charcoal records. *Frontiers in Ecology and the Environment* **5**, 499–506. doi:10.1890/060161

Gustafson EJ, Shvidenko AZ, Sturtevant BR, Scheller RM (2010) Predicting global change effects on forest biomass and composition in south-central Siberia. *Ecological Applications* **20**, 700–715. doi:10.1890/08-1693.1

Hansen MH, Frieswyk T, Glover JF, Kelly JF (1992) The Eastwide forest inventory data base: users' manual. USDA Forest Service, North Central Forest Experiment Station, General Technical Report GTR NC-151. (St Paul, MN)

Hurteau M, North M (2009) Fuel treatment effects on tree-based carbon storage and emissions under modeled wildfire scenarios. *Frontiers in Ecology and the Environment* **7**, 409–414. doi:10.1890/080049

Hurteau MD, Koch GW, Hugate BA (2008) Carbon protection and fire risk reduction: toward a full accounting of forest carbon offsets. *Frontiers in Ecology and the Environment* **6**, 493–498. doi:10.1890/070187

Keane RE, Veblen T, Ryan KC, Logan J, Allen C, Hawkes B (2002). The cascading effects of fire exclusion in the Rocky Mountains. In 'Rocky Mountain Futures: an Ecological Perspective'. (Ed. J Baron) pp. 133–153. (Island Press: Washington, DC)

Keeley JE, Davis FW (2007) Chaparral. In 'Terrestrial Vegetation of California, 3rd Edn'. (Eds MG Barbour, T Keeler-Wolf, AA Schoenherr) pp. 339–366. (University of California Press: Los Angeles, CA)

Keeley JE, Stephenson NL (2000) Restoring natural fire regimes to the Sierra Nevada in an era of global change In 'Wilderness Ecosystems, Threats, and Management'. (Eds DN Cole, SF McCool, WT Borrie, J O'Laughlin) USDA Forest Service, Rocky Mountain Research Station, Proceedings RMRS-P-15-VOL-5. (Ogden, UT)

Kilgore BM, Taylor D (1979) Fire history of a sequoia mixed conifer forest. *Ecology* **60**, 129–142. doi:10.2307/1936475

Laudenslayer WF, Darr HH (1990) Historical effects of logging on the forests of the Cascade and Sierra Nevada ranges of California. *Transactions of the Western Section of the Wildlife Society* **26**, 12–23.

Lehmkuhl JF, Kennedy M, Ford ED, Singleton PH, Gaines WL, Lind RL (2007) Seeing the forest for the fuel: integrating ecological values and fuels management. *Forest Ecology and Management* **246**, 73–80. doi:10.1016/J.FORECO.2007.03.071

Lenihan JM, Drapek RJ, Bachelet D, Neilson RP (2003) Climate change effects on vegetation distribution, carbon, and fire in California. *Ecological Applications* **13**, 1667–1681. doi:10.1890/025295

Lutz JA, van Wagtendonk JW, Thode AE, Miller JD, Franklin JF (2009) Climate, lightning ignitions, and fire severity in Yosemite National Park, California, USA. *International Journal of Wildland Fire* **18**, 765–774. doi:10.1071/WF08117

Mayer KE, Laudenslayer WF, Jr (1988) 'A Guide to Wildlife Habitats of California.' (State of California, Resources Agency: Sacramento, CA)

McKenzie D, Gedalof Z, Peterson DL, Mote P (2004) Climatic change, wildfire, and conservation. *Conservation Biology* **18**, 890–902. doi:10.1111/J.1523-1739.2004.00492.X

Merriam KE, Keeley JE, Beyers JL (2006) Fuel breaks affect non-native species abundance in California plant communities. *Ecological Applications* **16**, 515–527. doi:10.1890/1051-0761(2006)016[0515:FBANSA]2.0.CO;2

Miller C, Landres PB (2004) Exploring information needs for wildland fire and fuels management. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-127. (Fort Collins, CO)

Miller C, Urban DL (2000) Modeling the effects of fire management alternatives on mixed-conifer forests in the Sierra Nevada. *Ecological Applications* **10**, 85–94. doi:10.1890/1051-0761(2000)010[0085:MTEOFM]2.0.CO;2

Miller JD, Safford HD, Crimmins M, Thode AE (2009) Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. *Ecosystems* **12**, 16–32. doi:10.1007/S10021-008-9201-9

Milne E, Aspinall RJ, Veldkamp TA (2009) Integrated modelling of natural and social systems in land change science. *Landscape Ecology* **24**, 1145–1147. doi:10.1007/S10980-009-9392-2

Mladenoff DJ (2004) LANDIS and forest landscape models. *Ecological Modelling* **180**, 7–19. doi:10.1016/J.ECOLMODEL.2004.03.016

Noss RF, Franklin JF, Baker WL, Schoennagel T, Moyle PB (2006) Managing fire-prone forests in the western United States. *Frontiers in Ecology and the Environment* **4**, 481–487. doi:10.1890/1540-9295(2006)4[481:MFFITW]2.0.CO;2

Parsons DJ, Graber DM, Agee JK, van Wagtendonk JW (1986) Natural fire management in National Parks. *Environmental Management* **10**, 21–24. doi:10.1007/BF01866414

Peterson DL, Johnson MC, Agee JK, Jain TB, McKenzie DM, Reinhardt ER (2005) Forest structure and fire hazard in dry forests of the Western United States. USDA Forest Service, Pacific Northwest Research Station, General Technical Report PNW-GTR-628. (Portland, OR)

Preisler HK, Brillinger DR, Burgan RE, Benoit JW (2004) Probability based models for estimation of wildfire risk. *International Journal of Wildland Fire* **13**, 133–142. doi:10.1071/WF02061

R Development Core Team (2004) 'R: A Language and Environment for Statistical Computing.' (R Foundation for Statistical

Computing) Available at http://www.r-project.org/ [Verified 7 April 2011]

Radeloff VC, Hammer RB, Stewart SI, Fried JS, Holcomb SS, McKeefry JF (2005) The wildland–urban interface in the United States. *Ecological Applications* **15**, 799–805. doi:10.1890/04-1413

Raymond CL, Peterson DL (2005) Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. *Canadian Journal of Forest Research* **35**, 2981–2995. doi:10.1139/X05-206

Reinhardt ED, Keane RE, Calkin DE, Cohen JD (2008) Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. *Forest Ecology and Management* **256**, 1997–2006. doi:10.1016/J.FORECO.2008.09.016

Rhodes JJ, Baker WL (2008) Fire probability, fuel treatment effectiveness and ecological tradeoffs in western US forest. *The Open Forest Science Journal* **1**, 1–7. doi:10.2174/1874398600801010001

Rieman BE, Luce CH, Gresswell RE, Young MK (2003) Introduction to the effects of wildland fire on aquatic ecosystems in the western USA. *Forest Ecology and Management* **178**, 197–211. doi:10.1016/S0378-1127(03)00062-8

Roberts DW (1996) Landscape vegetation modelling with vital attributes and fuzzy systems theory. *Ecological Modelling* **90**, 175–184. doi:10.1016/0304-3800(95)00164-6

Scheller RM, Mladenoff DJ (2004) A forest growth and biomass module for a landscape simulation model, LANDIS: design, validation, and application *Ecological Modelling* **180**, 211–229. doi:10.1016/J.ECOLMODEL.2004.01.022

Scheller RM, Mladenoff DJ, Crow TR, Sickley TS (2005) Simulating the effects of fire reintroduction versus continued suppression on forest composition and landscape structure in the Boundary Waters Canoe Area, northern Minnesota (USA). *Ecosystems* **8**, 396–411. doi:10.1007/S10021-003-0087-2

Scheller RM, Domingo JB, Sturtevant BR, Williams JS, Rudy A, Gustafson EJ, Mladenoff DJ (2007) Design, development, and application of LANDIS-II, a spatial landscape simulation model with flexible spatial and temporal resolution. *Ecological Modelling* **201**, 409–419. doi:10.1016/J.ECOLMODEL.2006.10.009

Scheller RM, Van Tuyl S, Clark K, Hayden NG, Hom J, Mladenoff DJ (2008) Simulation of forest change in the New Jersey Pine Barrens under current and pre-colonial conditions. *Forest Ecology and Management* **255**, 1489–1500. doi:10.1016/J.FORECO.2007.11.025

Scheller RM, Gustafson EJ, Sturtevant BR, Ward BC, Mladenoff DJ (2010) Increasing the reliability of ecological models using modern software engineering techniques. *Frontiers in Ecology and the Environment* **8**(5), 253–260. doi:10.1890/080141

Schmidt DA, Taylor AH, Skinner CN (2008) The influence of fuels treatment and landscape arrangement on simulated fire behavior, Southern Cascade Range, California. *Forest Ecology and Management* **255**, 3170–3184. doi:10.1016/J.FORECO.2008.01.023

Schoennagel T, Veblen TT, Romme WH (2004) The interaction of fire, fuels, and climate across Rocky Mountain forests. *Bioscience* **54**, 661–676. doi:10.1641/0006-3568(2004)054[0661:TIOFFA]2.0.CO;2

Scott JH, Burgan RE (2005) Standard fire behavior fuel models: a comprehensive set for use with Rothermel's Surface Fire Spread Model. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-153. (Fort Collins, CO)

Spencer W, Rustigian-Romsos H, Strittholt J, Scheller R, Zielinski W, Truex R (2011) Using occupancy and population models to assess habitat conservation opportunities for an isolated carnivore population. *Biological Conservation* **144**, 788–803. doi:10.1016/J.BIOCON.2010.10.027

Stephens SL (1998) Effects of fuels and silvicultural treatments on potential fire behavior in mixed conifer forests of the Sierra Nevada, CA. *Forest Ecology and Management* **105**, 21–35. doi:10.1016/S0378-1127(97)00293-4

Stephens SL, Moghaddas JJ (2005) Experiemental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. *Forest Ecology and Management* **215**, 21–36. doi:10.1016/J.FORECO.2005.03.070

Stephens SL, Moghaddas JJ, Ediminster C, Fiedler CE, Hasse S, Harrington M, Keeley JE, McIver JD, Metlen K, Skinner CN, Youngblood A (2009) Fire treatment effects on vegetation structure, fuels, and potential fire severity in western US forests. *Ecological Applications* **19**, 305–320. doi:10.1890/07-1755.1

Stratton RL (2004) Assessing the effectiveness of landscape fuel treatments on fire growth and behavior. *Journal of Forestry* **102**, 32–40.

Sturtevant BR, Gustafson EJ, Li W, He HS (2004) Modeling biological disturbances in LANDIS: a module description and demonstration using spruce budworm. *Ecological Modelling* **180**, 153–174. doi:10.1016/J.ECOLMODEL.2004.01.021

Sturtevant BR, Scheller RM, Miranda BR, Shinneman D, Syphard AD (2009) Simulating dynamic and mixed-severity fire regimes: A process-based fire extension for LANDIS-II. *Ecological Modelling* **220**, 3380–3393. doi:10.1016/J.ECOLMODEL.2009.07.030

Sugihara NG, Van Wagtendonk JW, Fites-Kaufman J, Shaffer KE, Thode AE (2006) The future of fire in California's ecosystems. In 'Fire in California's Ecosystems'. (Eds NG Sugihara, JW van Wagtendonk, KE Shaffer, J Fites-Kaufman, AE Thode) pp. 538–543. (The University of California Press: Berkeley, CA)

Syphard AD, Franklin J, Keeley JE (2006) Simulating the effects of frequent fire on southern California coastal shrublands. *Ecological Applications* **16**, 1744–1756. doi:10.1890/1051-0761(2006)016[1744:STEOFF]2.0.CO;2

Syphard AD, Yang J, Franklin J, He HS, Keeley JE (2007) Calibrating a forest landscape model to simulate high fire frequency in Mediterranean-type shrublands. *Environmental Modelling & Software* **22**, 1641–1653. doi:10.1016/J.ENVSOFT.2007.01.004

Syphard AD, Radeloff VC, Keuler NS, Taylor RS, Hawbaker TJ, Stewart SI, Clayton MK (2008) Predicting spatial patterns of fire in a southern California landscape. *International Journal of Wildland Fire* **17**, 602–613. doi:10.1071/WF07087

US Census (2000) Census 2000 TIGER/Line Files. US Census Bureau, machine-readable data files. (Washington, DC) Available at http://www.census.gov/geo/www/tiger/ [Verified 19 April 2011]

Van Wagner CE (1987) Development and structure of the Canadian Forest Fire Weather Index System. Agriculture Canada, Canadian Forest Service, Forestry Technical Report 35. (Ottawa, ON)

van Wagtendonk JW, Fites-Kaufman J (2006) Sierra Nevada Bioregion. In 'Fire in California's Ecosystems'. (Eds NG Sugihara, JW van Wagtendonk, KE Shaffer, J Fites-Kaufman, AE Thode) pp. 264–294. (The University of California Press: Berkeley, CA)

Varner JM, Gordon DR, Putz FE, Hiers JK (2005) Restoring fire to long-unburned *Pinus palustris* ecosystems: novel fire effects and consequences for long-unburned ecosystems. *Restoration Ecology* **13**, 536–544. doi:10.1111/J.1526-100X.2005.00067.X

Ward BC, Mladenoff DJ, Scheller RM (2005) Landscape-level effects of the interaction between residential development and public forest management in northern Wisconsin, USA. *Forest Science* **51**, 616–632.

Westerling AL, Hidalgo HG, Cayan DR, Swetnam TW (2006) Warming and earlier spring increase western US forest wildfire activity. *Science* **313**, 940–943. doi:10.1126/SCIENCE.1128834

Xu C, Gertner GZ, Scheller RM (2007) Potential effects of interaction between CO_2 and temperature on Boundary Water Canoe Area's forest landscape response to global warming. *Global Change Biology* **13**, 1469–1483. doi:10.1111/J.1365-2486.2007.01387.X

Manuscript received 2 November 2009, accepted 3 August 2010