Evaluating potential trade-offs among fuel treatment strategies in mixed-conifer forests of the Sierra Nevada

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Abstract. Fuel treatments in fire-suppressed mixed-conifer forests are designed to moderate potential wildfire behavior and effects. However, the objectives for modifying potential fire effects can vary widely, from improving fire suppression efforts and protecting infrastructure, to reintroducing low-severity fire, to restoring and maintaining variable forest structure and wildlife habitat. In designing a fuel treatment, managers can alter the treatment's prescription, placement, and extent (collectively the "treatment strategy") to optimally meet one objective. However, the potential for trade-offs among different objectives is rarely tested systematically in fire-prone landscapes. To evaluate trade-offs in mechanical fuel treatment objectives related to fire severity, smoke production, forest heterogeneity, and avian wildlife habitat, we used a cross-platform modeling approach based on spatially explicit modifications of forest structure data for a 7820-ha landscape in the Lake Tahoe Basin, California. We examine whether (1) a more uniform treatment strategy aimed at fire hazard reduction (FHR) had negative effects on wildlife diversity, (2) a strategy focused on protecting the wildland–urban interface (WUI) left other portions of the landscape vulnerable to high-severity fire, and (3) increasing the extent of fuel treatments across the landscape led to greater reductions in fire severity and smoke emissions. When approximately 13% of the landscape was treated, the proportion of the landscape vulnerable to high-severity fire decreased by 13–44%, with the more uniform FHR strategy leading to greater reductions. Slight increases in predicted avian species richness that followed all treatment strategies were not closely linked to increases in canopy variability. The WUI protection strategy led to considerable reductions in fire severity at the landscape scale. Increasing the extent of treatments to 30% of the landscape did little to further reduce the area vulnerable to high-severity fire, with additional reductions of 4–7% depending on the prescription. However, increasing the extent of treatments reduced the extent of harmful downwind smoke impacts, primarily by reducing rate of fire spread. Treatment strategies will depend on specific management objectives, but we illustrate that trade-offs are not necessarily inherent in general outcomes of fuel treatments.

Key words: fire severity; forest management; fuel treatments; Lake Tahoe Basin; particulate matter; smoke; trade-offs; wildlife habitat.

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INTRODUCTION

A century of fire suppression in drier mixed-conifer forests of western North America, where fire was historically much more frequent, has led to increases in fuel and forest density (Safford and Stevens, *in press*). These altered conditions have increased the risk of undesirable social and ecological effects of modern-day wildfires that often burn at high intensity over large areas (Hessburg et al. 2016). Undesirable effects of wildfire in fire-suppressed forests include the loss of human life and property to fire (Calkin et al. 2014, Stephens et al. 2014*b*), public health impacts of smoke (Stephens et al. 2014*b*), and large contiguous areas of forest burned at high severity with near-complete tree mortality (Lydersen et al. 2014, Stephens et al. 2014*b*, Miller and Quayle 2015).

In response to the risks posed by increased fuel and forest density, contemporary forest management policy promotes the application of fuel reduction activities ("fuel treatments") in fire-prone forests (Schultz et al. 2012). While the application of fire itself may constitute a fuel treatment, treatments often involve some use of mechanical thinning and manipulation of surface fuels (Agee and Skinner 2005). There are presently many strategies for the design and placement of mechanical fuel treatments across landscapes (Finney 2007, North et al. 2009), with optimal strategies depending on the objectives of the fuel treatment. While most fuel treatment strategies have objectives involving the modification of potential fire behavior, there are different reasons for seeking to modify fire behavior (Collins et al. 2010, Calkin et al. 2014). For example, fuel treatment objectives can include the following: (1) slowing fire spread, particularly within the wildland–urban interface (WUI), and enabling effective suppression of the fire to protect infrastructure (Winter et al. 2002); (2) reducing adverse health impacts of smoke from uncontrolled wildfires (North et al. 2012*b*, Jaffe et al. 2013); (3) promoting low-severity wildfire across the landscape to preserve large legacy trees, carbon storage, and seed sources for reforestation (Reinhardt et al. 2008, Loudermilk et al. 2014); and (4) restoring the variable forest structure associated with a frequent fire regime, which often involves increasing the heterogeneity of canopy cover within the treatment and modifying habitat quality for wildlife (Lydersen and North 2012, Churchill et al. 2013, Sollman et al. 2016).

Given the range of mechanical fuel treatment objectives, managers must define primary treatment objectives to guide decisions regarding those components of treatments over which they have control: namely prescription type, placement, and extent of treatments. Together, these three components constitute a "treatment strategy." Treatment prescriptions guide decisions about where and how much to reduce fuels within a treatment unit. For example, prescriptions can call for thinning some or all trees below a target diameter or thinning to a target canopy cover across the landscape, if the objective is to facilitate fire suppression within the treatment (Agee and Skinner 2005). Prescriptions can also apply different treatment intensities to different portions of the landscape if the objective is to restore historical forest structure (North et al. 2009, Churchill et al. 2013). Treatment placement and extent refer to the specific locations of, and area covered by, fuel treatments across a landscape (Finney 2007). For example, treatments can be placed within the WUI if the objective is to facilitate fire suppression around infrastructure (which also depends on reducing the vulnerability of structures within the WUI; Calkin et al. 2014), within areas with the greatest fuel accumulation if the objective is to promote low-severity wildfire (Reinhardt et al. 2008), or within areas where forests are most departed from their historical condition if the objective is structural restoration (North et al. 2009). If the objective is more than fire containment, treatment extent may need to be greater when attempting to reduce potential smoke emissions, increase the predominance of low-severity fire, or accomplish landscape-scale restoration (Schultz et al. 2012). However, the extent of mechanical fuel treatments is often constrained by financial costs, administrative boundaries, steep topography, and distance to existing road networks (North et al. 2015*a*).

There are many factors in play when deciding treatment prescription, placement, and extent, and these choices entail numerous potential trade-offs (Vogler et al. 2015). Specifically, designing a treatment strategy to maximize one primary objective may have less optimal or even negative effects on other treatment objectives. For example, at the stand scale, treatment prescriptions

that uniformly reduce fuels to very low levels in high-risk areas (e.g., shaded fuelbreaks or defensible fuel profile zones; Agee et al. 2000) can reduce the severity and smoke produced by subsequent wildfires and facilitate fire suppression, but may also reduce desirable structural heterogeneity and associated habitat diversity within the treated area (Tempel et al. 2015). Conversely, treatment prescriptions that restore stand-scale heterogeneity associated with historical conditions may be less effective at reducing fire severity and smoke impacts within the treatment.

At the landscape scale, treatment placements that target the WUI may leave other areas of the landscape vulnerable to negative ecological effects of uncharacteristic high-severity wildfire, particularly if much of the landscape is in a high-fuel condition (Ager et al. 2010). Alternative treatment placements can target high-fuel areas in order to promote low-severity fire, or areas that are highly departed from their historical condition to promote structural restoration, but these treatment placements can then leave the WUI vulnerable. The trade-offs associated with treatment placement will generally depend on the spatial alignment of high-fuel load areas and high departure areas with each other and with the WUI (Ager et al. 2010).

Trade-offs relating to treatment placement may arise because it is often impractical to mechanically treat the entire extent of a landscape that may be available for mechanical treatment (North et al. 2012*b*, Long et al. 2014), forcing managers to make decisions on optimal treatment placement (Collins et al. 2010, Stephens et al. 2014*a*). Given these potential trade-offs, there is substantial manager interest in weighing the costs and benefits of different treatment strategies that vary prescription, placement, and extent within a given landscape (Long et al. 2014). Mechanically thinning a subset of the available landscape might allow more efficient use of funding; circumvent numerous legal, operational, and administrative constraints (North et al. 2015*a*); and avoid impacts to sensitive wildlife species; however, the potential trade-offs associated with such a strategy have not been systematically tested (Long et al. 2014).

In this article, we present a case-study analysis of a representative watershed in the western Lake Tahoe Basin, California, to illustrate a method for evaluating trade-offs among different treatment strategies with different objectives. Our approach integrates several modeling platforms that can analyze standard spatial data sets commonly used to guide forest management decisions. Fire behavior modeling software allows comparison of predicted fire behavior patterns under alternative treatment strategies, based on satellite-derived fuels and forest structure data that can be modified in different ways to simulate fuel treatments (Finney 2004, Schmidt et al. 2008, Vaillant et al. 2009). Emissions modeling can be used to modify forest structure data to compare treatment effects on smoke production from subsequent wildfires (Larkin et al. 2009, Ottmar 2014). Wildlife habitat models can predict occupancy probability for multiple species in response to fuel treatments using similar forest structure data (White et al. 2013*b*). Advances in forest reconstructions at the landscape scale are improving our ability to target restoration efforts in areas that are most departed from their historical range of variability (Taylor et al. 2014). Collectively, this increase in data availability and modeling capacity enables landscape-scale case studies to investigate the dynamics of fuel treatment trade-offs.

By linking these modeling platforms to a single landscape using the same underlying data, we examine the costs and benefits of alternative management approaches to fuel treatments. Specifically, we simulated different treatment strategies representing a combination of different prescriptions, placements, and extents (Table 1) and modeled their effects on subsequent fire behavior, smoke production, and wildlife habitat. For each strategy, we evaluated how well it met its stated objective(s) in comparison with other strategies (Table 1), by evaluating the proportional change in the relevant variables. We also evaluated a set of hypothesized trade-offs for each treatment strategy (Table 1). Our intent is to present a framework for systematically comparing different treatment strategies that might be used elsewhere in the western United States to improve fuel treatment outcomes.

Methods

Study area

Our study area is a 7820-ha area hydrologic unit (Ward Creek-Frontal Lake Tahoe #1605010 10403, hereafter "the landscape") that includes

Table 1. Summary of treatment strategies, objectives, and possible trade-offs.

Characteristic	Treatment strategy				
	WUI	FHR-sub	FHR-max	RES-sub	RES-max
Treatment prescription	Uniform	Uniform	Uniform	Variable	Variable
Treatment placement	WUI	Highest flame lengths	Highest flame lengths	Most departed	Most departed
Treatment extent	Subset	Subset	Maximum	Subset	Maximum
Area treated (ha)	1038	1036	2357	1036	2357
Percentage of landscape	13	13	30	13	30
Percentage of treatable	44	44	100	44	100
Objective	Reduce flame lengths in WUI, to facilitate suppression	Reduce flame lengths in high-risk areas, to promote low-severity wildfire and reduced smoke impacts	Same as FHR-sub with greater reductions in flame length and smoke	Increase canopy cover heterogeneity and wildlife diversity	Same as RES-sub with greater increases in canopy cover heterogeneity and wildlife diversity
Hypothesized trade-offs	Greater fire severity at a landscape scale; reduced canopy cover heterogeneity and biodiversity within treatments	Greater fire severity in WUI, reduced canopy cover heterogeneity and biodiversity within treatments	Greater fire severity in WUI, greatly reduced canopy cover heterogeneity and biodiversity within treatments	Much greater fire severity and smoke production at a landscape scale	Greater fire severity and smoke production at a landscape scale

Notes: Strategy names refer to wildland–urban interface (WUI), fire hazard reduction (FHR), and restoration (RES). Further details on treatment prescriptions are given in Table 2.

the Ward and Blackwood watersheds on the west shore of Lake Tahoe (Fig. 1). This landscape was selected because the US Forest Service manages a high percentage of the watershed, and it is a high-priority area for fuel treatments and forest restoration as it includes a substantial area of WUI. It also includes field sites sampled within the Lake Tahoe Upland Fuels Research Project (Manley et al. 2012) and a study of reference conditions (Maxwell et al. 2014, Taylor et al. 2014) that could be used to quantify structural and fuel characteristics before and after treatments. To represent pretreatment conditions, we used the Vegetation Inventory Strata map (TBEVM 3.0 or "Eveg layer"; Fig. 1b) developed from IKONOS hyperspectral satellite data (http://www.fs.fed.us/r5/rsl/projects/gis/data/tahoe/TnfStrata03_3.html) in 2002 (Greenberg et al. 2005), and raster fuels information from the LANDFIRE 2001 refresh program with 30 m resolution and no pretreatment adjustments (http://www.landfire.gov). Moghaddas et al. (2010) used a similar imagery-based approach to capture pretreatment fuels and forest structure information.

Fuel treatment strategies and objectives

Within our study area, we developed five alternative treatment strategies designed to compare trade-offs among different outcomes (Table 1). The main objective of the WUI strategy was to maximally reduce flame lengths within the area of the landscape designated as a WUI defense zone (Fig. 2a). The WUI strategy was based on predetermined locations within this landscape that had this objective, which had already been either proposed or implemented (see *Fuel treatment placement and extent*). The main objective of the fire hazard reduction (FHR) treatment strategy was to maximally reduce potential fire severity across the entire landscape, in order to promote low-severity fire effects (Table 1). The main objectives of the restoration (RES) treatment strategy were to restore the variable forest structure associated with frequent fire and to increase the diversity of wildlife habitat within the treatment (Table 1). The FHR and RES strategies were implemented both within a subset of the landscape (FHR-sub and RES-sub) and across the maximum treatable area (FHR-max and RES-max; see *Fuel treatment*
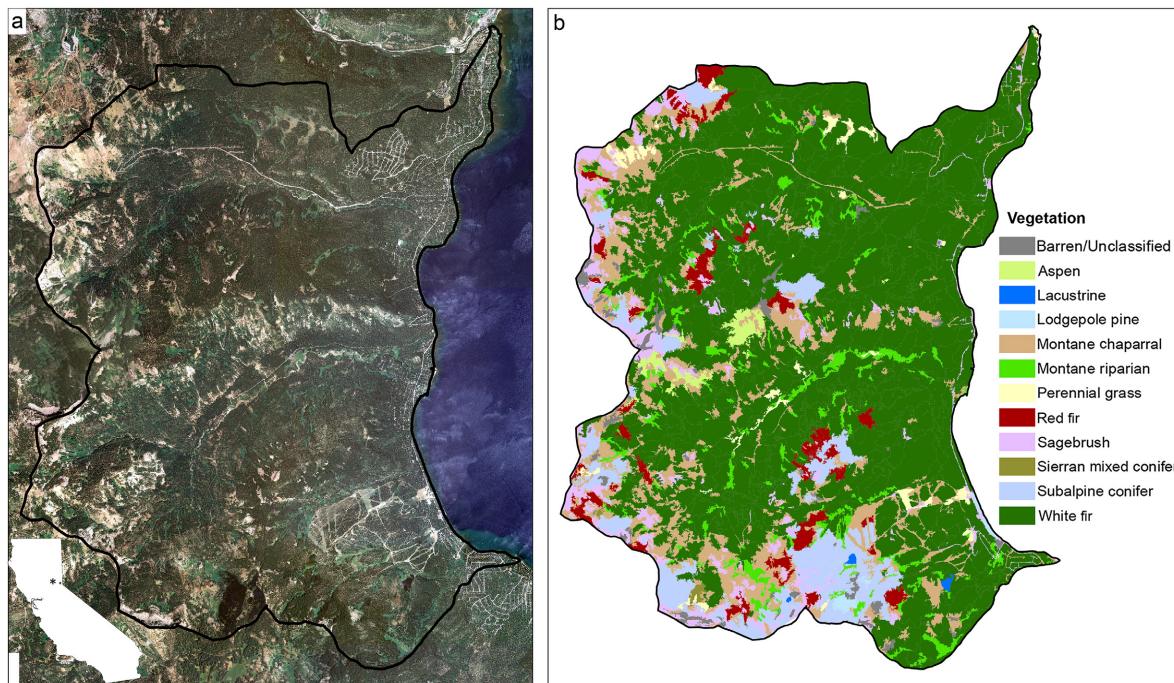
Fig. 1.   Ward Creek-Frontal Lake Tahoe Watershed. National Agriculture Imagery Program (NAIP) imagery from 2005 is shown in (a), with the watershed location given in the inset figure. Current vegetation (b) was derived from EVeg layer using California Wildlife Habitat Relations vegetation types. Treatments were restricted to the white fir and Sierra mixed-conifer vegetation types.

placement and extent), to investigate how the magnitude of trade-offs changed with increasing extent of fuel treatments.

Fuel treatment prescriptions.—We developed a set of prescription rules for each treatment strategy, which we implemented using 2001 LANDFIRE fuel layers for fuel models, canopy cover, canopy base height, and canopy bulk density (Table 2, Fig. 3). For the WUI and FHR strategies, we defined a uniform treatment prescription to mimic shaded fuel break or defensible fuel profile zone type treatments (Agee et al. 2000). The prescription rules were the same between the two strategies, although the locations of the treatments differed (see *Fuel treatment placement and extent*). Fuel models were reclassified to commonly used post-treatment models (Scott and Burgan 2005, Appendix S1: Fig. S1). All fuel models classified as moderate load (broadleaf) litter or large down logs (TL6 and TL7, respectively; Scott and Burgan 2005) were reclassified as moderate load conifer litter (TL3). Fuel models classified as very high load dry climate timber–shrub (TU5), which generally gives

the highest flame lengths of all fuel models in this landscape, were reclassified as either low load dry climate timber–grass–shrub or moderate load, humid climate timber–shrub (TU1 or TU2, respectively), with an even proportion of pixels receiving each reassignment and reassignments made randomly (Table 2). For the WUI and FHR strategies, canopy cover was uniformly reduced to 45%, canopy base height was uniformly increased to 2.5 m, and canopy bulk density was uniformly decreased to 0.1 kg/m^3, for all pixels where those critical thresholds had been exceeded (Table 2). Maximum canopy height, which can also influence fire behavior, was not altered in our treatments. All pixel reassignments were performed using the raster package in R (Hijmans and van Etten 2014).

For the restoration treatment strategies (RES-sub and RES-max), our goal was to use topography to inform a variable (heterogeneous) treatment prescription. Our reassignment pixel values were the same as for WUI and FHR strategies (except for canopy cover; Table 2). However, rather than applying the pixel reassignments
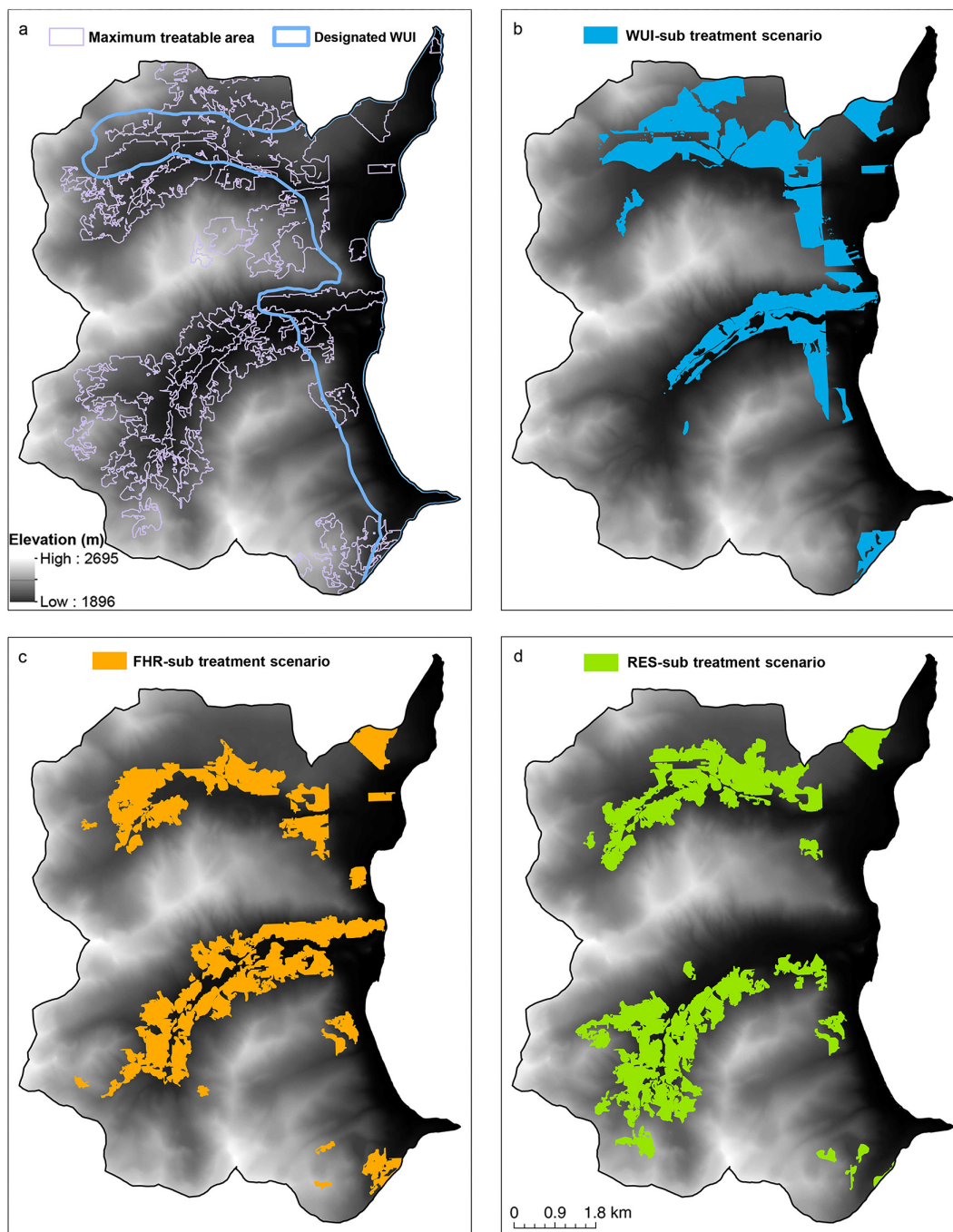
Fig. 2. Maximum treatable area (a) and treatment placement location for three different strategies (b–d), overlaid on a digital elevation model. Maximum treated area in (a) is all suitable vegetation types that were available to be mechanically treated (see *Methods*); this is the extent of the "max" treatment strategies (Table 1). The blue line in (a) delineates the area designated as WUI defense zone. FHR, fire hazard reduction; RES, restoration; WUI, wildland–urban interface.

Table 2. Prescriptions for different treatment strategies.

Strategy and slope position	Fire behavior fuel models			Canopy cover (%)		Canopy base height (m)		Canopy bulk density (kg/m³)	
	Original fuel model	Treated fuel model (%)	Percentage of pixels changed	Original value	Treated value	Original value	Treated value	Original value	Treated value
WUI, FHR									
All	TL6, TL7	TL3 (100)	100	>45	45	<2.5	2.5	>0.1	0.1
	TU5	TU1 (50) TU2 (50)	100	<45	NA	>2.5	NA	<0.1	NA
RES									
Ridge	TL6, TL7	TL3 (100)	50	All	0.55–0.65 of original	<2.5	2.5	>0.1	0.1
	TU5	TU1 (50) TU2 (50)	50			>2.5	NA	<0.1	NA
SW>30	TL6, TL7	TL3 (100)	50	All	0.55–0.65 of original	<2.5	2.5	>0.1	0.1
	TU5	TU1 (50) TU2 (50)	50			>2.5	NA	<0.1	NA
SW<30	TL6, TL7	TL3 (100)	40	All	0.65–0.75 of original	<2.5	2.5	>0.1	0.1
	TU5	TU1 (50) TU2 (50)	40			>2.5	NA	<0.1	NA
NE>30	TL6, TL7	TL3 (100)	35	All	0.75–0.85 of original	<2.5	2.5	>0.1	0.1
	TU5	TU1 (50) TU2 (50)	35			>2.5	NA	<0.1	NA
NE<30	TL6, TL7	TL3 (100)	30	All	0.80–0.90 of original	<2.5	2.5	>0.1	0.1
	TU5	TU1 (50) TU2 (50)	30			>2.5	NA	<0.1	NA
Valley	TL6, TL7	TL3 (100)	25	All	0.85–0.95 of original	<2.5	2.5	>0.1	0.1
	TU5	TU1 (50) TU2 (50)	25			>2.5	NA	<0.1	NA

Notes: The prescriptions for FHR-sub and FHR-max were identical, as were the prescriptions for RES-sub and RES-max. FHR, fire hazard reduction; RES, restoration; WUI, wildland–urban interface.

uniformly within a treatment unit, we reassigned a fraction of pixels based on slope position (Table 2) using the Land Management Tool (http://ice.ucdavis.edu/project/landscape_man agement_unit_lmu_Tool) (North et al. 2012*a*). Reductions in fuel models and canopy bulk density and increases in canopy base height were greatest on ridge tops and southwest-facing slopes and were less on northeast-facing slopes and valley bottoms (Table 2). Pixels to be changed were randomly selected in R, and the same selected pixels were changed in each layer. For canopy cover, to build on existing heterogeneity within the LANDFIRE layer, we reassigned all pixels to a smaller fraction of their original values, with greater reductions in canopy cover on ridge tops and southwest-facing slopes and lower reductions in canopy cover on northeast-facing slopes and valley bottoms (Table 2).

Fuel treatment placement and extent.—For the two different prescriptions defined above (uniform in WUI and FHR, and variable in RES), we either applied them to the maximum treatable area within the landscape or to a subset of the treatable area (Table 1). We defined the maximum treatable area within the landscape as the area that could be treated mechanically (2357 ha, or 30% of the total area landscape), using a hierarchy of constraints that removes areas that are unproductive or higher-elevation forest, legally restricted (i.e., wilderness and roadless), operationally prohibitive (i.e., steep slopes, distance from existing roads) and with administrative limitations (i.e., spotted owl habitat, riparian; Fig. 2a; North et al. 2015*a*). We termed the uniform treatment applied to the entire treatable area as "FHR-max," and the variable treatment applied to the entire area as "RES-max" (Table 1).
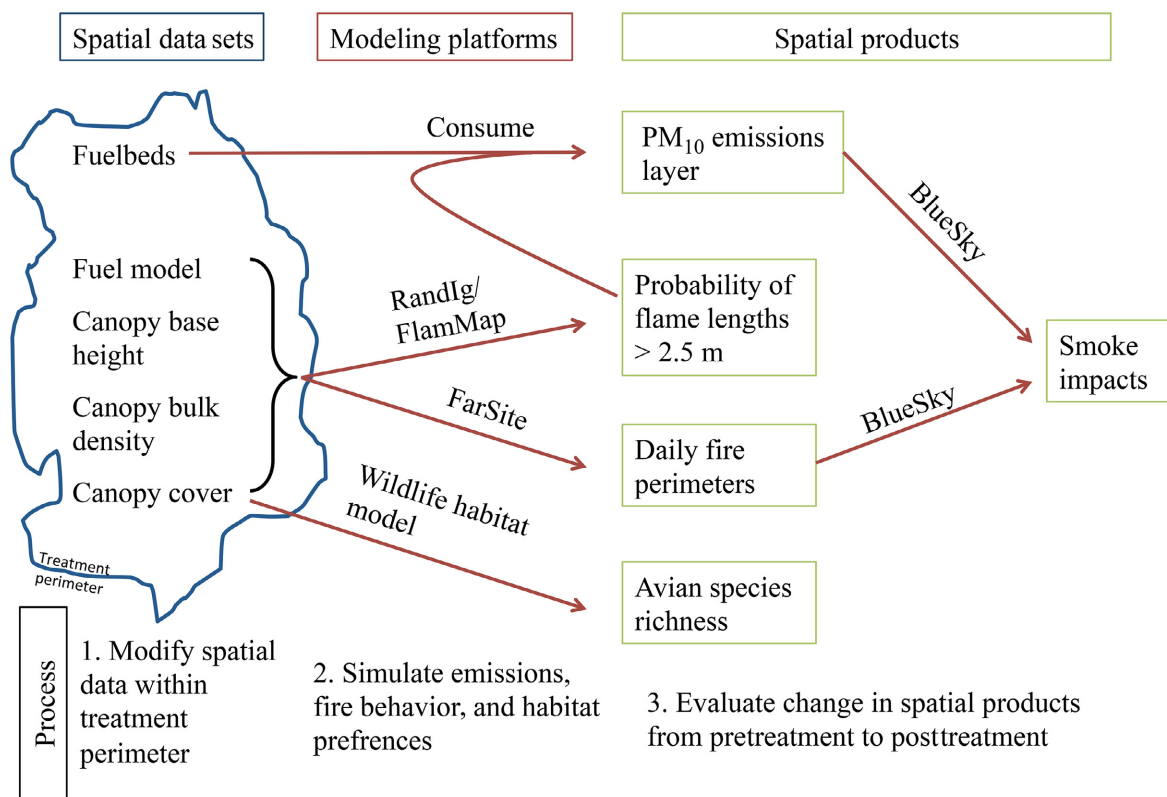
Fig. 3. Modeling framework schematic. A series of spatial data sets are modified within the assigned treatment perimeters and then processed through the appropriate modeling platform to generate a series of spatial products that are then used to evaluate treatment effects. Note that some spatial products are used to generate others.

In addition to treating the entire treatable area, we also applied the two different treatment prescriptions (uniform and variable) to three different but overlapping subsets of the treatable area, to investigate trade-offs relating to different commonly applied rules for treatment placement (Table 1, Fig. 2). For the WUI strategy, we placed the treatments in locations where the US Forest Service has either scheduled and/or implemented mechanical fuel treatments within the Ward/Blackwood watershed since 2005, which fall primarily in the WUI defense zone, a high-priority fire suppression zone (Fig. 2a, b). These treatments cover 1038 ha, or 13% of the total landscape area (Table 1).

We reasoned that the 1038 ha of treatments in the WUI strategy represented a realistic test of a landscape treatment strategy targeting 10–20% of a landscape (Long et al. 2014). Thus, for the two other treatment strategies where we treated

less than the maximum area available for treatment ("FHR-sub" and "RES-sub"), we developed placement rules designed to select areas for treatment that approximately equaled the 1038 ha treated in the WUI strategy, so we could control for the effects of treatment extent while testing for the effects of treatment placement (FHR-sub, which had the same treatment prescription as the WUI, but different locations) and treatment prescription (RES-sub). We ranked EVeg polygons, which are designed to capture relatively homogenous conditions across the landscape and are often used to delineate treatment units in US Forest Service projects (Nelson et al. 2015), by treatment priority based on the placement rules below, until an area equivalent to the area treated in the WUI strategy was reached (Table 1).

To locate treatments for the FHR-sub strategy, we simulated flame lengths using pretreatment LANDFIRE data under 97th percentile weather

conditions (see *Fire behavior models* below for more details) and calculated the mean flame length for each EVeg polygon ≥ 1 ha. We then ordered the polygons by mean flame length and selected polygons in order of decreasing mean flame length until a total area of approximately 1038 ha was reached (Fig. 2c). Thus, the FHR-sub treatment strategy is identical to the WUI treatment strategy in prescription and extent, but differs in placement.

To locate treatments for the RES-sub strategy, we identified areas that had the greatest increases in tree density over their historical estimates. We estimated the historical tree density for all trees greater than 10 cm dbh using data from Maxwell et al. (2014), who used predictive vegetation mapping combined with historical forest reconstructions to estimate historical forest conditions across the entire Lake Tahoe Basin. Maxwell et al. (2014) developed a raster layer of historical vegetation types, where each vegetation type was assigned a total tree density value for all stems greater than 10 cm dbh. We averaged these pixel values within each EVeg polygon to determine the mean historical tree density for each EVeg polygon. Because EVeg data are unreliable for calculating density of modern trees less than 40 cm (Greenberg et al. 2005), we developed a small-tree correction factor using contemporary vegetation data collected from plots ($n = 40$) in the Lake Tahoe Basin Management Unit (P. M. Manley, *unpublished data*) in which fixed-radius plots (0.25 ac) were established and all trees and snags greater than 10 cm dbh were tagged and measured prior to and following mechanical treatment. From these plots, we were able to calculate current density (in stems/ha) for trees greater than 40 cm and trees between 10 and 40 cm dbh. The mean density of trees greater than 40 cm dbh from the plot data was 85 stems/ha, very close to the mean estimate from the EVeg data for the polygons covering these plot locations (82.4 stems/ha). We calculated the correction factor as the mean density of trees between 10 and 40 cm dbh for those plots, which was 243, and added that value to the tree density for trees greater than 40 cm dbh in each EVeg polygon, in order to compare current density of trees greater than 10 cm dbh with the historical estimates from Maxwell et al. (2014). For each EVeg polygon, we then calculated the mean departure index as the difference between the current corrected density and the estimated

historical density and selected polygons in order of decreasing departure index (from more departed to less departed) until a total area of approximately 1038 ha was reached (Fig. 2d).

Process modeling

Having defined the treatment placement and extent, and modified the relevant spatial data layers within the treatments according to the treatment prescription rules, we then used a series of modeling platforms to simulate potential wildfire spread, severity and emissions, as well as wildlife habitat suitability, across the entire landscape (Fig. 3).

Fire behavior models.—We quantified the effects of the different treatment strategies on potential fire severity in the study area using the Random-Ignitions module (RandIg) of the fire simulation software FlamMap. RandIg uses the minimum travel time algorithm (Finney 2002) to simulate fire growth during discrete burn periods under constant weather conditions. Each simulated fire approximates the growth and behavior of single large spread events, which account for a majority of area burned in large fires (Finney et al. 2007). We simulated 10,000 random ignitions throughout the landscape under 97th percentile summer weather conditions based on weather station data from the Homewood Remote Access Weather Station (RAWS) within the study area (Appendix S1: Table S1). We parameterized each RandIg ignition to burn for 12 h, which produced a mean fire size of 745 ha. This approximates the area burned in a single 10-h rapid-fire spread period (968 ha) on the Angora Fire in the southern Lake Tahoe Basin in 2007 (Murphy et al. 2007). Although the temperature and relative humidity conditions in our simulations were more extreme than during the burn period on the Angora Fire, the Angora Fire was preceded by record high temperatures and low humidities that were comparable to our values (Randy Striplin, *personal communication*). The 97th percentile sustained 6.1 m (20 ft) wind speed for our simulations from the Homewood RAWS was 24.1 km/h (15 miles/h). We adjusted this value based on calibrations in Crosby and Chandler (1996) to 32.2 km/h (20 miles/h; Appendix S1: Table S1) to represent the maximum sustained 1-min wind speed, which is a better predictor of extreme fire behavior (Crosby and Chandler

1996). Our estimate of mean sustained 6.1 m (20 ft) wind speed over the course of the day was generally consistent with estimates of sustained wind speed on a high-severity burn day during the 2007 Angora Fire in the Lake Tahoe Basin (14–21 km/h [9–13 miles/h] on 24 June 2007), although our maximum 1-min sustained 6.1 m (20 ft) wind speed was less than maximum wind speeds on the Angora Fire (45 km/h [28 miles/h]; Murphy et al. 2007; Randy Striplin, *personal communication*). Wind direction for a given ignition was assigned to either 202°, 225°, or 247° with equal probability, as these southwesterly winds are the most common wind direction for 97th percentile wind speed conditions based on Homewood RAWS data.

The 10,000 fire simulations generated a probability distribution of flame lengths, in 0.5 m classes, for each pixel within the landscape. We assumed the likelihood of high-severity (stand-replacing) fire would be greatest when flame lengths exceeded 2.5 m (8 ft), as these flame lengths are commonly associated with tree torching and crown fire initiation (e.g., Collins et al. 2013). We compared the effectiveness of different treatment strategies in reducing flame lengths by estimating for each strategy the total area with greater than 50% probability of experiencing greater than 2.5 m flame lengths. It should be noted that these flame length probabilities represent the conditional hazard given an ignition occurring under relatively high fire weather, which was constant for a given burn period. Constant fire weather throughout a 12-h burn period is an oversimplification of weather conditions in actual wildfire, but is a necessary assumption for our modeling approach. As such, our assessments of fire hazard are best interpreted as relative comparisons among treatment strategies from the upper end of potential fire behavior in this watershed.

Emissions models.—We estimated potential particulate matter (PM) emissions (e.g., smoke) from wildfire under 97th percentile weather conditions for the entire landscape (Appendix S1: Table S1). To estimate fuel consumption during the simulated wildfire, we characterized fuels in the Fuel Characteristics Classification System (FCCS) to represent different vegetation types and post-treatment fuel characteristics. The FCCS fuelbed base map layer used in this analysis was based on the Lake Tahoe Fuelbed Handbook and

map layer (Ottmar and Safford 2011). For each of the five treatment strategies, the fuelbed base map layer (Appendix S1: Fig. S2) was updated to modify the fuelbeds in the same pixels that were modified for the fire behavior strategies (Table 2). These post-treatment fuelbeds represent the various harvest and fuel treatment techniques in mixed-conifer and Jeffrey pine-white fir forests that have actually been conducted in the study area in the past 10 yr (Appendix S1: Table S2).

We generated emissions estimates from these fuelbeds under wildfire conditions using Consume version 4.2, using the wildfire conditions detailed in Appendix S1: Table S1. Consume predicts fuel consumption, pollutant emissions, and heat release based on input fuel characteristics (from FCCS fuelbeds), fuel moistures and other environmental variables. Because Consume captures the inherent complexity of wildland fuels through a close interface with the FCCS, specific fuel strata and/categories can be targeted for prescription or noted as a potential source of pollutant emissions depending on the fuel treatment and burn strategy. To parameterize the canopy consumption and shrub consumption percentages in Consume, which are drivers of emissions, we used three different consumption percentages for three different flame length classes, based on the output from the RandIg models described above (Fig. 3). Low-severity fire was modeled as flame lengths less than 1 m (3.28 ft), with canopy consumption of 0% and shrub consumption of 25%. Moderate-severity fire was modeled as flame lengths 1–2.5 m, with canopy consumption of 25% and shrub consumption of 50%. High-severity fire was modeled as flame lengths greater than 2.5 m (8.2 ft), with canopy consumption of 75% and shrub consumption of 75%. We then used Consume to calculate three layers of total PM_{10} (PM < 10 μm in diameter) emissions produced by fuel consumption at these three flame length classes and combined these layers into a single emissions layer weighted by how likely each flame length class was relative to the other two, on a pixel-by-pixel basis. Thus, emissions in areas more likely to burn at high severity were higher than emissions in areas more likely to burn at low severity, for a given fuelbed.

We also examined how emissions from the different treatment strategies might be dispersed

across a broader region, under an extreme wildfire event that is characteristic of high fire-risk scenarios in this region. We simulated burn day perimeters for each of the five treatment strategies using FARSITE (Finney 2004), under a single wildfire event burning under the same 97th percentile weather conditions used to model wall to wall fire spread using RandIg (see *Fire behavior models* above). The simulated fire began as an ignition on a ridgetop in the southwest corner of the study area where there is road access, and was fueled by southwesterly winds, burning almost the entire area within a 4-d span. To estimate the potential for downwind smoke impacts under the different strategies, we estimated the PM emissions for each burn day from the weighted Consume output.

We used these daily emissions estimates for each treatment strategy as inputs to a series of single-day runs of the BlueSky modeling framework (Fig. 3; playground.airfire.org), a common tool for modeling smoke transport (Goodrick et al. 2013). The BlueSky modeling framework yields outputs in average hourly PM emissions ($\mu g/m^3$) over a given 2-km grid cell within 10 m of the ground surface. The model currently cannot use probabilistically derived (e.g., 97th percentile) meteorological inputs, but instead requires meteorology from actual days in the meteorological record. We chose 12–15 August 2014 as days that closely represented the 97th conditions in the above fire modeling exercise. We mapped contours of $PM_{2.5}$ concentrations produced by the BlueSky model runs, which indicate the dispersion of fine (< 2.5 μm in width) PM that poses the primary human health concern. $PM_{2.5}$ emissions from forest fire scale linearly with PM_{10} emissions, by a factor of 0.847. We compared the likely patterns of $PM_{2.5}$ dispersion among the different treatment strategies using the contour indicating $PM_{2.5}$ concentrations greater than 20 $\mu g/m^3$. We evaluated the spatial extent of the 20 $\mu g/m^3$ contour for the nighttime period following day 2, which had the greatest difference in emissions among the strategies (Table 4), to compare how different treatment strategies might affect downwind exposure to unhealthy smoke levels.

Wildlife habitat models.—To predict the impact of the simulated fuel treatments on wildlife, we used a preexisting avian point count data set on the passerines and near passerines in the Lake Tahoe Basin. Point counts were conducted from May to July (which captures the breeding period for most of the species included) of 2002–2005 as part of a multispecies monitoring program. These data included detections of 66 bird species collected during two to three repeat visits to 742 point count stations distributed across the basin. These data have previously been analyzed using a multispecies modeling approach that estimated species-specific occurrence probabilities relative to abiotic and biotic (e.g., forest structure) variables (White et al. 2013*a*, *b*). Here, we use this data and a similar modeling approach to quantify how treatment prescription, placement, and extent impact the occurrence and distribution of each of these 66 species across the watershed.

To accomplish this goal, we first modeled how abiotic (elevation, easting and percentage of urban development) and biotic (canopy and shrub cover) variables within a 150-m radius of each point count station influenced the probability of occurrence of each species (corrected for species-specific differences in detection probabilities; see Appendix S2 for more details). Using a Bayesian framework, we used these species-specific occurrence probabilities to generate occurrences (0 if absent, 1 if present) with Bernoulli trials, which give a distribution of presence/absence values (3000 values consisting of 0 or 1 values) for each species in each treatment strategy, for each 30-m pixel. Canopy cover values for each pixel were based on LANDFIRE data modified to our treatment strategies (see *Fuel treatment prescriptions* above). Shrub cover values were based on EVeg data and were not varied in the prescriptions. To estimate species richness, we summed the total number of individuals that were predicted to be present for each Bernoulli trial, which produced 3000 estimates of total species richness for each treatment strategy, and took the mean value of these estimates for each pixel. The modeled treatment effects on wildlife habitat do not assume any subsequent wildfire post-treatment; they are simply the effects of modifying forest structure during the fuels treatment.

RESULTS

Regardless of the treatment strategy, the amount of change in the outcomes of interest

across the landscape was generally minor (Fig. 4a, Table 3). The WUI, FHR-sub and FHR-max treatments were all more effective at achieving their respective objectives compared to alternative strategies. The WUI strategy delivered the greatest reduction in flame length within the WUI defense zone, while the FHR-sub and FHR-max strategies were the most effective at reducing flame length across the landscape, as measured by the reduction in the proportion of the landscape with greater than 50% probability of experiencing flame lengths greater than 2.5 m (Fig. 4a). While the FHR-max strategy had the greatest reduction in flame length across the

landscape, it was only slightly more effective than the FHR-sub strategy, despite treating over twice the area. However, the FHR-max strategy was much more effective than the FHR-sub strategy at reducing PM_{10} emissions across the landscape (Fig. 4a), despite comparable reductions within the treatment (Fig. 4b).

The trade-offs of the WUI, FHR-sub, and FHR-max strategies with respect to structural heterogeneity and wildlife diversity were mixed. All three strategies reduced canopy heterogeneity, as measured by the standard deviation of canopy cover within the treatments, much more than the RES strategies (Fig. 4b), a function of the uniform
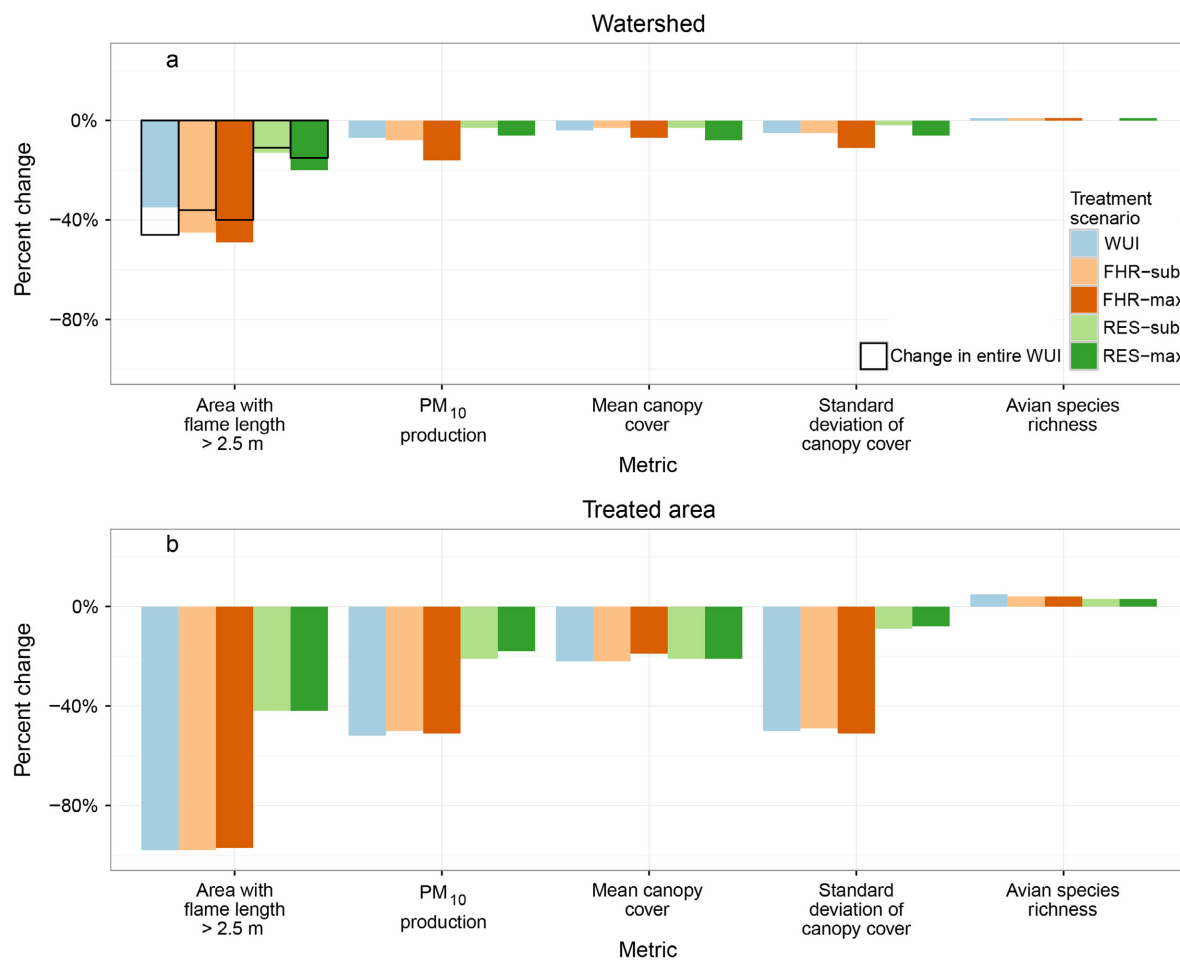

Fig. 4. A summary of the outcomes of different treatment strategies, measured as percentage change from a no-treatment alternative. Percentage change is estimated across the entire landscape (a) and within the treated area only (b); black boxes in (a) indicate the percentage change in the area with a greater than 50% likelihood of flame lengths greater than 2.5 m within the entire WUI defense zone (blue line in Fig. 2a) caused by a given treatment. FHR, fire hazard reduction; RES, restoration; WUI, wildland–urban interface.

Table 3. Comparison of outcomes among treatment strategies.

Outcome	Scope	Pre/post-treatment	Treatment strategy				
			WUI	FHR-sub	FHR-max	RES-sub	RES-max
Proportion of landscape with >50% probability of flame lengths >2.5 m	Entire WUI	Baseline	0.215	0.215	0.215	0.215	0.215
		Post-treatment	0.117	0.137	0.129	0.192	0.182
	Landscape	Baseline	0.092	0.092	0.092	0.092	0.092
		Post-treatment	0.060	0.051	0.047	0.080	0.074
	Treated area	Baseline	0.198	0.296	0.142	0.166	0.142
		Post-treatment	0.004	0.005	0.004	0.097	0.082
PM_{10} production across landscape (metric tons)	Landscape	Baseline	3037.06	3037.06	3037.06	3037.06	3037.06
		Post-treatment	2827.66	2807.48	2545.45	2953.94	2860.31
	Treated area	Baseline	390.56	403.14	955.64	383.74	955.64
		Post-treatment	185.71	203.15	467.42	302.22	779.72
Mean canopy cover (%)	Landscape	Baseline	45.23	45.23	45.23	45.23	45.23
		Post-treatment	43.56	43.65	42.11	43.79	41.78
	Treated area	Baseline	56.15	55.35	53.79	52.73	53.79
		Post-treatment	43.53	43.38	43.44	41.85	42.32
Standard deviation of canopy cover	Landscape	Baseline	17.43	17.43	17.43	17.43	17.43
		Post-treatment	16.49	16.56	15.56	17.12	16.33
	Treated area	Baseline	11.79	11.98	11.64	11.56	11.64
		Post-treatment	5.84	6.05	5.71	10.57	10.76
Mean avian species richness	Landscape	Baseline	20.03	20.03	20.03	20.03	20.03
		Post-treatment	20.19	20.18	20.32	20.13	20.26
	Treated area	Baseline	19.62	19.83	20.14	20.19	20.14
		Post-treatment	20.60	20.70	20.96	20.74	20.79

Notes: Pre- and post-treatment values are shown for each outcome, both across the entire landscape and within the treated area only. The flame length outcome is also reported for the entire WUI defense zone shown in Fig. 2a. FHR, fire hazard reduction; RES, restoration; WUI, wildland–urban interface.

treatment prescriptions within these strategies. However, the reduction of canopy cover heterogeneity across the landscape by the WUI and FHR strategies was very minor, at less than 10%, and was actually comparable to the RES strategies (Fig. 4a, Table 3). Furthermore, the WUI and FHR strategies actually led to increases in predicted avian species richness within the treatments that were greater than the increases from the RES strategies (Fig. 4b).

The RES strategies therefore had mixed success in achieving their objectives. While they clearly maintained more canopy cover heterogeneity within the treated area than the WUI and FHR strategies (Fig. 4b), they still caused a slight decrease in heterogeneity relative to pretreatment conditions. Perhaps more surprising, the RES treated areas had a smaller increase in predicted avian species richness than the WUI- and FHR-treated areas. The median change in individual species occurrence probabilities was fairly constant across strategies, but the WUI and FHR strategies had more species with greater increases in occurrence probability than the RES

strategy (Fig. 5). The magnitude of change in avian richness across the landscape, however, was very low for all treatment strategies (Fig. 4a). The hypothesized trade-offs of the RES treatments were as expected; reductions in potential flame length and PM_{10} emissions relative to the untreated scenario were lower in magnitude than the WUI and FHR strategies (Fig. 4).

Simulated fire behavior within the study area was generally dominated by low- to moderate-flame length classes, even under 97th percentile weather conditions. The area with a greater than 50% probability of burning at greater than 2.5 m flame lengths was 706 ha, or 9% of the total study area, without any treatment (Table 3). However, all treatment placements but particularly the FHR and WUI strategies resulted in substantial treatment area being placed in zones with heavy fuel models and high potential flame lengths (Appendix S1: Figs. S1, S3), leading to 13–44% reductions in the area prone to high-severity fire across the landscape (Fig. 4).

For a single high-risk wildfire simulation under 97th percentile weather conditions, the FHR-sub
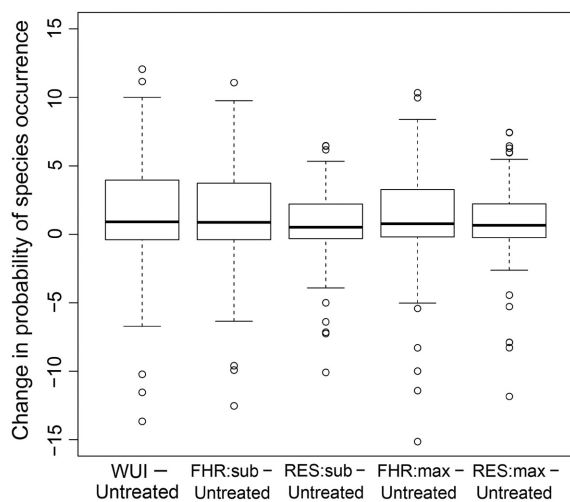
Fig. 5. The median and interquartile range for the change in the probability of individual species occurrence under different treatment scenarios in the study area. FHR, fire hazard reduction; RES, restoration; WUI, wildland–urban interface.

and especially FHR-max scenarios were the most effective at slowing the rate of fire spread (Fig. 6). Because the high-risk fire scenario that we identified involved an ignition in the southwestern portion of the landscape with southwesterly winds, the FHR and RES treatment strategies and particularly the max-treatments were positioned in closer proximity to the early days of fire spread, relative to the WUI treatment strategy (Fig. 2). With a more uniform effect on the fuel models that drive fire spread (Appendix S1: Fig. S1), the FHR strategies had 222 fewer burned ha than the RES strategies for the subtreatments, and 486 fewer burned ha for the max-treatments, by the end of the second burn day (Table 4). The reductions in burned area in the FHR strategies relative to a no-treatment alternative were even greater, by 287 ha for FHR-sub and 527 ha for FHR-max. Accordingly, the FHR strategies resulted in a smaller downwind area affected by smoke emissions above 20 μg/m³, relative to either the no-treatment alternative or the restoration scenarios, which were all similar to each other (Fig. 7).

Discussion

Our results clearly demonstrate that modifying fuel treatment prescriptions, placement and landscape extent have differential effects across a range of resources. However, for all treatment strategies, we considered the magnitude of impacts at the landscape scale tended to be fairly small, relative to other studies in similar forest types (Collins et al. 2011, 2013). We highlight four key findings from our simulations and discuss how applicable they are to other similar landscapes. First, the proportion of the landscape with high potential flame lengths was affected more by treatment prescription than by treatment placement or extent. Second, the potential smoke impacts at the landscape scale were affected by both prescription and extent, but less affected by placement. Third, the change in canopy heterogeneity within treatments was strongly affected by prescription, but changes at the landscape scale were fairly minor. Fourth, changes in avian richness were minor overall and were not tightly coupled to changes in canopy heterogeneity within treatments but rather to how much a previously closed canopy was opened.

Treatment prescriptions that called for a uniform reduction of fuels (WUI and FHR) had a much greater reduction in the proportion of the landscape with high potential flame lengths, compared to the heterogeneous restoration treatments (Fig. 4). This is primarily due to the more complete reduction of fuels within the treated areas (Appendix S1: Fig. S1). However, increasing the extent of the intensive FHR treatments from 13% of the landscape to the maximum 30% of the landscape caused little additional reduction in area with high potential flame lengths. Beyond the initial 13% treated area, most of the additional area burned at high severity was on private land at the eastern edge of the landscape rather than the higher-elevation forests on public land that were treated by scaling up the treated area to the maximum 30% (Appendix S1: Fig. S3). This finding illustrates that for some landscapes, smaller treatments of high-vulnerability portions of the landscape may be sufficient to reduce potential high-severity fire, particularly if the fire hazard is relatively low prior to treatment (Loehle 2004). This study landscape also had high overlap between the WUI defense zone (Fig. 2a) and the area with the highest probability of high-severity fire (Appendix S1: Fig. S3). As a result, the WUI and FHR-sub treatments had
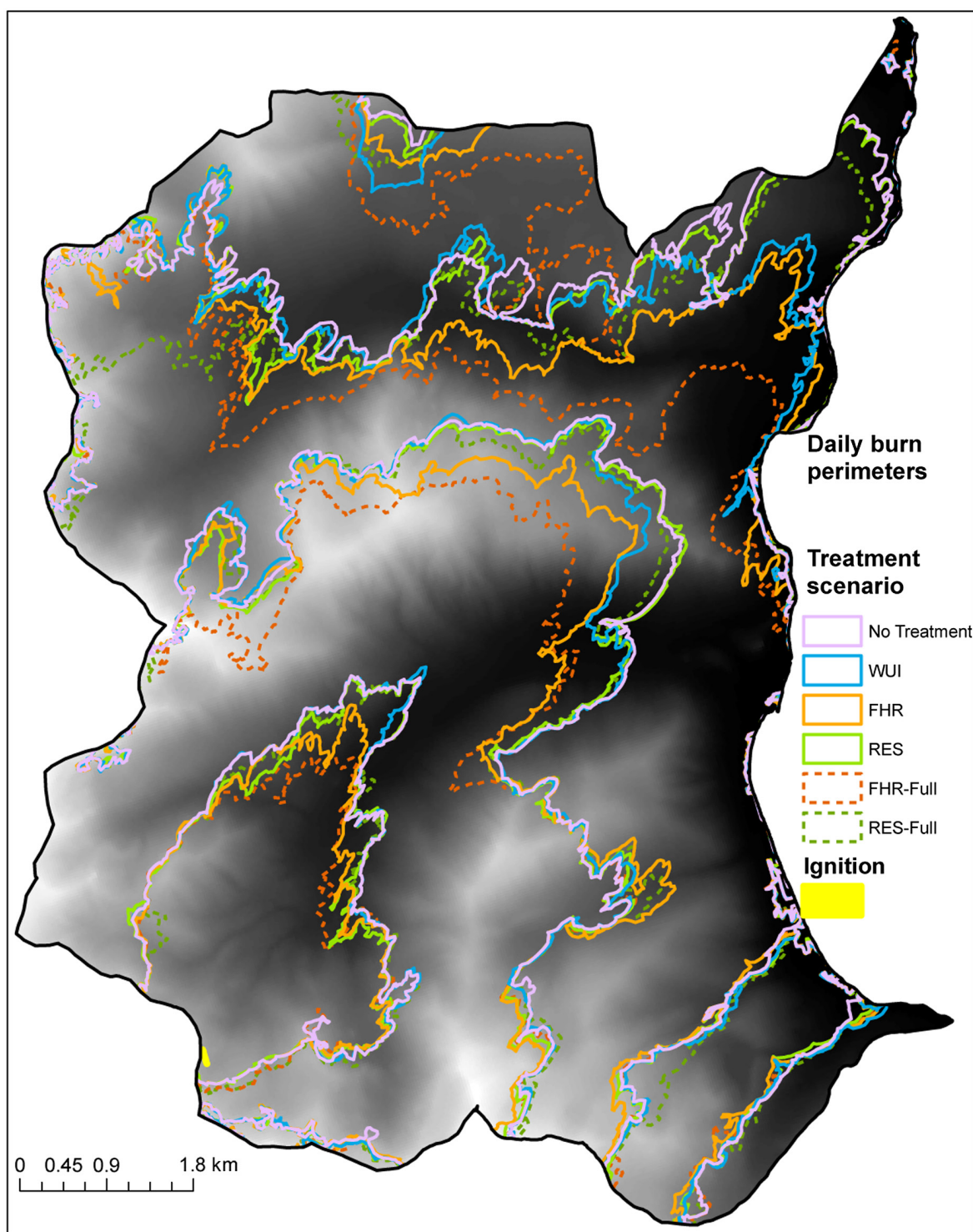
Fig. 6. Daily burn perimeters for different treatment strategies and the no-treatment alternative. The ignition line is shown in yellow at the southwest corner of the watershed. Four groups of perimeters reflect 12–15 August burn days. FHR, fire hazard reduction; RES, restoration; WUI, wildland–urban interface.

Table 4. Daily emissions of PM_{10}, and area burned for the simulated 4-d fire event, within the target study area for each of five treatment strategies and a no-treatment alternative.

| | 12 August | 13 August | 14 August | 15 August | |
Treatment strategy	d1	d2	d3	d4	Cumulative
Emissions (PM_{10})					
No treatment	238	1170	1141	438	2987
WUI	232	1127	1001	412	2772
FHR-sub	187	994	1049	518	2748
RES-sub	205	1130	1127	438	2899
FHR-max	146	816	1015	498	2474
RES-max	216	1107	1007	476	2806
Burn area (ha)					
No treatment	733	2518	3035	1337	7624
WUI	730	2462	2876	1510	7578
FHR-sub	629	2335	2814	1795	7574
RES-sub	687	2499	3035	1391	7611
FHR-max	581	2143	2654	2002	7381
RES-max	726	2484	2707	1679	7596

Notes: Metric tons PM_{10} for a given burn day. FHR, fire hazard reduction; RES, restoration; WUI, wildland–urban interface.

very similar effects on reducing flame lengths across the landscape and within the WUI defense zone (Fig. 4) despite following different placement rules. These results suggest that in some landscapes, there is potential for mechanical fuel treatments to simultaneously meet multiple objectives, for example, WUI protection and landscape FHR (Winter et al. 2002, Reinhardt et al. 2008).

One hypothesized trade-off (Table 1) was that treatment strategies that were not focused on reducing flame lengths across the landscape (i.e., the RES strategies) would be less effective at reducing potential smoke impacts from wildfire. However, we found that smoke impacts were generally less sensitive than potential flame length was to treatment prescription, while the extent of the treatment appeared to be more relevant in predicting smoke impacts (Fig. 4). For example, the total potential PM_{10} emissions from the RES-max strategy were similar to the WUI and FHR-sub strategies, despite the RES-max strategy having a greater fraction of the landscape with potential high-severity fire (Table 3). Downwind smoke effects appear driven largely by area burned rather than fuel loads and severity (Figs. 6, 7). Much of the upper Ward and Blackwood watersheds has "stringers" of higher-elevation forest, dominated by a mix of white fir, red fir, and subalpine species, which readily propagates fire through the landscape even if flame

lengths rarely exceed 2.5 m (Fig. 1b; Appendix S1: Fig. S3).

Mechanical treatments may need to be more extensive if the objective is to reduce smoke impacts by reducing area burned in a given period. It has been argued that fuel treatments ought not be implemented to stop fire or enhance suppression effectiveness, but rather to reduce uncharacteristic fire effects when fire eventually does occur (Reinhardt et al. 2008). Our results demonstrating greater reductions in potential flame length (Appendix S1: Fig. S3) than the rate of spread (Fig. 6) support this assertion. However, our analysis does not account for the impact of fire suppression, which would likely be facilitated by the overall reduction in flame lengths. If successful, suppression would likely reduce the area burned, which is the primary driver of smoke impacts. However, given the ecological benefits of fire, and the important role of recurring prescribed fires and managed wildfires in maintaining landscapes in a resilient condition, it is necessary to tolerate some smoke production in order to achieve other management objectives (North et al. 2015b, Schweizer and Cisneros 2016).

The treatment prescription had strong effects on canopy heterogeneity. RES treatment prescriptions slightly reduced variation in canopy cover at the 30-m scale within the treatment (Fig. 4b), despite the prescription explicitly attempting to maintain variation in canopy and
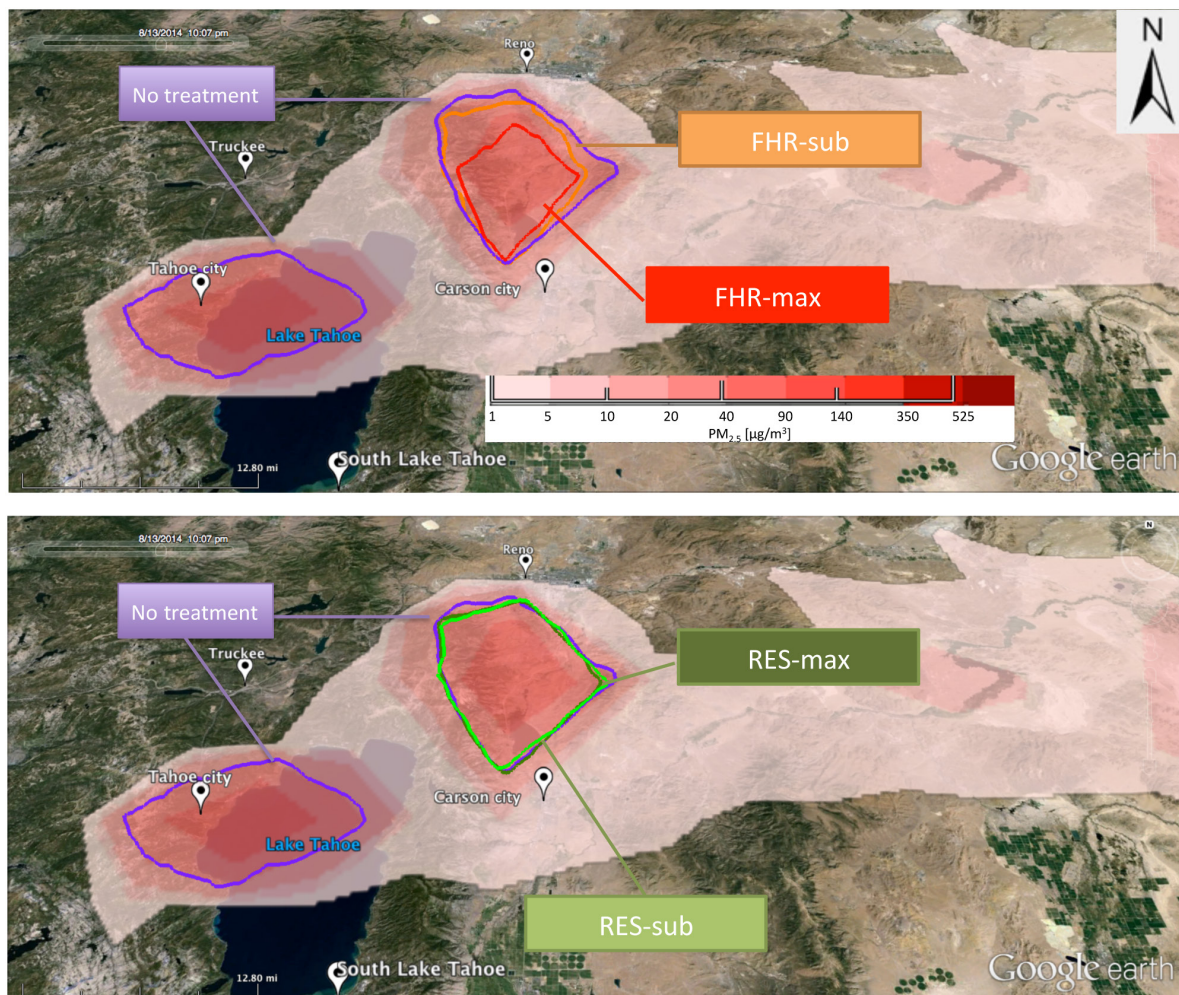
Fig. 7. Smoke impacts for different treatment strategies and the no-treatment alternative. Contours show locations of areas where PM$_{2.5}$ levels exceeded 20 μg/m^3 at 9 pm on the night of 13 August (day 2; the day with the greatest change in emissions among all scenarios) among the different treatment strategies. The underlying PM$_{2.5}$ concentrations (pink shades) are from the no-treatment alternative. On this particular burn day, the FHR scenarios reduced the area with greater than 20 μg/m^3 in the downwind Reno, Nevada area (upper right), by approximately 20% (FHR-sub) to 56% (FHR-max). FHR, fire hazard reduction; RES, restoration.

other fuels at that scale (Table 2). This illustrates that *variability* in canopy cover can still be quite high prior to fuel treatments, even if the *total* canopy cover across the treated area is also high (Stevens et al. 2015). However, the variable RES treatments maintained greater variation in canopy cover within treatments and may be a better approximation of historical forest structure compared to the more uniform treatment prescriptions (North et al. 2009). Moreover, such variable treatments may accentuate heterogeneity in

stand structure and composition under subsequent fires (Stevens et al. 2015), if tree clumps within the treatment burn at high severity as they did in our simulation (Appendix S1: Fig. S3). Despite this important effect of treatment prescription on canopy cover variation within the treated area, the standard deviation of canopy cover changed relatively little at the landscape scale (Fig. 4a), where treatments that covered the greatest extent (FHR-max and RES-max) actually had the greatest reduction in canopy cover

variance. All five treatment strategies generally targeted areas that had high canopy cover prior to treatments, and thus, all strategies reduced the contribution of very high canopy cover stands to the overall landscape-scale canopy cover.

The primary trade-off that we hypothesized for our set of treatment strategies, namely that treatments most effective at reducing fire and smoke impacts might also have negative effects on habitat for wildlife species (Table 1), was not borne out by our simulations. The main reason for this was that avian species richness was not closely linked to heterogeneity in canopy cover and forest structure within treated areas (Fig. 4b). In fact, avian richness generally increased following treatments and increased more following uniform treatment prescriptions (WUI and FHR) than variable treatment prescriptions (RES; Appendix S1: Fig. S4). This suggests that the reduction in canopy cover modeled in the WUI and FHR treatments may be a good compromise between species preferring more closed canopy conditions and those that prefer more open canopy conditions.

While overall richness increased more in the uniform treatments, there were fewer species experiencing declines in abundance of more than 5% in the more variable restoration treatments (Fig. 5). Depending on the level of concern for the declining species, there may actually be a trade-off between reducing fire hazards and the conservation of these species. Sensitive wildlife species, particularly those with special legal protections, are a particular concern in landscape restoration (Truex and Zielinski 2013, Tempel et al. 2014, 2015). Many of these species are associated with patches of dense canopy cover and large trees. Our modeling suggests potential to increase landscape (gamma) diversity by increasing differentiation among habitats (beta diversity; see also White et al. 2013b). Although there are potential downsides to species that favor high canopy cover, research has suggested that some of these species, such as Northern flying squirrels, can tolerate localized canopy reductions by redistributing their populations across the landscape within untreated patches (Sollman et al. 2016).

The relatively minor differences in outcomes between the strategies are strongly affected by the ubiquitous and significant structural changes resulting from a century of fire suppression (Safford and Stevens, *in press*). While fuel treatments generally reduced fire severity, effects were localized and did not dramatically change fire extent without uniformly treating ~30% of the landscape (Fig. 6), because much of the landscape outside of the treatments continues to provide fuel loads and continuity for rapid-fire expansion. This condition also affected the modeled wildlife response, with such extensive high canopy cover across the landscape that greater canopy reductions, even when more uniformly applied (i.e., the WUI and FHR strategies), improved habitat for a majority of species. Our modeling comparison suggests that the extent and degree of fuel loading in fire-suppressed landscapes may create such inertia that different strategies have similar limited outcomes. To effect greater change and maintain the desired variability associated with frequent fire regimes, managers will likely need to increase treatment extent beyond the limited land base available to mechanical methods using prescribed burning and managed wildfire (North et al. 2012b, 2015b). This is particularly true as climate change increases the likelihood of large unplanned fire events (Stephens et al. 2014b). Forest restoration efforts may use historical conditions as a guideline, but it will be important to design treatments that maintain forest resilience to future conditions that may include greater incidence of fire, drought, and insect outbreaks (Earles et al. 2014, Safford and Stevens, *in press*).

Management of fire-dependent landscapes involves balancing multiple objectives such as WUI protection, restoration of ecosystem structure, and fire regimes and maintaining wildlife habitat, which makes decisions over optimal treatment strategies difficult, particularly in areas with an extensive WUI (Ager et al. 2007). Our findings indicate that mechanical treatments over a relatively small fraction (10–20%) of a landscape might be a reasonable first step in a treatment strategy for broader landscape restoration by encouraging low-intensity fire and targeting high-risk areas. This approach may generate relatively benign impacts for wildlife overall and allow treatments to avoid particularly sensitive habitat areas for species associated with more closed canopy conditions. This initial approach may continue to rely on suppression of wildfires to avoid harmful smoke emissions that are strongly tied to the rate and extent of fire spread However, smoke emissions should be

recognized as a natural component of fire-prone forests (Schweizer and Cisneros 2016). Most importantly, outcomes from fuel treatments may depend greatly on subsequent use of fire (Stevens et al. 2014). The benefits of promoting fine-grain structural variability may be more fully realized following moderate burning that creates diverse postburn stand structures and biotic communities that are resilient to future wildfires.

Acknowledgments

Literature Cited

Agee, J. K., and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83–96.

Agee, J. K., B. Bahro, M. A. Finney, P. N. Omi, D. B. Sapsis, C. N. Skinner, J. W. van Wagtendonk, and C. Phillip Weatherspoon. 2000. The use of shaded fuelbreaks in landscape fire management. Forest Ecology and Management 127:55–66.

Ager, A. A., A. J. McMahan, J. J. Barrett, and C. W. McHugh. 2007. A simulation study of thinning and fuel treatments on a wildland–urban interface in eastern Oregon, USA. Landscape and Urban Planning 80:292–300.

Ager, A. A., N. M. Vaillant, and M. A. Finney. 2010. A comparison of landscape fuel treatment strategies to mitigate wildland fire risk in the urban interface and preserve old forest structure. Forest Ecology and Management 259:1556–1570.

Calkin, D. E., J. D. Cohen, M. A. Finney, and M. P. Thompson. 2014. How risk management can prevent future wildfire disasters in the wildland-urban interface. Proceedings of the National Academy of Sciences USA 111:746–751.

Churchill, D. J., A. J. Larson, M. C. Dahlgreen, J. F. Franklin, P. F. Hessburg, and J. A. Lutz. 2013. Restoring forest resilience: from reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management 291:442–457.

Collins, B. M., S. L. Stephens, J. J. Moghaddas, and J. Battles. 2010. Challenges and approaches in planning fuel treatments across fire-excluded forested landscapes. Journal of Forestry 108:24–31.

Collins, B. M., S. L. Stephens, G. B. Roller, and J. J. Battles. 2011. Simulating fire and forest dynamics for a landscape fuel treatment project in the Sierra Nevada. Forest Science 57:77–88.

Collins, B. M., H. A. Kramer, K. Menning, C. Dillingham, D. Saah, P. A. Stine, and S. L. Stephens. 2013. Modeling hazardous fire potential within a completed fuel treatment network in the northern Sierra Nevada. Forest Ecology and Management 310:156–166.

Crosby, J. S., and C. C. Chandler. 1996. Get the most from your windspeed observation. Fire Control Notes 27:12–13.

Earles, J. M., M. P. North, and M. D. Hurteau. 2014. Wildfire and drought dynamics destabilize carbon stores of fire-suppressed forests. Ecological Applications 24:732–740.

Finney, M. A. 2002. Fire growth using minimum travel time methods. Canadian Journal of Forest Research 32:1420–1424.

Finney, M. A. 2004. FARSITE: fire area simulator-model development and evaluation. USDA Forest Service, Rocky Mountain Research Station, Logan, Utah, USA.

Finney, M. A. 2007. A computational method for optimising fuel treatment locations. International Journal of Wildland Fire 16:702–711.

Finney, M. A., R. C. Seli, C. W. McHugh, A. A. Ager, B. Bahro, and J. K. Agee. 2007. Simulation of long-term landscape-level fuel treatment effects on large wildfires. International Journal of Wildland Fire 16:712–727.

Goodrick, S. L., G. L. Achtemeier, N. K. Larkin, Y. Liu, and T. M. Strand. 2013. Modelling smoke transport from wildland fires: a review. International Journal of Wildland Fire 22:83–94.

Greenberg, J. A., S. Z. Dobrowski, and S. L. Ustin. 2005. Shadow allometry: estimating tree structural parameters using hyperspatial image analysis. Remote Sensing of Environment 97:15–25.

Hessburg, P. F., et al. 2016. Tamm Review: management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. Forest Ecology and Management 366:221–250.

Hijmans, R., and J. van Etten. 2014. raster: geographic data analysis and modeling. R package version 517:2.2-12. http://CRAN.R-project.org/package=raster

Jaffe, D. A., N. Wigder, N. Downey, G. Pfister, A. Boynard, and S. B. Reid. 2013. Impact of wildfires on ozone exceptional events in the western U.S. Environmental Science & Technology 47:11065–11072.

Larkin, N. K., S. M. O'Neill, R. Solomon, S. Raffuse, T. Strand, D. C. Sullivan, C. Krull, M. Rorig, J. Peterson, and S. A. Ferguson. 2009. The BlueSky

smoke modeling framework. International Journal of Wildland Fire 18:906–920.

Loehle, C. 2004. Applying landscape principles to fire hazard reduction. Forest Ecology and Management 198:261–267.

Long, J. W., L. Quinn-Davidson, and C. N. Skinner. 2014. Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. General Technical Report PSW-GTR-247. USDA Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Loudermilk, E. L., A. Stanton, R. M. Scheller, T. E. Dilts, P. J. Weisberg, C. Skinner, and J. Yang. 2014. Effectiveness of fuel treatments for mitigating wildfire risk and sequestering forest carbon: a case study in the Lake Tahoe Basin. Forest Ecology and Management 323:114–125.

Lydersen, J., and M. North. 2012. Topographic variation in structure of mixed-conifer forests under an active-fire regime. Ecosystems 15:1–13.

Lydersen, J. M., M. P. North, and B. M. Collins. 2014. Severity of an uncharacteristically large wildfire, the Rim Fire, in forests with relatively restored frequent fire regimes. Forest Ecology and Management 328:326–334.

Manley, P. N., D. D. Murphy, B. M. Pavlik, A. E. Stanton, T. G. Biasiolli, A. M. White, M. D. Timmer, B. M. Collins, and G. B. Roller. 2012. Lake Tahoe Upland Fuels Research Project: investigating the effects of fuels reduction treatments on forest structure, fire risk, and wildlife. Final Report to U.S. Department of Interior, Bureau of Land Management, Pacific Southwest Research Station, Albany, California, USA. http://fs.fed.us/psw/partnerships/tahoescience/documents/p017_UplandFuelsFinalReport.pdf

Maxwell, R. S., A. H. Taylor, C. N. Skinner, H. D. Safford, R. E. Isaacs, C. Airey, and A. B. Young. 2014. Landscape-scale modeling of reference period forest conditions and fire behavior on heavily logged lands. Ecosphere 5:art32. http://dx.doi.org/10.1890/ES13-00294.1

Miller, J. D., and B. Quayle. 2015. Calibration and validation of immediate post-fire satellite derived data to three severity metrics. Fire Ecology 11:12–30.

Moghaddas, J. J., B. M. Collins, K. Menning, E. E. Y. Moghaddas, and S. L. Stephens. 2010. Fuel treatment effects on modeled landscape-level fire behavior in the northern Sierra Nevada. Canadian Journal of Forest Research 40:1751–1765.

Murphy, K., T. Rich, and T. Sexton. 2007. An assessment of fuel treatment effects on fire behavior, suppression effectiveness, and structure ignition on the Angora Fire. R5-TP-025. USDA Forest Service, South Lake Tahoe, California, USA.

Nelson, M. L., C. K. Brewer, and S. J. Solem, editors. 2015. Existing vegetation classification, mapping, and inventory technical guide. Version 2.0. General Technical Report WO-90. USDA Forest Service, Gen. Tech. Report WO-90, Washington, DC, USA.

North, M., P. Stine, K. O'Hara, W. Zielinski, and S. L. Stephens. 2009. An ecosystem management strategy for Sierran mixed-conifer forests. General Technical Report PSW-GTR-220. USDA Forest Service, Pacific Southwest Research Station, Albany, California, USA.

North, M., R. N. Boynton, P. A. Stine, K. F. Shipley, E. C. Underwood, N. E. Roth, J. H. Viers, and J. F. Quinn. 2012a. Geographic information system landscape analysis using GTR 220 concepts. Pages 107–115 in M. North, editor. Managing Sierra Nevada Forests. General Technical Report PSW-GTR-237. USDA Forest Service, Pacific Southwest Research Station, Albany, California, USA.

North, M., B. M. Collins, and S. Stephens. 2012b. Using fire to increase the scale, benefits, and future maintenance of fuels treatments. Journal of Forestry 110:392–401.

North, M., A. Brough, J. Long, B. Collins, P. Bowden, D. Yasuda, J. Miller, and N. Sugihara. 2015a. Constraints on mechanized treatment significantly limit mechanical fuels reduction extend in the Sierra Nevada. Journal of Forestry 113:40–48.

North, M. P., S. L. Stephens, B. M. Collins, J. K. Agee, G. Aplet, J. F. Franklin, and P. Z. Fulé. 2015b. Reform forest fire management. Science 349: 1280–1281.

Ottmar, R. D. 2014. Wildland fire emissions, carbon, and climate: modeling fuel consumption. Forest Ecology and Management 317:41–50.

Ottmar, R. D., and H. D. Safford. 2011. FCCS fuelbeds for The Lake Tahoe Basin Management Unit. USDA Forest Service, http://www.fs.fed.us/pnw/fera/fccs/supplementary_fuelbeds/pathways/lake_tahoe_fuelbed_handbook.pdf

Reinhardt, E. D., R. E. Keane, D. E. Calkin, and J. D. Cohen. 2008. Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. Forest Ecology and Management 256:1997–2006.

Safford, H. D., and J. T. Stevens. In press. Natural Range of Variation (NRV) for yellow pine and mixed conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. General Technical Report. USDA Forest Service, Pacific Southwest Research Station, Albany, California, USA.

Schmidt, D. A., A. H. Taylor, and C. N. Skinner. 2008. The influence of fuels treatment and landscape arrangement on simulated fire behavior, Southern

Cascade range, California. Forest Ecology and Management 255:3170–3184.

Schultz, C. A., T. Jedd, and R. D. Beam. 2012. The Collaborative Forest Landscape Restoration Program: a history and overview of the first projects. Journal of Forestry 110:381–391.

Schweizer, D. W., and R. Cisneros. 2016. Forest fire policy: change conventional thinking of smoke management to prioritize long-term air quality and public health. Air Quality, Atmosphere & Health. http://dx.doi.org/10.1007/s11869-016-0405-4

Scott, J. H., and R. E. Burgan. 2005. Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. General Technical Report RMRS-153. USDA Forest Service, Rocky Mountain Research Station, Logan, Utah, USA.

Sollman, R., A. M. White, G. L. Tarbill, P. N. Manley, and E. E. Knapp. 2016. Landscape heterogeneity compensates for fuel reduction treatment effects on Northern flying squirrel populations. Forest Ecology and Management 373:100–107.

Stephens, S. L., et al. 2014a. California spotted owl, songbird, and small mammal responses to landscape fuel treatments. BioScience 64:893–906.

Stephens, S. L., et al. 2014b. Temperate and boreal forest mega-fires: characteristics and challenges. Frontiers in Ecology and the Environment 12:115–122.

Stevens, J. T., H. D. Safford, and A. M. Latimer. 2014. Wildfire-contingent effects of fuel treatments can promote ecological resilience in seasonally dry conifer forests. Canadian Journal of Forest Research 44:843–854.

Stevens, J. T., H. D. Safford, S. Harrison, and A. M. Latimer. 2015. Forest disturbance accelerates thermophilization of understory plant communities. Journal of Ecology 103:1253–1263.

Taylor, A. H., A. M. Vandervlugt, R. S. Maxwell, R. M. Beaty, C. Airey, and C. N. Skinner. 2014. Changes in forest structure, fuels and potential fire behaviour since 1873 in the Lake Tahoe Basin, USA. Applied Vegetation Science 17:17–31.

Tempel, D. J., R. J. Gutiérrez, S. A. Whitmore, M. J. Reetz, R. E. Stoelting, W. J. Berigan, M. E. Seamans, and M. Z. Peery. 2014. Effects of forest management on California Spotted Owls: implications for reducing wildfire risk in fire-prone forests. Ecological Applications 24:2089–2106.

Tempel, D. J., et al. 2015. Evaluating short- and long-term impacts of fuels treatments and simulated wildfire on an old-forest species. Ecosphere 6:art261. http://dx.doi.org/10.1890/ES15-00234.1

Truex, R. L., and W. J. Zielinski. 2013. Short-term effects of fuel treatments on fisher habitat in the Sierra Nevada, California. Forest Ecology and Management 293:85–91.

Vaillant, N. M., J. Fites-Kaufman, A. L. Reiner, E. K. Noonan-Wright, and S. N. Dailey. 2009. Effect of fuel treatments on fuels and potential fire behavior in California, USA, National Forests. Fire Ecology 5:14–29.

Vogler, K. C., A. A. Ager, M. A. Day, M. Jennings, and J. D. Bailey. 2015. Prioritization of forest restoration projects: tradeoffs between wildfire protection, ecological restoration and economic objectives. Forests 6:4375.

White, A. M., E. F. Zipkin, P. N. Manley, and M. D. Schlesinger. 2013a. Conservation of avian diversity in the Sierra Nevada: moving beyond a single-species management focus. PLoS ONE 8:e63088.

White, A. M., E. F. Zipkin, P. N. Manley, and M. D. Schlesinger. 2013b. Simulating avian species and foraging group responses to fuel reduction treatments in coniferous forests. Forest Ecology and Management 304:261–274.

Winter, G. J., C. Vogt, and J. S. Fried. 2002. Fuel treatments at the wildland-urban interface: common concerns in diverse regions. Journal of Forestry 100:15–21.

Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.1445/supinfo