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Landscape-scale fuel treatment and wildfire impacts on carbon stocks and fire hazard in California spotted owl habitat

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Abstract. Forest managers are challenged with meeting numerous demands that often include wildlife habitat and carbon (C) sequestration. We used a probabilistic framework of wildfire occurrence to (1) estimate the potential for fuel treatments to reduce fire risk and hazard across the landscape and within protected California spotted owl (*Strix occidentalis occidentalis*) habitat and (2) evaluate the consequences of treatments with respect to terrestrial C stocks and burning emissions. Silvicultural and prescribed fire treatments were simulated on 20% of a northern Sierra Nevada landscape in three treatment scenarios that varied in the land area eligible for treatment. Treatment prescriptions varied with topography, vegetation characteristics, and ownership. We then simulated many wildfires in the treated and untreated landscapes. Additional simulations allowed us to consider the influence of wildfire size on estimated emissions. Treatments constrained to the land area outside of spotted owl activity centers reduced the probability of burning and potential fire intensity within owl habitat and across the landscape relative to no-treatment scenarios. Allowing treatment of the activity centers achieved even greater fire hazard reductions within the activity centers. Treatments also reduced estimated wildfire emissions of C by 45–61%. However, emissions from prescribed burning exceeded simulated reductions in wildfire emissions. Consequently, all treatment scenarios resulted in higher C emissions than the no-treatment scenarios. Further, for wildfires of moderate size (714–2133 ha), the treatment scenarios reduced the C contained in live tree biomass following simulated wildfire. When large wildfires (8070–10,757 ha) were simulated, however, the treatment scenario retained more live tree C than the no-treatment scenario. Our approach, which estimated terrestrial C immediately following wildfire, did not account for long-term C dynamics, such as emissions associated with post-wildfire decay, C sequestration by future forest growth, or longer-term C sequestration in structural wood products. While simulated landscape fuel treatments in the present study reduced the risk of uncharacteristically severe wildfire across the landscape and within protected habitat, the C costs of treatment generally exceeded the C benefits.

Key words: ArcFuels; California spotted owl; forest thinning; prescribed fire; *Strix occidentalis occidentalis;* wildfire emissions.

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Introduction

Forest managers in fire-prone ecosystems seek to balance a complex set of sometimes competing objectives that include providing wildlife habitat, avoiding catastrophic disturbance, and supporting local economies. In recent years, maintaining and increasing the capacity of forests to store carbon (C) has been added to these considerations due to concern over the effects of rising atmospheric greenhouse gas concentrations on the earth's climate. In dry forests across much of the western United States, meeting these objectives is complicated by the increasing area and severity of wildfires occurring in concert with climate change (McKenzie et al. 2004, Stephens 2005, Westerling et al. 2006, Miller et al. 2009).

A high-visibility example of competing objectives in forest management is spotted owl (*Strix occidentalis occidentalis*) conservation in California. The northern (*S. occidentalis caurina*) and Mexican (*S. occidentalis lucida*) spotted owl subspecies have been listed as Threatened under the Endangered Species Act. Management directives for the California subspecies focus on conserving nesting and roosting habitat by identifying protected activity centers (PACs): sites that include 121 ha (300 ac) of the best-quality habitat near known nest sites (Verner et al. 1992). Given the multi-storied, dense canopy forest characteristics of nesting and roosting sites, the potential vulnerability of PACs to high-severity fire is a challenge to owl conservation (Collins et al. 2010, Stephens et al. 2016*b*). While low- to moderate-severity wildfire within nesting and roosting habitat may not negatively impact owls in the short term (Bond et al. 2002), longer-term effects of high-severity wildfire can include significant habitat loss due to direct and indirect tree mortality (Gaines et al. 1997, Jones et al. 2016, Stephens et al. 2016*b*). However, due to uncertainty concerning the effects of fuels reduction activities, management options for reducing wildfire hazard within PACs are restricted to light prescribed burning, although some thinning is permitted in the wildland–urban interface (USDA Forest Service 2004).

There is concern that such constraints on management activities limit the effectiveness of landscape-scale treatments intended to reduce the threat of uncharacteristically severe wildfire (Collins et al. 2010, Tempel et al. 2015). Fire modeling studies have shown that treating a portion of the landscape can alter simulated fire behavior within and outside of treated areas and that strategically locating fuel treatments across the landscape has the potential to maximize treatment benefits while minimizing area treated (Finney et al. 2007, Schmidt et al. 2008). Restrictions on fuel treatment location and severity limit real-world application of treatment optimization methods. Even so, there may be significant opportunity for active management outside of high-quality owl habitat on fire-prone landscapes (Ager et al. 2007, Prather et al. 2008, Gaines et al. 2010).

Given their demonstrated ability to alter wildfire behavior and effects (Martinson and Omi 2002, Pollet and Omi 2002, Ritchie et al. 2007, Fulé et al. 2012), fuel treatments that address accumulated fuels and reduce stand density (e.g., prescribed burning, forest thinning, mastication) are commonly applied in dry western forests where wildfires were once frequent. It is less certain how treatments influence C stocks, and how to maximize C storage in frequent-fire systems. In the absence of disturbance, untreated forests may sequester the most C (Hurteau and North 2009, Stephens et al. 2009, Hurteau et al. 2011). However, high-severity wildfires can rapidly convert C sinks to sources, and burned forests may continue to be C sources for decades (Dore et al. 2008, 2012). Treatments can reduce wildfire emissions (Finkral and Evans 2008, Hurteau and North 2009, 2010, North et al. 2009*a*, Reinhardt and Holsinger 2010, Wiedinmyer and Hurteau 2010, North and Hurteau 2011) and may retain more live tree C post fire (Hurteau and North 2009, North and Hurteau 2011, Stephens et al. 2012). Yet fuel treatments are associated with significant C emissions, releasing C to the atmosphere during harvest operations, burning, and/or biomass transport, and the C cost of treating forest fuels may exceed its C benefits (Campbell et al. 2011, Campbell and Ager 2013). The circumstances under which treatments might lead to a net gain in C have not yet been resolved.

Recently, as a result of concern over the C costs of fossil fuel use and the threat of wildfire, interest in harvesting historically low-value woody biomass has increased (Evans and Finkral 2009). Utilizing forest biomass for energy production can help to reduce the cost of fuel treatments, support local economies, offset fossil fuel use, and reduce the C and smoke emissions

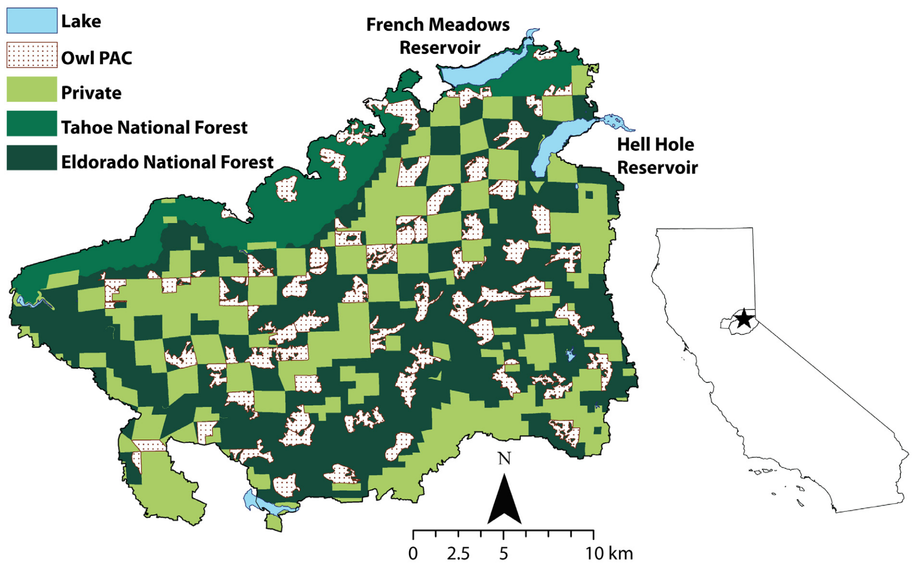

Fig. 1. Study area in Tahoe and Eldorado counties, northern Sierra Nevada, California. Land ownership and owl protected activity center (PAC) locations.

associated with fuel treatments (Reinhardt et al. 2008). Concerns remain over the sustainability of biomass removals, funding, and the availability of markets (Evans and Finkral 2009).

The focus of our research was to (1) evaluate whether withholding some land area from treatment influences potential wildfire hazard across the landscape and within California spotted owl habitat, (2) estimate the short-term C consequences of treatments, and (3) quantify the biomass harvested in treatments. We simulated fuels reduction treatments and wildfire in a northern Sierra Nevada study area that encompassed 61 spotted owl PACs. In order to evaluate the C balance of the treatment scenarios, we quantified the C contained in the forest biomass harvested in each treatment scenario, the C emitted during prescribed fire and wildfires, and the C remaining within onsite pools. We confined our analysis to the immediate changes in C stocks and emissions, but recognize that a full accounting of treatment effects would also include long-term C dynamics (e.g., Dore et al. 2008, Malmsheimer et al. 2011).

Methods

Study area

The study area was defined by a long-term demographic study site for the California spotted owl (*S. occidentalis occidentalis*). The 55,398-ha area contains 61 owl PACs. The study area is located ~20 km west of Lake Tahoe in the northern Sierra Nevada, with elevation ranging from 300 to 2400 m. The climate is Mediterranean, with warm, dry summers and cool, wet winters. Vegetation at lower elevations in the study area is montane mixed-conifer forest. The forest type is dominated by ponderosa pine (*Pinus ponderosa* Dougl.), Douglas-fir (*Pseudotsuga menziesii* var. *menziesii* (Mirb.), sugar pine (*Pinus lambertiana* Dougl.), incense-cedar (*Calocedrus decurrens* [Torr.] Florin.), white fir (*Abies concolor* (Gord. and Glend.)), Franco), and California black oak (*Quercus kelloggii* Newb.). California red fir (*Abies magnifica* var. *magnifica* Andr. Murray) has a stronger presence above ~2000 m (Barbour and Minnich 2000), but the red fir forest type is present on only ~5% of our core study area.

One-third of the study area is privately held in a generally checkerboard pattern of ownership (Fig. 1). The remaining 37,120 ha is managed by the Tahoe and Eldorado National Forests. Young forests dominate private land in the study area due to historical and active logging, while intermediate and mature forests are relatively abundant on public land (Laymon 1988, Bias and Gutiérrez 1992).

Table 1. Description of Chatfield (2005) cover classes.

Cover class	Description
1	Hardwood forest (>10% hardwood canopy closure and <10% conifer canopy closure)
2	Clearcut or shrub/small tree (<15.3 cm dbh)
3	Pole (15.3–28 cm dbh) forest
4	Medium (28–61 cm dbh) conifer/mixed-conifer forest with low to medium canopy closure (30–69%)
5	Medium (28–61 cm dbh) conifer/mixed-conifer forest with high canopy closure (≥70%)
6	Mature (≥61 cm dbh) conifer/mixed-conifer forest with low to medium canopy closure (30–69%)
7	Mature (≥61 cm dbh) conifer/mixed-conifer forest with high canopy closure (≥70%)
8	Water

Vegetation and fuels data

The vegetation classification map developed in Chatfield (2005) forms the basis of our study area. Using aerial photographs combined with field accuracy assessment, Chatfield (2005) digitized eight land cover classes consistent with the California Wildlife Habitat Relationships (CWHR; Mayer and Laudenslayer 1988) system. A description of the cover classes is provided in Table 1. From the resulting cover class map, we delineated polygons to represent stands of similar vegetation composition and structure (n = 4470) based on aerial photographs and topography (Fig. 2).

Stands were populated with vegetation data collected in 2007 in 382 sampling plots located within 10 km of the study area's northern boundary, based on the assumption that the characteristics of the plots are representative of the study area. These vegetation data included tree species, heights, diameters, and crown ratios. See Collins et al. (2011) for a detailed description of data collection. To populate stands in the core study area with plot data, we first assigned a Chatfield cover class to each sampling plot based on species composition, canopy cover, and tree diameter distribution. We then used a Most Similar Neighbor procedure (Crookston et al. 2002) to select five nearest neighbor plots for each stand using the Random Forest method with the R package *yaimpute* (version 1.0-22; Crookston and Finley 2008). Variables used in identifying nearest neighbors were topographic relative moisture index, eastness, northness, slope, and elevation. Stands were populated with data only from plots belonging to the same cover class. In order to increase variability in stand conditions, three of the five plots initially selected to represent each stand were chosen randomly to contribute data to the stand. Each plot contributed data to an average of 35.5 stands (range: 1–437).

The method in which surface fuels are represented for fire modeling has important implications for findings related to expected fire behavior and effects. Fuel models are representations of fuelbed properties such as the distribution of fuel between particle size classes, heat content, and dead fuel moisture of extinction for use in the Rothermel (1972) surface fire spread model. As representations, fuel models artificially constrain the variation in surface fuel conditions. In order to represent a range of pretreatment fuel conditions for fire modeling, we overrode fuel model assignments made by the Fire and Fuels Extension to the Forest Vegetation Simulator (FVS-FFE, Dixon 2002, Reinhardt and Crookston 2003) and selected two fuel models for each stand. Fuel models representing the low end of the range were assigned following the selection logic of Collins et al. (2011); high-end models were selected to amplify surface fire behavior relative to the low-end models (Appendix S1: Table S1; Collins et al. 2013). This approach to assigning fuel models to stands has been demonstrated to result in modeled fire behavior that is more consistent with observed fire effects than default fuel model assignments (Collins et al. 2013). An alternative approach could be to use the Landfire surface fuel model layer (e.g., Scott et al. 2016). However, we opted to tie fuel model assignments to the specific forest structural characteristics for each stand (Lydersen et al. 2015) as represented by the imputed plots rather than the remotely sensed dominant vegetation characteristics captured by Landfire.

Study area data were processed in the western Sierra variant of FVS to obtain the data layers required for fire behavior modeling. Due to the potential for spurious fire modeling results near study area edges, we obtained additional canopy fuel and surface fuel data layers from Landfire (www.landfire.gov) for an area adjacent to the study area boundary defined by a 10-km minimum bounding rectangle (Fig. 2). The reason for using Landfire data for the buffer area was that

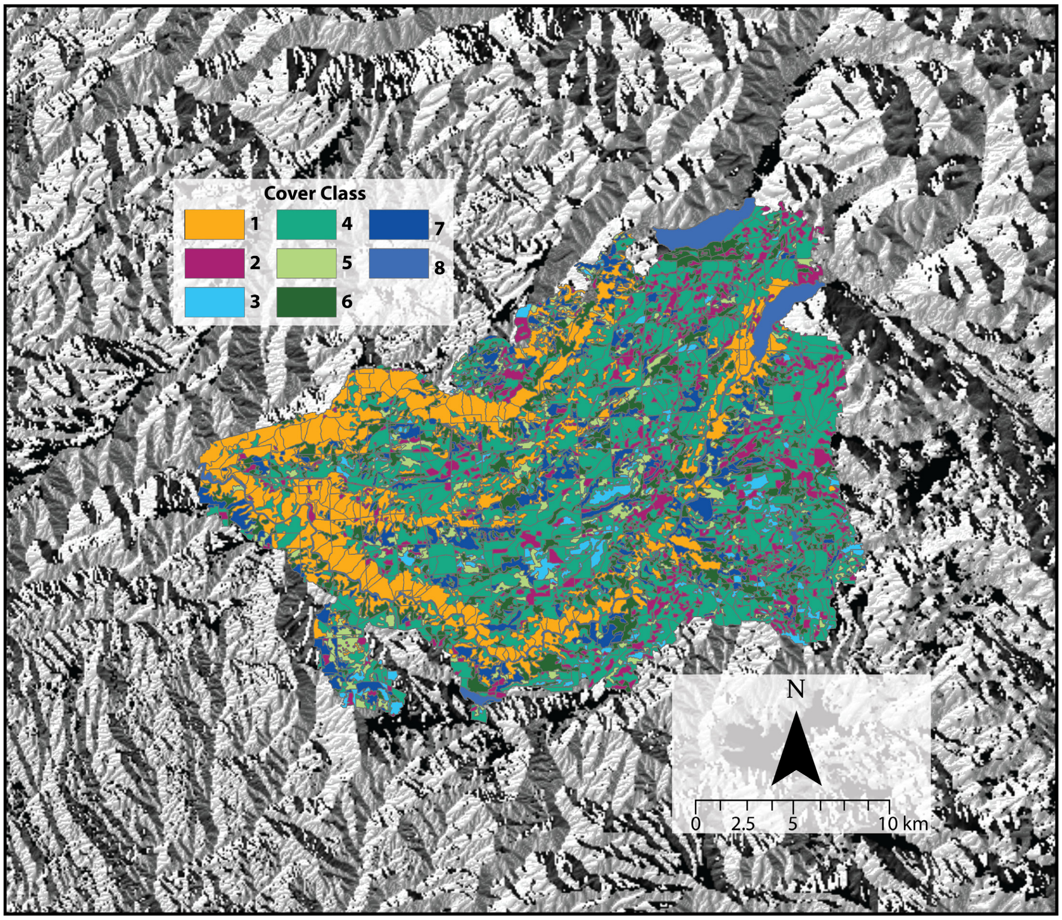

Fig. 2. Land cover classes (Chatfield 2005) within the core study area, stand polygons, and 10-km minimum bounding rectangle for fire spread modeling. See Table 1 for description of classes.

we did not have a vegetation map with a similar classification scheme and level of detail outside of our core study area (Fig. 2). We merged study area and Landfire data layers to build 90 × 90 m resolution landscape files for fire behavior modeling in Randig, described below. This allowed us to include wildfires originating outside of the study area in our analysis.

Wildfire, fuel treatments, and carbon loss modeling

We used ArcFuels (Ager et al. 2006) to streamline fuel treatment planning and analysis of effects. ArcFuels is a library of ArcGIS macros that facilitates communication among the array of models and other programs commonly used in fuel treatment planning at the landscape scale (vegetation growth and yield simulators, fire behavior models, ArcGIS, and desktop software). Our process, depicted in Fig. 3, involved:

1. fire behavior modeling in Randig (Finney 2006) to identify stands with high fire hazard;
2. prioritizing stands for treatment using the Landscape Treatment Designer (LTD) (Ager et al. 2012);
3. modeling fuel treatments in FVS-FFE;
4. fire behavior modeling for the post-treatment and untreated landscapes; and
5. developing C loss functions from simulated burning with FVS-FFE.

Conditional burn probability and flame length.— Wildfire growth simulations were performed in Randig, a command-line version of FlamMap (Finney 2006). Randig uses the minimum travel

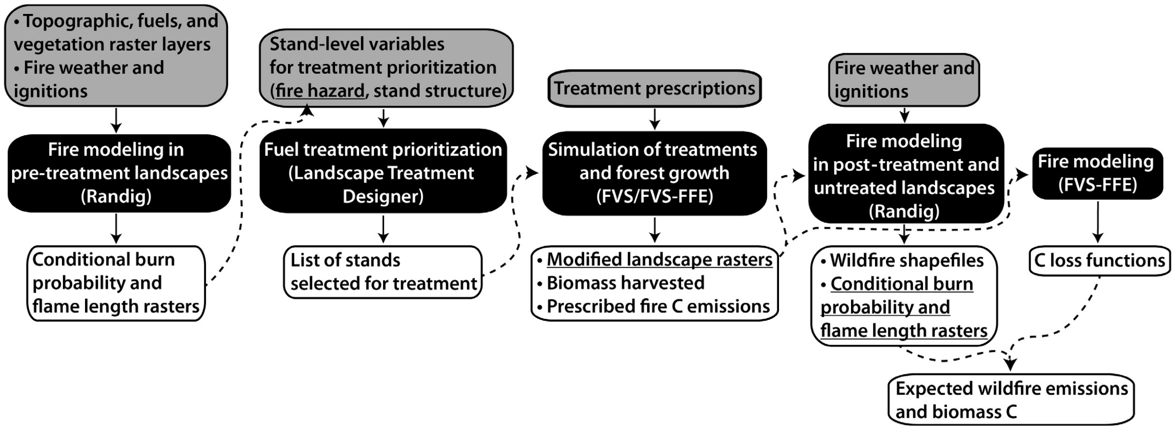

Fig. 3. Work flow used in the present study to evaluate landscape fuel treatment effects on wildfire hazard and carbon pools and emissions.

time algorithm (Finney 2002) to simulate fire growth during discrete burn periods under constant weather conditions. Simulating many burn periods with Randig generates a burn probability surface for the study landscape. Simulations were conducted at 90-m resolution for computational efficiency. We simulated 80,000 randomly located ignitions with a 5-h burn period for all scenarios, including no treatment. The burn period was selected to produce fire sizes that approximated area burned in spread events of historical large wildfires near the study area. Large daily spread events in previous wildfires in the northern Sierra Nevada have burned >2000 ha (Dailey et al. 2008, Safford 2008); average fire sizes from our simulations ranged from 715 to 2133 ha. (The exceptional growth observed in the 2014 King Fire is addressed in a subsequent subsection.) The combination of ignition number and burn period was sufficient to ensure that 99% of pixels in burnable fuel types experienced fire at least once (average: 64–1891 fires).

Randig outputs were used both in prioritizing stands for treatment and in evaluating the effects of treatment. We performed one Randig run for each fuel model range (low and high) within each scenario (no treatment, S1, S2, and S3) using landscape files representing the year immediately following treatment, 2009. Simulations were also completed for the 2007 pre-treatment landscape for use in treatment prioritization, for a total of 10 modeling runs.

To evaluate the effect of treatments on fire risk and fire hazard, we assessed changes in conditional burn probability (CBP) and conditional flame length (CFL) between the treatment scenarios and the untreated landscape based on wildfire simulations. It is important to note that the burn probabilities estimated in this study are not empirical estimates of the likelihood of wildfire occurrence (e.g., Preisler et al. 2004, Brillinger et al. 2006, Parisien et al. 2012). Rather, we use CBP, the likelihood that a pixel will burn given a single ignition in the study area, and assuming the simulation conditions described. From the simulation of many fires, Randig calculates a pixel-level distribution of flame lengths (FL) in twenty 0.5-m classes between 0.5 and 10 m. Conditional flame length, the probability-weighted FL given that a fire occurs (Ager et al. 2010), was calculated by combining burn probability estimates with FL distributions summarized at the stand level:

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

where BP is CBP, BP_i is the probability of burning at the ith FL class, and F_i is the midpoint FL of the ith FL class.

To estimate the effect of treatment on fire risk and hazard, we first computed average pixel-level BP and CFL for treated and untreated stands in each scenario. Then, we calculated average BP and CFL for the same stands within the no-treatment landscape. The effect of each treatment scenario was estimated as the proportional change in each fire metric between the untreated and treated landscapes.

We obtained weather and fuel moisture inputs for wildfire modeling from the Bald Mountain and Hell Hole remote automated weather stations (RAWS), based on recommendations from local USDA Forest Service fire and fuel managers. We used 95th percentile weather conditions from the 1 June to 30 September period (1989–2013). This period represents the typical fire season for the study area, encompassing 85% of wildfires and 93% of the area burned within a 161-km (100-mi) radius of the study area between 1984 and 2012 (Monitoring Trends in Burn Severity database, Eidenshink et al. 2007).

Weather and fuel moisture inputs for wildfire simulations are provided in Appendix S1: Table S2. These conditions are similar to those occurring during recent large wildfires in and near the study area (e.g., 2001 Star Fire, 2008 American River Complex, 2013 American Fire). In addition to using Randig to model fire spread and intensity, we used FVS-FFE to project effects of prescribed fires and wildfires (described below). Wind inputs varied somewhat between fire models: FVS-FFE requires only a single wind speed, while multiple wind scenarios were applied in Randig fire simulations. Wind speeds, azimuths, and relative proportions for Randig simulations followed Collins et al. (2011).

Spatial optimization of fuel treatments.—Stands were selected for treatment based on modeled pre-treatment wildfire hazard and stand structure using the LTD, which allows multiple objectives to be combined in the spatial prioritization of fuel treatments. Three treatment scenarios varied in the land designations eligible for treatment:

Scenario 1: Public land, excluding spotted owl habitat
Scenario 2: Public land, including spotted owl habitat
Scenario 3: All lands: public and private ownerships

Objectives were consistent across treatment scenarios, but differed in the land area available for treatment. For all LTD runs, we directed the model to maximize a total score that comprised numeric stand structure and fire hazard rankings (Appendix S1: Table S3). The stand structure ranking (0, 1, 2) was based on cover class category: Cover classes most conducive to thinning were ranked highest. Fire hazard ranking (0, 2, 3) was assigned according to stand-level CFL as calculated from FL probability files generated in Randig simulations for the 2007 pre-treatment landscape.

To isolate the effect of varying land designations in the area available for treatment, total area treated was held constant between scenarios (20% of the core study area). In order to exclude small, spatially isolated treatment areas that would be impractical from a management standpoint, we required a minimum treatment area of 12.1 ha (30 ac). To achieve this, the treatment prioritization process was iterative. In each step, we eliminated all stands selected by LTD for treatment that were not contiguous with a ≥12.1-ha treatment area. The rationale for this is based on the cost associated with re-locating equipment necessary to implement mechanical and/or fire treatments (D. Errington, *personal communication*, El Dorado National Forest). We then calculated the treatment area remaining. This process was repeated until total treatment area summed to the target area (~11,080 ha).

We simulated fuel treatments using FVS-FFE. Stands selected for treatment were assigned one of 13 treatment prescriptions depending on topography, vegetation cover class, ownership, and overlap with owl PACs (Appendix S1: Table S4). In an effort to promote landscape-scale heterogeneity, basal area targets for commercial thinning on public land varied with topography (aspect and slope position: canyon/drainage bottom, mid-slope, and ridge) (North et al. 2009*b*, North 2012). All thinning treatments were simulated as thin-from-below harvests, and thinning within owl PACs was limited to hand thinning. We assumed that trees ≥25.4 cm (10 in) dbh would be harvested for wood products (FVS VOLUME keyword) and that the biomass contained in smaller trees and in the tops and branches of larger trees would be utilized as feedstocks for bioenergy conversion. Therefore, all thinning (except hand thinning) treatments were simulated as whole tree harvests (FVS keyword YARDLOSS). Treatments preferentially retained fire-resistant species, with relative retention preference as follows: black oak>ponderosa pine>sugar pine>Douglas-fir>incense-cedar>red fire>white fir.

Prescribed fires were simulated in the year following thinning (2009). Broadcast burning was applied except within owl PACs, on private land, and on steep slopes (>35%), where follow-up burning was limited to pile burning. To capture a more realistic range of post-treatment surface fuel conditions, stands selected for treatment were randomly assigned to one of three post-treatment fuel models for each fuel model range: TL1 (181), TL3 (183), or TL5 (185) (low range); TL3 (183), TL5 (185), or SB1 (201) (high range) (Scott and Burgan 2005). Weather conditions for prescribed fire modeling were based on recommendations from a local fire management specialist (B. Ebert, *personal communication*).

Biomass and carbon effects of treatment.—Simulated treatment prescriptions varied according to site characteristics such as topography and land ownership (Appendix S1: Table S4). We tracked the C emitted from burning, removed during harvesting, and contained in live and dead aboveground biomass with FVS-FFE carbon reports (Reinhardt and Crookston 2003, Hoover and Rebain 2008). FVS converts biomass to units of C using a multiplier of 0.5 for all live and dead C pools (Penman et al. 2003) except duff and litter pools, for which a multiplier of 0.37 is applied (Smith and Heath 2002). Stand C is partitioned into a number of pools including aboveground live tree, standing dead tree, herb and shrub, litter and duff, woody surface fuel, and belowground live and dead tree root C; we limited our analysis to aboveground pools of C. FVS-FFE also reports the C emitted during burning and that contained in harvested biomass (Rebain et al. 2009). Treatment effects were assessed by comparing expected aboveground biomass C and emissions between the treated and untreated landscapes.

We developed C loss functions for each FVS treelist by simulating burning with FVS-FFE at a range of FLs (SIMFIRE and FLAMEADJ keywords) (Ager et al. 2010, Cathcart et al. 2010). The FL values supplied to FLAMEADJ were the 20 midpoints of the 0.5-m FL classes (0.5–10 m) found in Randig FL probability output files. As noted by Ager et al. (2010) and Cathcart et al. (2010), it is not currently possible to precisely match fire behaviors between Randig and FVS. The FLs reported in Randig outputs are the total of surface fire and, if initiated, crown fire. In contrast, the FLs supplied to FVS-FFE via the FLAMEADJ keyword are treated as surface fire FLs, and when FLAMEADJ is parameterized with only a predefined FL, the model does not use the input FL in crown fire simulations. To estimate fire effects in FVS-FFE, we parameterized FLAMEADJ with percent crowning (PC) and scorch height in addition to FL. Scorch height and critical FL for crown fire initiation (FLCRIT) were based on Van Wagner (1977). We estimated PC using a downward concave function where PC = 32% when flame length = FLCRIT and PC = 100% when FL is ≥30% of stand top height (the average height of the 40 largest trees by diameter) (Ager et al. 2010; A. Ager, *personal communication*).

The derived C loss functions were combined with the probabilistic estimates of surface fire behavior produced in Randig simulations to estimate the "expected C" emitted in wildfire or contained in biomass. We estimated expected C emissions and post-fire biomass C for each pixel as follows:

$$E[\mathrm{C}]_i = \sum_{i=0}^{20} \left[\mathrm{BP}_{ij} \times \mathrm{C}_{ij}\right]$$

where $E[\mathrm{C}]_j$ is the expected wildfire emissions of C from pixel j, or biomass C in pixel j, in mass per unit area; BP_{ij} is the probability of burning at the ith FL class for pixel j; and C_{ij} is the C emitted from pixel j, or the biomass C remaining in pixel j post-wildfire, given burning at the ith FL class.

Expected C emissions and biomass C were summed across all pixels in the core study area to obtain total expected wildfire emissions and expected terrestrial C for each treatment scenario.

In order to compare our modeling results to other analyses that reported wildfire emissions on a per area basis, we used a different method to estimate C emissions per area burned. Because wildfires burned both the core and buffer areas of our study landscape while emissions were estimated only in the core area, we used conditional expected wildfire emissions to approximate the emissions from a wildfire burning entirely within the core study area. Conditional expected emissions are those produced for an area given that the area is burned. Conditional emissions were estimated for each pixel as follows:

Table 2. Total area and proportion of area treated by category in each treatment scenario.

Treatment category	SC1 Area (ha)	SC1 Proportion of area treated	SC2 Area (ha)	SC2 Proportion of area treated	SC3 Area (ha)	SC3 Proportion of area treated
Treated	11,081	1.00	11,082	1.00	11,081	1.00
Avail. for treatment†	22,042	1.99	28,998	2.62	45,647	4.12
Owl habitat treated	0.0	0.00	2769	0.25	1127	0.10
Private land treated	0.0	0.00	0.0	0.00	5685	0.51
Hand thin	1499	0.14	3819	0.34	1612	0.15
Biomass thin	8404	0.76	7240	0.65	9470	0.85
Commercial thin	7765	0.70	6916	0.62	9470	0.85
Broadcast burn	9410	0.85	7247	0.65	3785	0.34
Pile burn	1671	0.15	3835	0.35	7296	0.66

† Total land area potentially available for treatment in each scenario. The area available for treatment increased from Scenarios 1 to 3 as restrictions on the area available for treatment were relaxed.

$$C[WC]_j = \sum_{i=1}^{20} \left[\frac{BP_{ij}}{BP_j} \times WC_{ij} \right]$$

where $C[WC]_j$ is the C emitted by wildfire from pixel j in mass per unit area; BP_j is the probability that pixel j is burned; BP_{ij} is the probability of burning at the ith FL class, and WC_{ij} is the C emitted from pixel j when burned at the ith FL class.

Conditional expected emissions were averaged across all pixels to obtain an estimate of wildfire emissions per area burned.

Large fire revision.—Wildfire modeling was calibrated to produce fire sizes that approximated area burned in spread events of historical large wildfires near the study area. However, during the course of the study, a very large fire encountered our study area. The King Fire began on 13 September 2014 in El Dorado County and burned 39,545 ha—more than an order of magnitude greater than our modeled wildfires, including >25% of the study area. Given the potential for very large wildfires in this region demonstrated by the King Fire, we completed additional wildfire modeling to estimate the C effects of treatment given the occurrence of a very large fire.

Randig modeling was repeated for the no-treatment and S3 scenarios using the high fuel model range and a revised burn period, number of simulated ignitions, wind speed, and wind directions. Burn period was increased from 5 to 12 h; number of ignitions was reduced by half to 40,000. Wind directions and relative probabilities (Appendix S1: Table S5) were those recorded at Hell Hole RAWS between 04:00 and 19:00 hours on 17 September, the day of the King Fire's largest spread event. We used the probable 1-min maximum wind speed as calculated from the maximum gust recorded on that day: 33 km/h (20.5 mph), based on maximum gust of 54.7 km/h (34 mph) (Crosby and Chandler 1966). These settings produced average fire sizes of NT = 10,757 ha (no-treatment scenario) and 8070 ha (S3). Average fire size was limited by the size of our buffered study area: Longer burn periods resulted in an increasing number of simulated wildfires that burned to the study area boundary.

Results

Treatment simulation

Table 2 provides a summary of the area treated in each scenario. Scenario 1 (S1) was the most restrictive with respect to the land area available for treatment, which more than doubled between S1 and S3. Because treatment prescriptions varied with land designation (public, owl PAC, private), and the designations available for treatment varied between scenarios, the relative proportions of thinning and burning methods also varied between scenarios. Commercial and biomass thinning were applied most frequently in S3, which permitted treatment of private land. Spotted owl activity centers composed 25% of the area treated in S2 vs. 10% in S3 and 0% in S1, the scenario in which PACs were not subject to treatment. As a result, the area treated with hand thinning in S2 was more than twice that in S1 and S3. Due to the

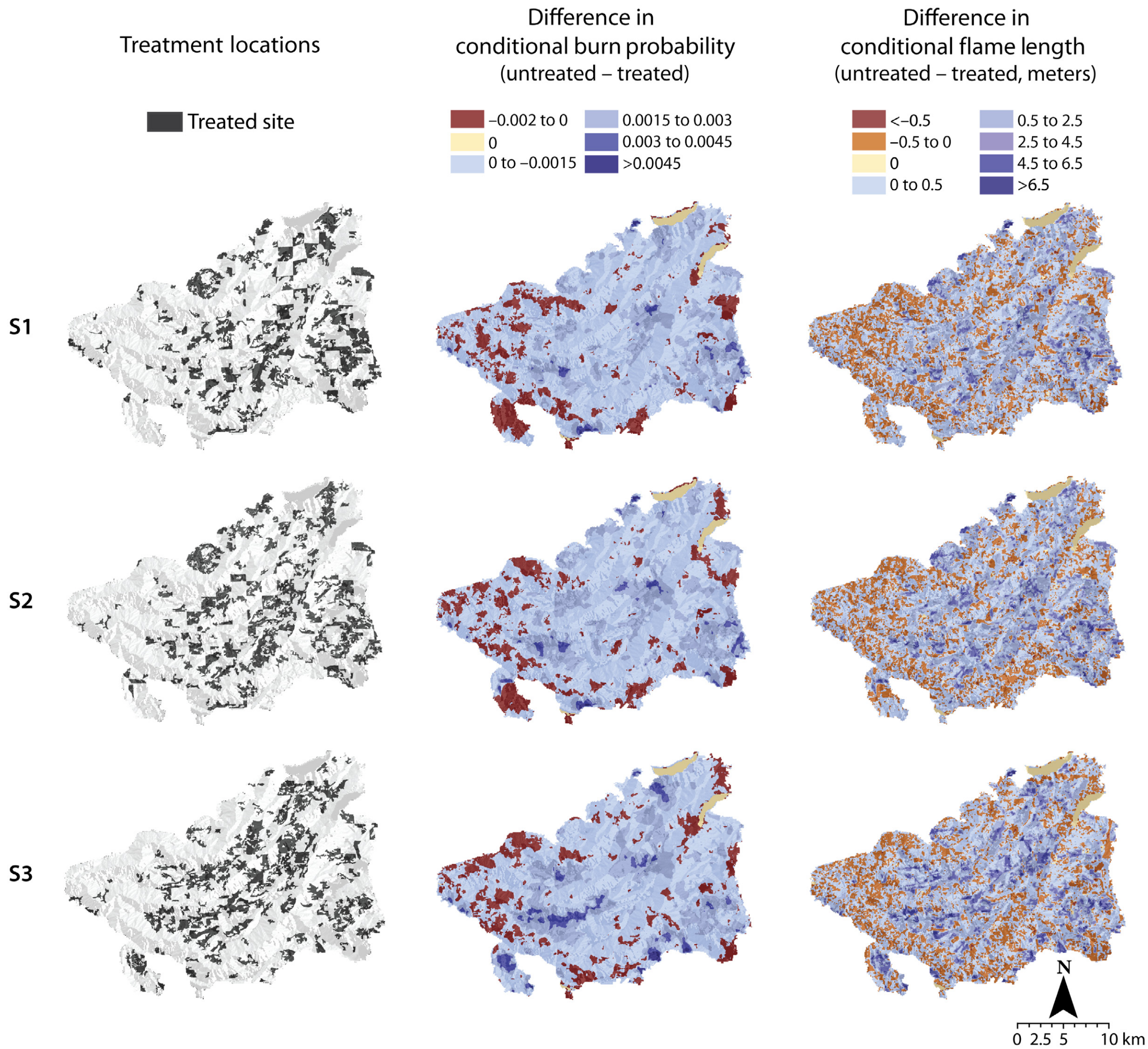

Fig. 4. Low fuel model range treatment locations and difference in conditional burn probability (CBP) and conditional flame length (CFL) (untreated-treated) for each treatment scenario. Negative values indicate an increase in CBP or CFL, while positive values represent a reduction. CBP is the likelihood that a pixel will burn given a single ignition on the landscape and assuming the simulation conditions described in Appendix S1: Table S1 and in the text. Conditional flame length is the probability-weighted flame length, given these same assumptions.

inclusion of PACs in S2 and both PACs and private land in S3, the proportion of area treated with pile burning increased between S1 and S3, while broadcast burn area exhibited an opposite trend. Despite the variation in land designations available for treatment, the pattern of treatment placement was generally similar between scenarios, with treatments concentrated in the central and eastern portions of the study area (Figs. 4, 5).

Landscape-scale burn probability and fire hazard

*Conditional burn probability.—*The pixel-to-pixel change in CBP between the untreated scenario and each treatment scenario is mapped in Figs. 4 (LO FM) and 5 (HI FM). Treatment reduced landscape burn probability by approximately 50% (Table 3), from 0.0124 (NT) to 0.0062 (S1), 0.0059 (S2), and 0.0055 (S3). Within treatment units, average CBP fell by 69–76% to 0.0033–0.0035; outside of treated stands, CBP fell to 0.0060–0.0069. Some

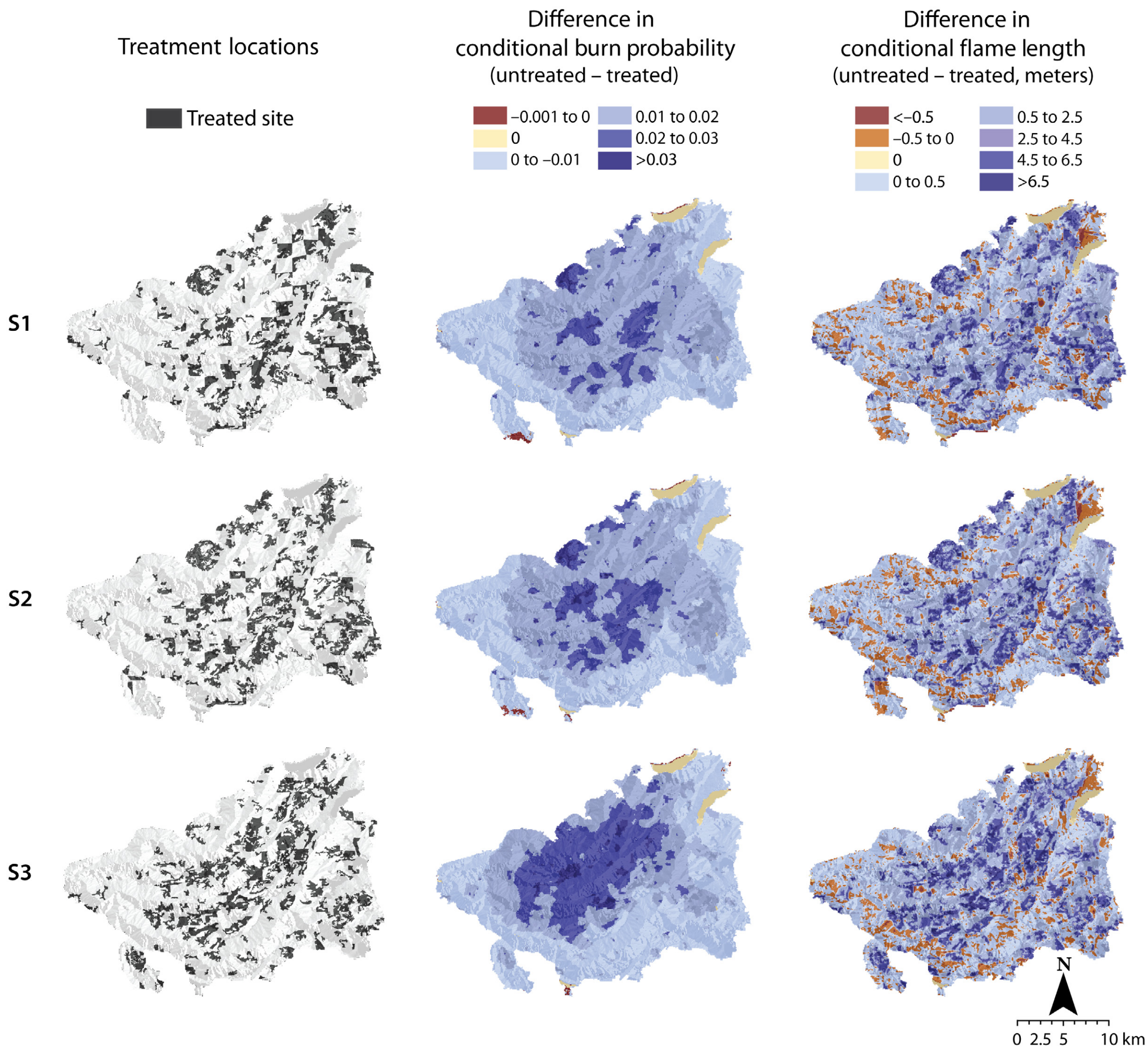

Fig. 5. Treatment locations and high fuel model range difference in conditional burn probability and conditional flame length (untreated-treated) for each treatment scenario.

increases in CBP were also observed, particularly for the low fuel model range (Fig. 4).

The influence of treatment on owl PAC likelihood of burning was similar to that observed for stands in general. For treated PACs, average CBP fell by ~70% relative to no treatment for the same stands. Although PACs were not eligible for treatment in S1, all treatment scenarios had a large impact on estimated PAC CBP. Average PAC CBP was reduced from 0.013 to 0.0063 in S1, 0.0049 in S2, and 0.0054 in S3, a 49–64% decrease relative to PACs in the no-treatment landscape (Table 3).

Fire hazard.—Treatments reduced average landscape CFL by ~1 m, from 3.6 m (NT) to 2.5–2.7 m. Pixel-level CFL was reduced by a maximum of 8.0 m (LO FM) and 9.0 m (HI FM). Increases in CFL were also observed, however, particularly near the study area's western and southwestern boundaries where treatments were least concentrated (Figs. 4, 5). Maximum pixel-level CFL increases were 2.5 m (LO FM) and 3.1 m (HI FM).

Because fire hazard was used in prioritizing stands for treatment, the estimated pre-treatment CFL in stands selected for treatment (4.3–5.1 m)

Table 3. Proportional change in burn probability for treatment scenarios compared to the NT scenario.

Stand type	LO FM S1	LO FM S2	LO FM S3	HI FM S1	HI FM S2	HI FM S3	AVG S1	AVG S2	AVG S3
	Proportional change relative to NT			Proportional change relative to NT			Proportional change relative to NT		
All PACs	−0.45	−0.64	−0.56	−0.53	−0.63	−0.59	−0.49	−0.64	−0.57
Treated PACs	NA	−0.81	−0.76	NA	−0.57	−0.67	NA	−0.69	−0.72
Untreated PACs	−0.45	−0.53	−0.51	−0.53	−0.54	−0.58	−0.49	−0.53	−0.54
All stands	−0.44	−0.48	−0.53	−0.50	−0.53	−0.56	−0.47	−0.50	−0.54
Treated stands	−0.76	−0.79	−0.63	−0.72	−0.74	−0.74	−0.74	−0.76	−0.69
Untreated stands	−0.35	−0.39	−0.49	−0.45	−0.47	−0.51	−0.40	−0.43	−0.50

Notes: NT, no treatment; PACs, protected activity centers. Proportions are ratios of treatment values to no-treatment values as calculated for the *same stands*. Treatment and no-treatment values were calculated as the average pixel value for the low and high fuel model range (LO FM and HI FM) within each stand category and treatment scenario.

was greater than in stands not selected (3.2–3.3 m). After treatment, average CFL within treated stands fell to 1.3 (S1 and S2) and 1.7 m (S3). CFL in untreated stands was also reduced as a result of the influence of treatments on fire spread and intensity. CFL fell by 0.5–0.8 m (9–16%) relative to CFL in the same stands within the no-treatment landscape (Table 4).

Although spotted owl PACs were not treated in S1, relative to PACs in the NT landscape, PAC CFL was reduced by 10% (to 3.2 m) in S1. Treating PACs had a much larger impact on potential fire intensity, however. Average treated PAC CFL fell to 1.3 and 1.4 m in S2 and S3, respectively.

Carbon consequences of landscape fuel treatments

Prior to treatment, aboveground landscape carbon totaled 147.05 tonnes/ha, on average. Treatments removed 14% of pre-treatment C from treated stands, or 23.74 tonnes/ha, totaling 81,772–119,103 tonnes of C in harvested biomass and merchantable material (Tables 5 and 6).

Both the treatment scenarios and the choice of fuel models were important influences on estimated C emissions from burning. As the least restrictive treatment scenario in terms of treatment location and the only scenario to include treatment of private land, where broadcast burning was precluded as a treatment option, the S3 treatment scenario was associated with the lowest wildfire and prescribed burning emissions (Tables 5 and 6). For each treatment scenario, expected wildfire emissions increased by more than an order of magnitude between the low and high fuel model ranges. This difference was the result of increasing fire intensity as well as wildfire size. Average wildfire size nearly doubled between fuel model ranges in the treatment scenarios and tripled in the no-treatment scenario (Fig. 6). For a given treatment scenario, wildfire emissions on a per hectare basis were approximately two tonnes greater for the high fuel

Table 4. Proportional change in conditional flame length for treatment scenarios compared to the NT scenario.

Stand type	LO FM S1	LO FM S2	LO FM S3	HI FM S1	HI FM S2	HI FM S3	AVG S1	AVG S2	AVG S3
	Proportional change relative to NT			Proportional change relative to NT			Proportional change relative to NT		
All PACs	−0.09	−0.42	−0.28	−0.12	−0.43	−0.27	−0.10	−0.42	−0.28
Treated PACs	NA	−0.71	−0.75	NA	−0.71	−0.73	NA	−0.71	−0.74
Untreated PACs	−0.09	−0.14	−0.11	−0.12	−0.17	−0.14	−0.10	−0.16	−0.13
All stands	−0.22	−0.26	−0.31	−0.25	−0.26	−0.28	−0.24	−0.26	−0.30
Treated stands	−0.65	−0.69	−0.52	−0.71	−0.71	−0.73	−0.68	−0.70	−0.62
Untreated stands	−0.08	−0.09	−0.21	−0.11	−0.12	−0.12	−0.09	−0.10	−0.16

Notes: NT, no treatment; PACs, protected activity centers. Proportions are ratios of treatment values to no-treatment values as calculated for the *same stands*. Treatment and no-treatment values were calculated as the average pixel value for the low and high fuel model range (LO FM and HI FM) within each stand category and treatment scenario.

Table 5. Expected biomass carbon, expected wildfire C emissions, and C harvested and emitted in fuel treatments for NT and treatment (S1, S2, S3) scenarios using the low fuel model range in fire modeling.

Carbon pool	NT	S1	S2	S3
Untreated stands		Tonnes C		
Live	5,004,505	3,915,044	3,695,489	3,954,131
Dead	1,845,659	1,421,704	1,378,824	1,466,202
Wildfire emissions	2372	1137	1039	904
Treated stands				
Live	...	881,358	1,072,625	826,366
Dead	...	260,089	299,244	220,932
All harvested biomass	...	88,773	81,772	119,103
Prescribed fire emissions	...	178,530	169,693	122,599
Wildfire emissions	...	48	56	61
All stands				
Live	5,004,505	4,796,402	4,768,113	4,780,497
Dead	1,845,659	1,681,793	1,678,068	1,687,135
Treatment (harvested and emitted)	...	267,303	251,465	241,702
Wildfire emissions	2372	1186	1095	965
Grand totals	6,852,536	6,746,684	6,698,741	6,710,299

Notes: LF indicates the large fire scenarios. Expected C is that remaining in the core study area following treatment, if applicable, and a random ignition and wildfire in the larger buffered study area, as estimated from the simulation of many wildfires. Live C is that contained in live aboveground herb, shrub, and tree biomass; dead C is the C contained in litter, duff, woody surface fuel, and aboveground portions of tree snags. NT, no treatment.

model range than for the low range. In contrast to the large influence of fuel model choice on wildfire emissions, the effect of fuel model range on prescribed fire emissions was minimal, with only a 1% increase in emissions between the low and high fuel model ranges for a given treatment scenario.

Although treatment significantly reduced wildfire emissions, combined emissions from prescribed burning on 20% of the landscape and wildfire exceeded wildfire emissions in the no-treatment scenarios (Tables 5 and 6). Relative to the no-treatment scenarios, treatment reduced estimated wildfire emissions by approximately 54% (low fuel model range), 59% (high fuel model range), and 45% (large fire scenarios). Yet prescribed burning was a far more significant source of emissions than were wildfires of moderate size, with emissions from treatment exceeding wildfire emissions by 111,259–177,344 tonnes. Even for the large wildfire simulations, where landscape treatments nearly halved estimated wildfire emissions, the combined carbon emissions from prescribed burning and wildfire in the treatment scenario surpassed wildfire emissions in the no-treatment scenario by 45% (Table 6, Fig. 7).

The total quantity of aboveground C expected to remain on the landscape following treatment and a randomly ignited wildfire was greatest for the no-treatment scenarios (Tables 5 and 6). For modeled wildfires of moderate size, treatment reduced both the live and dead C pools relative to the no-treatment scenarios, and terrestrial C in the no-treatment scenarios was 4–5% greater (323,316 –434,960 tonnes) than in any of the treatment scenarios (Tables 5 and 6). In comparison, under large wildfire conditions, the treatment scenario retained slightly more live biomass C:~15,000 tonnes, or 0.3% more than the no-treatment scenario. However, treatment also reduced necromass C by 288,000 tonnes (12%), resulting in a 3% overall decrease in onsite biomass C relative to an untreated landscape (Table 6).

The proportional changes in aboveground biomass C pools between the treatment and no-treatment scenarios are summarized in Table 7. For all treatment scenarios, the consumption of duff, litter, and downed woody fuels with prescribed burning contributed to a net reduction in these C pools relative to the untreated landscape. Conversely, treatments protected more C in the live understory (herb and shrub) pool—the result of reduced wildfire size and intensity in the treatment scenarios. Treatments in the moderate wildfire scenarios reduced live tree biomass C in comparison with no-treatment levels (Fig. 8).

Table 6. Expected biomass carbon, expected wildfire C emissions, and C harvested and emitted in fuel treatments for NT and treatment (S1, S2, S3) scenarios using the high fuel model range in fire modeling.

Carbon pool	NT	S1	S2	S3	NT-LF	S3-LF
Untreated stands			Tonnes C			
Live	4,910,239	3,879,854	3,664,690	3,923,358	4,586,898	3,750,774
Dead	1,912,429	1,448,947	1,403,313	1,489,629	2,130,034	1,605,591
Wildfire emissions	31,831	12,934	12,001	11,293	137,622	68,061
Treated stands						
Live	...	896,670	1,115,918	853,503	...	847,374
Dead	...	280,182	325,628	238,130	...	247,744
All harvested biomass	...	88,773	81,772	119,103	...	119,103
Prescribed fire emissions	...	180,357	171,247	123,539	...	123,539
Wildfire emissions	...	798	951	986	...	8285
All stands						
Live	4,910,239	4,776,524	4,780,607	4,776,861	4,586,898	4,598,148
Dead	1,912,429	1,729,129	1,728,941	1,727,759	2,130,034	1,853,334
Treatment (harvested and emitted)	...	282,862	265,971	254,921	...	254,921
Wildfire emissions	31,831	13,732	12,952	12,279	137,622	76,346
Grand totals	6,854,499	6,802,247	6,788,471	6,771,820	6,854,554	6,782,749

Notes: LF indicates the large fire scenarios. Expected terrestrial C is that remaining in the core study area following treatment, if applicable, and a random ignition and wildfire in the larger buffered study area, as estimated from the simulation of many wildfires. Live C is that contained in live aboveground herb, shrub, and tree biomass; dead C is the C contained in litter, duff, woody surface fuel, and aboveground portions of tree snags. NT, no treatment.

Notably, in the large modeled wildfire scenarios (NT-LF and S3-LF), treatments resulted in a 400,000-tonne increase in landscape-level live tree C over the no-treatment scenario.

Discussion

Fuel treatments in protected habitat

Because there are often competing objectives between managing forests for resilience to fire and drought and protecting owl habitat, we assessed potential fire occurrence and hazard based on treatment scenarios that included and omitted treatment of PACs. Conducting fuels management outside of occupied owl habitat has been suggested as a means of reducing fire risk within occupied sites (Jenness et al. 2004, Tempel et al. 2015). Ager et al. (2007) reported that fuel treatments on 20% of a western Oregon landscape reduced the probability of northern spotted owl nesting and roosting habitat loss by 44%, even though that habitat type was not treated. As in Ager et al. (2007), we observed modifications in fire intensity and burn probability within owl habitat even when it was left untreated. In the S1 treatment scenario, in which owl activity centers were not eligible for treatment, the effect of treating other stands reduced both fire hazard (by 9–12% for CFL, or approximately 0.4 m) and CBP (by ~45%) within PACs. It is difficult to assess the significance of this proportional reduction in CBP given that the absolute differences in probabilities were relatively modest (Figs. 3, 4). Ager et al. (2007) noted that allowing treatment of owl habitat would have significantly reduced estimated habitat loss in their study. In this study, it is expected that habitat quality may be reduced within treated PACs in the short term through the removal of small-diameter trees (i.e., lower vertical structural heterogeneity). However, two structural attributes for suitable spotted owl nesting habitat identified by Tempel et al. (2015), high canopy cover and large tree density, would not be altered. While we did not directly estimate habitat loss, we did observe much larger reductions in fire hazard within PACs that were treated as measured by CFL (71–75% reduction, equivalent to 2–3 m). It should be noted that the effect of widespread treatments within PACs on spotted owl nesting and foraging behavior is unknown.

One concern with designating some land area unavailable for treatment is that it may limit the potential for treatments to alter fire behavior across the landscape (e.g., Finney 2001). Including all stands in the potential treatment pool allows the highest-priority stands, with respect to simulated

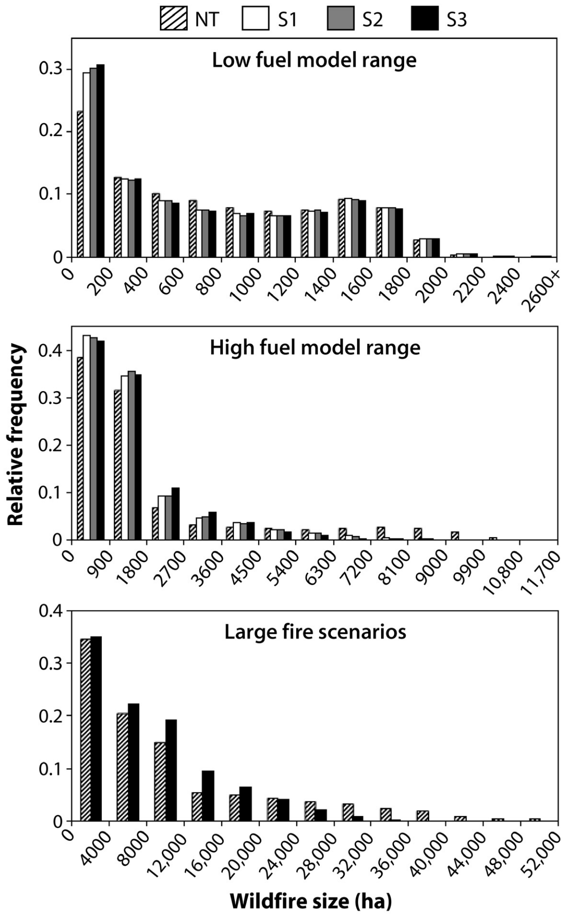

Fig. 6. Wildfire size relative frequency distributions from wildfire simulations. Bar color represents no-treatment (NT) and treatment scenarios (S1–S3).

fire spread, to be treated, which would be expected to achieve the greatest modification of landscape fire behavior and effects. In the present study, although the land area potentially available for treatment more than doubled between S1 and S3, landscape-level effects of treatment on modeled fire risk and hazard were fairly similar (compared with the no-treatment landscape, all-stand CBP fell by 47% and 54% in S1 and S3, while CFL fell by 24% and 30%). Dow et al. (2016) also found that incorporating modest restrictions on treatment area availability (24% of the landscape unavailable) had minimal consequences for modeled fire size and hazard. The modest changes in estimated fire metrics we observed may also be due to similarity in the general pattern of treatment placement between scenarios, which probably led to similar effects on landscape-level fire behavior. The true effect of increasing the land area available for treatment may be partially obscured by the varying frequency of treatment prescriptions between scenarios. For example, the hand thinning treatments applied within PACs would be expected to have a milder effect on potential wildfire behavior than more severe prescriptions, and hand thinning was twice as common in S2 as in the other scenarios.

Terrestrial carbon and burning emissions

Landscape treatments reduced wildfire emissions by reducing the emissions produced per area burned by wildfire as well as average wildfire size. On average, wildfires in the treated landscapes released 19.3–21.6 tonnes C/ha, while wildfires in the untreated landscapes released 23.4–25.4 tonnes C/ha. Modeled wildfires decreased in size by 7% (low fuel model range), 36% (high fuel model range), and 25% (large fire scenario) relative to untreated landscapes (Fig. 6). Since the burn period for simulated wildfires was held constant between scenarios, this reduction in average wildfire size is the result of reduced spread rates derived from fuel treatments.

Despite the influence of treatments on wildfire intensity, size, and expected emissions, treatment-related emissions exceeded the avoided wildfire emissions conferred by treatment. Prescribed burning in our study, a combination of broadcast

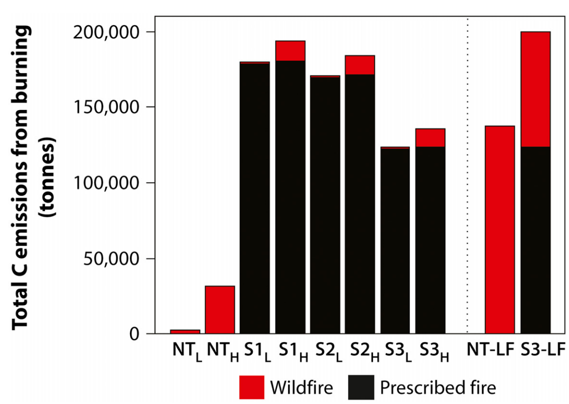

Fig. 7. Carbon emissions (tonnes) from wildfire and prescribed burning. *X*-axis labels indicate no-treatment (NT) and treatment scenarios (S1–S3); subscripts denote fuel model ranges used in fire modeling (L: low, H: high). Large fire scenarios, which were modeled with the high fuel model range only, are indicated by LF.

Table 7. Proportional change in expected carbon by biomass pool for treatment scenarios compared to the no-treatment landscape.

Treatment scenario	Standing dead	Down dead wood	Forest floor	Herb/shrub	Live tree
S1	0.04	−0.17	−0.13	0.14	−0.04
S2	0.01	−0.16	−0.12	0.14	−0.04
S3	−0.01	−0.15	−0.10	0.16	−0.04
S3-LF	−0.16	−0.13	−0.08	0.17	0.00

Notes: For example, a value of −0.10 represents a 10% decline in biomass C from the no-treatment scenario. Treatment and no-treatment values were calculated as the average of low and high fuel model range values, except in the case of the large fire (LF) scenarios, which were modeled for the high fuel model range only. Expected C is that remaining after a random ignition and wildfire in the buffered study area as estimated from simulating 80,000 ignitions (LF: 40,000 ignitions).

C pool categories are those reported in Forest Vegetation Simulator Carbon Reports. *Standing dead*: aboveground portion of standing dead trees, *Down dead wood*: woody surface fuels, *Forest floor*: litter and duff, *Herb/shrub*: herbs and shrubs, *Live tree*: aboveground portion of live trees.

and pile burning, released 11.1–16.3 tonnes C/ha. For comparison, studies conducted in comparable forest types have estimated prescribed fire emissions of 12.7 tonnes C/ha (warm, dry ponderosa pine habitat types; Reinhardt and Holsinger 2010)

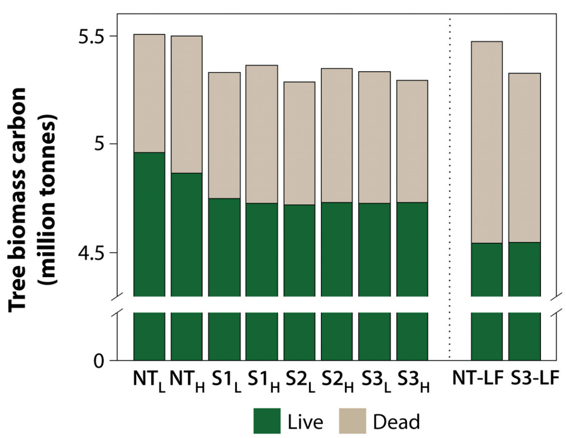

Fig. 8. Expected carbon contained in aboveground live and dead tree biomass. Expected C is that remaining in the core study area following treatment (if applicable) and a single random ignition within the larger buffered study area. *X*-axis labels indicate no-treatment (NT) and treatment scenarios (S1–S3); subscripts denote fuel model ranges used in fire modeling (L: low, H: high). Large fire scenarios, which were modeled for the high fuel model range only, are indicated by LF.

and 14.8 tonnes C/ha (an old-growth mixed-conifer reserve in the southern Sierra Nevada; North et al. 2009*a*). Relative to the approximately 158,000 tonnes C emitted in prescribed burning, avoided wildfire emissions, at 1186–19,551 tonnes for wildfires of moderate size, were small. A similar study in southern Oregon with average modeled wildfires of 2350 and 3500 ha (treatment and no-treatment scenarios, respectively) found that treatments reduced expected wildfire emissions by 6157 tonnes of C (Ager et al. 2010).

Surface fuels, represented with surface fuel models in commonly used modeling software, are the most influential inputs determining predicted fire behavior (Hall and Burke 2006). Fire behavior, fire sizes, and emissions in this study varied according to fuel model assignment, highlighting the importance of selecting the appropriate fuel model to represent fuel conditions (see Collins et al. 2013). We show a 12- to 14-fold change in wildfire emissions due solely to the choice of fuel models (Tables 5 and 6). Indeed, the range of fuel models used in recent studies investigating fuel treatments and simulated fire behavior in mixed-conifer forests is noteworthy. Incorporating a range of fuel models into analyses such that outcome variability can be reported facilitates comparison of effects across studies.

Our estimates of the aboveground C benefits of treatment under the moderate wildfire scenarios, with average fire sizes of ≤2133 ha, are likely conservative. The effect of modeled wildfire size on the C consequences of fuel treatment was considerable, emphasizing the importance of this variable in studies of the climate benefits of treatment. Avoided wildfire emissions resulting from treatment increased to 61,276 tonnes C when large wildfires (8070–10,757 ha) were simulated. The treatment scenario, given large wildfires, also protected a greater portion of live tree C. If the ~40,000-ha King Fire is representative of the magnitude of future wildfires in the region, C accounting should improve with respect to treatment favorability. Similarly, if multiple wildfires were to encounter the study area within the effective lifespan of treatments, the C gains associated with avoided emissions in the treatment scenarios would increase.

Our approach to estimating the C consequences of fuel treatments has a number of limitations. A full accounting of treatment effects would project through time the consequences of both treatment and wildfire. Our analysis is static, incorporating only the short-term C costs and benefits of treatment. Simulating wildfire in the year immediately following treatment maximizes the apparent benefits of treatment. Over time, as surface fuels accumulate and vegetation regenerates, maintenance would be required to retain the effectiveness of treatments (Martinson and Omi 2013), increasing the C costs of reduced fire hazard. In addition, the C contained in fire-killed biomass will ultimately be emitted to the atmosphere, although biomass decay could be delayed through conversion to long-lived wood products such as building materials (Malmsheimer et al. 2011). It is also important to note that our analysis did not include stochastic wildfire occurrence. Estimates of burn probability in the present study are not estimates of the likelihood of wildfire occurrence based on historical fire sizes and frequency (e.g., Preisler et al. 2004, Mercer and Prestemon 2005, Brillinger et al. 2006), but rather are conditional on a single randomly ignited wildfire within the buffered study area.

Conclusions

Our findings generally support those of Campbell et al. (2011), who concluded from an analysis of fire-prone western forests that the C costs of treatments are likely to outweigh their benefits under current depressed fire frequencies. In a more recent paper, Campbell and Ager (2013) concluded that "none of the fuel treatment simulation scenarios resulted in increased system carbon," primarily from the low incidences of treated areas being burned by wildfire. However, our interpretation of these findings differs from those discussed in Campbell et al. (2011) and Campbell and Ager (2013), especially in light of recent and projected future trends in fire activity (Westerling et al. 2011, Miller and Safford 2012, Dennison et al. 2014). The current divergence of increasing surface air temperatures and low fire activity is unlikely to be sustained, further suggesting greater future fire activity (Marlon et al. 2012). If increased fire activity is realized, then the likelihood of a given area being burned in a wildfire increases. This differs from the simple increase in stand-level fire frequency modeled by Campbell et al. (2011) because increases in fire likelihood are not necessarily associated with corresponding decreases in fire severity, as assumed by Campbell et al. (2011). Increased fire likelihood could very well lead to positive feedbacks in fire severity, and ultimately to vegetation type conversion (Coppoletta et al. 2016)—effects that would have significant implications for carbon storage.

Due to the significant emissions associated with treatment and the low likelihood that wildfire will encounter a given treatment area, forest management that is narrowly focused on C accounting alone would favor the no-treatment scenarios. Landscape treatments protected more C in live tree biomass only when large wildfires were simulated. While treatment favorability improved with large wildfire simulation, the no-treatment scenario still produced fewer emissions than the treatment scenario. Given the potential for large wildfire in the region as demonstrated by the 2014 King Fire, and the increasing frequency of large wildfires and area burned in California expected from climate modeling studies (Lenihan et al. 2008, Westerling et al. 2011), we suggest that future studies of fuel treatment–wildfire–C relationships should incorporate the potential for large wildfires at a frequency greater than those observed over the last 20–30 yr. Others have argued that treatments to increase forest resilience should be a stand-alone, top land management priority independent of other ecosystem values such as carbon sequestration and fire hazard reduction (Stephens et al. 2016*a*).

We also note that the potential benefits of fuels management are not restricted to avoided wildfire emissions. Here, we show that landscape fuel treatments can alter fire hazard across the landscape both within and outside of treated stands, and have the potential to affect the likelihood of burning and fire intensity within protected California spotted owl habitat. Underscoring the risk to sensitive habitat, the 2014 King Fire encountered 31 PACs within our study area, leading to the greatest single-year decline in habitat occupancy recorded over a 23-yr study period (Jones et al. 2016). Modest simulated treatments within activity centers significantly reduced potential fire intensity relative to both the no-treatment landscape and a treatment scenario that did not permit treatment within PACs, indicating that active management may be desirable to protect habitat in the long term (Roloff et al. 2012).

However, treatments conducted outside of owl habitat also reduced wildfire hazard.

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Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.1648/full