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Restoring surface fire stabilizes forest carbon under extreme fire weather in the Sierra Nevada

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Abstract. Climate change in the western United States has increased the frequency of extreme fire weather events and is projected to increase the area burned by wildfire in the coming decades. This changing fire regime, coupled with increased high-severity fire risk from a legacy of fire exclusion, could destabilize forest carbon (C), decrease net ecosystem exchange (NEE), and consequently reduce the ability of forests to regulate climate through C sequestration. While management options for minimizing the risk of high-severity fire exist, little is known about the longer-term carbon consequences of these actions in the context of continued extreme fire weather events. Our goal was to compare the impacts of extreme wildfire events on carbon stocks and fluxes in a watershed in the Sierra National Forest. We ran simulations to model wildfire under contemporary and extreme fire weather conditions, and test how three management scenarios (no-management, thin-only, thin and maintenance burning) influence fire severity, forest C stocks and fluxes, and wildfire C emissions. We found that the effects of treatment on wildfire under contemporary fire weather were minimal, and management conferred neither significant reduction in fire severity nor increases in C stocks. However, under extreme fire weather, the thin and maintenance burning scenario decreased mean fire severity by 25%, showed significantly greater C stability, and unlike the no-management and thin-only management options, the thin and maintenance burning scenario showed no decrease in NEE relative to the contemporary fire weather scenarios. Further, under extreme fire weather conditions, wildfire C emissions were lowest in the thin and maintenance burning scenario, (reduction of 13.7 Mg C/ha over the simulation period) even when taking into account the C costs associated with prescribed burning. Including prescribed burning in thinning operations may be critical to maintaining C stocks and reducing C emissions in the future where extreme fire weather events are more frequent.

Key words: *Abies magnifica;* carbon sequestration; Collaborative Forest Landscape Restoration Project; Dinkey Creek; fire emission; forest management; LANDIS-II; mixed-conifer; *Pinus ponderosa;* prescribed fire; wildfire.

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Introduction

Increased temperature and greater inter-annual precipitation variability resulting from ongoing climate change are projected to increase the area burned by wildfire across much of the western United States (Westerling et al. 2011*a*, *b*, Moritz et al. 2012). Fire season length and the area burned by wildfire have already increased as a result of warmer temperatures and earlier spring snowmelt (Westerling 2016). These climatic trends are also increasing the frequency of extreme fire weather events (Collins 2014). Given the contribution of forest carbon (C) uptake to regulating climate and the role of wildfire in emitting C stored in forests back to the atmosphere, understanding how changing fire weather conditions will alter forest C dynamics is central to informing forest management and climate policy decision making (Bonan 2008, van der Werf et al. 2010, Wiedinmyer et al. 2011, Millar and Stephenson 2015).

Decadal increases in area burned correlate with warming temperatures and earlier spring snowmelt across the western United States, with the rate of change varying regionally (Westerling 2016). In some forest types and regions, such as the mixed-conifer forests of the Sierra Nevada, the effects of changing climate and fire weather are compounded by a century of fire exclusion that has altered forest structure and increased surface fuels, such that the likelihood of large, severe wildfire has increased (Agee and Skinner 2005, Hessburg et al. 2005, Stephens et al. 2007, Miller et al. 2009). Thus, changing climate and associated increases in the frequency of extreme fire weather exacerbate the flammability of forests where historically frequent fires maintained forest structures that were more resistant to high-severity wildfire (Marlon et al. 2012, Collins 2014, Hurteau et al. 2014).

The role of modifying forest structure and fuel loads in historically frequent-fire forests by lowering tree density and reintroducing surface fire has been demonstrated as an effective means of reducing the risk of high-severity wildfire (Stephens et al. 2012). While treatments effectively reduce the severity and rate of spread of wildfire, their efficacy is contingent on the timing of wildfires following treatment and the spatial distribution of treatments across a landscape (Finney et al. 2007, McGinnis et al. 2010). In the context of climate regulation and forest C dynamics, efforts to restore forest structure and fire regimes have been a point of vigorous debate because of the C stock reductions incurred with treatment, the low probability of wildfire occurring, and the effective lifespan of treatments in modifying fire behavior (Hurteau et al. 2008, Campbell et al. 2012, Campbell and Ager 2013, Hurteau 2013). However, the role of increasingly extreme fire weather has the potential to alter wildfire size and severity, and treatment effectiveness, adding to the uncertainty already associated with forest C dynamics under future climate scenarios (Collins 2014).

Reducing tree density to restore forest structure typically involves a 30–40% reduction in live tree C (Finkral and Evans 2008, North et al. 2009, Stephens et al. 2009). However, the effectiveness of thinning treatments is improved with the reintroduction of surface fire (Ager et al. 2013, Collins et al. 2013, Loudermilk et al. 2013, 2014), and under some conditions, prescribed burning is the only option that is operationally available (North et al. 2012). While prescribed burning emits C to the atmosphere, per unit area emissions can be both substantially lower than wildfire and re-sequestered in a relatively short time by subsequent regrowth of vegetation (Wiedinmyer and Hurteau 2010, Wiechmann et al. 2015). Furthermore, moderating fire severity alters subsequent C source–sink dynamics as tree mortality decreases with decreasing fire severity (Meigs et al. 2009, North and Hurteau 2011, Dore et al. 2012, Earles et al. 2014).

Given the established relationships among changing climate, increasing area burned, and increasing frequency of extreme fire weather, we sought to quantify treatment efficacy and its effects on net landscape C dynamics under more extreme fire weather in a forested watershed in the southern Sierra Nevada Mountains, California. We hypothesized that (1) under contemporary fire weather, thinning alone and thinning combined with maintenance burning would decrease fire severity relative to no-management, but that under extreme fire weather, thinning combined with maintenance burning would be required to reduce fire severity; (2) the C stock reductions from thinning and maintenance burning would result in decreased landscape C storage under contemporary fire weather and increased landscape C storage under extreme fire weather conditions; (3) the C sink strength of the forest would be greatest in the fully treated landscape, in spite of an overall reduction in biomass, due to increased stability of live tree biomass and reduced resource competition between trees; and (4) under extreme fire weather, C emissions due to wildfire would be lowest under the thinning and maintenance burning treatment relative to the control because of reduced fire severity.

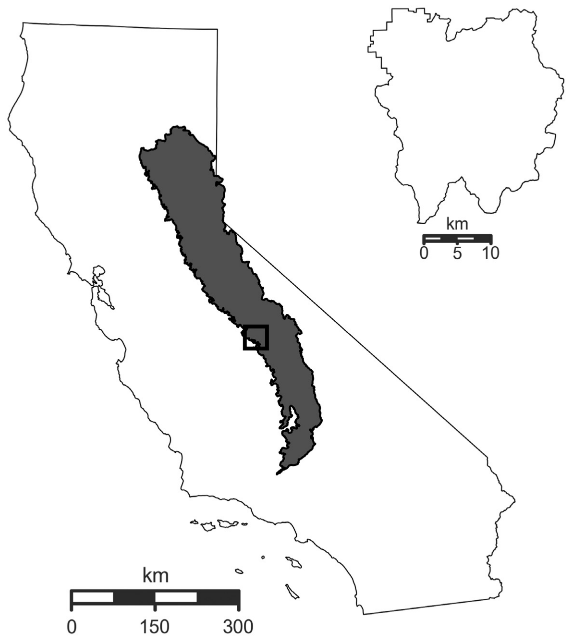

Fig. 1. Area map of the simulation extent. The Dinkey Creek watershed is located in the black extent indicator, in the Sierra Nevada (gray) of central California.

Methods

Study area

The Dinkey Creek watershed covers approximately 87,500 ha in the southern Sierra Nevada, California (Fig. 1).

Climate across the watershed is characterized as ranging from hot Mediterranean to upper boreal, with elevation ranging from approximately 300 to 3000 m. Following this elevation gradient, precipitation ranges from 50 to 100 cm annually, with a larger percentage falling as snow at higher elevations. Along this gradient, mean daily minimum temperatures range from −3° to 10°C and mean daily maximum temperatures range from 12° to 25°C (DAYMET, Thornton et al. 2012).

Soils are relatively shallow, with depth decreasing across the elevation gradient, and correspondingly, the soil orders range from Alfisols to Inceptisols and Entisols as soils become less developed. The substrate is predominately granitic with outcrops of sandstone and basalt at higher elevations (SSURGO, NRCS 2013). Vegetation across the lower elevation zone is dominated by a mixture of shrubs (*Arctostaphylos* sp., *Ceanothus* sp.), oaks (*Quercus chrysolepis, Quercus douglasii, Quercus kelloggii, Quercus wislizeni*), *Pinus sabiniana*, *Pinus ponderosa,* and *Calocedrus decurrens* as elevation increases toward the ecotone with mid-montane forests. The lower-montane mixed-conifer forest is predominantly comprised of *Abies concolor*, *C. decurrens*, *Pinus lambertiana*, and *Pinus jeffreyi*. As elevation increases, upper-montane forests are dominated by *Abies magnifica*, with patches of *Pinus contorta* and *P. jeffreyi*. The highest elevation forests within the Dinkey Creek watershed are comprised of *Pinus monticola, A. magnifica, Tsuga mertensiana, P. jeffreyi,* and *P. contorta*. Historically, fire frequency decreased with increasing elevation on the western slope of the Sierra Nevada and mean fire return intervals ranged from 4 yr at the lowest elevations to 15 yr at mid-elevation and up to 175 yr at the highest elevations (Caprio and Swetnam 1993, Scholl and Taylor 2006).

A legacy of fire suppression throughout the Sierras has substantially altered the forest types with the shortest fire return intervals (Scholl and Taylor 2006, Stephens et al. 2007, Beaty and Taylor 2008). Consequently, transitions in stand structure have increased stem density and the proportion of fire-intolerant species, increasing the risk of stand-replacing fire throughout the watershed.

Model description and parameterization

We used the landscape disturbance and succession model LANDIS-II (v6.0) to simulate the effects of forest management and wildfire on landscape-scale forest C dynamics. LANDIS-II simulates tree and shrub species-specific age-cohorts of biomass across a gridded, spatially explicit landscape (Scheller et al. 2007). Species grow and compete within grid cells and disperse across grid cells following disturbance. The model describes the landscape in terms of plant species, functional groups, and ecoregions, and uses extensions to incorporate additional processes into the modeling framework. We used the Century Succession (v4.0.1) extension to track landscape C dynamics, the Dynamic Fire and Fuels System (v2.0.5) and the Dynamic Fuels Leaf Biomass (v2.0) extensions to simulate stochastic fire and changes to forest fuels across the landscape, and the Leaf Biomass Harvest extension (v2.0.3) to simulate management scenarios. LANDIS-II has been used extensively to model forest C dynamics in the context of management, future climate, and disturbances such as wildfire and bark beetles (Scheller and Mladenoff 2008, Sturtevant et al. 2009, Xu et al. 2009, Scheller et al. 2011*a*, *b*, Syphard et al. 2011, Loudermilk et al. 2013, Martin et al. 2015, Kretchun et al. 2016, Laflower et al. 2016).

The core LANDIS-II model requires a spatially explicit initial community, composed of age-cohorts of biomass by species. In this study, we leveraged the 150-m gridded initial communities layer developed by Liang et al. (*in press*), which we spatially resampled to 1-ha grid cells. The initial communities layer was developed by stratifying both US Forest Service Forest Inventory and Analysis (FIA) data and the landscape based on a suite of biophysical attributes and species age-cohorts were parameterized by sampling FIA data to populate each stratum (see Liang et al., *in press*). We further modified this initial communities layer to include two shrub functional types, a nitrogen fixer and a resprouter, with species parameterization based on *Ceanothus* sp. and *Arctostaphylos* sp., which are prevalent throughout the watershed. Functionally, the inclusion of shrubs in the model allows gaps in the canopy to become populated with understory vegetation, which can inhibit tree seedling establishment and growth via light competition. The shrub component also facilitates the spread of fire by increasing the fuel continuity of the landscape.

Ecoregion parameters were developed and calibrated following previously established procedures (Scheller et al. 2011*a*, *b*, Loudermilk et al. 2013) using publicly available data sets. We used values from the literature and gSSURGO data (NRCS 2013) to establish soil parameters across the watershed, and leveraged the broad edaphic gradient in geologic parent material coupled with elevation to define eight distinct ecoregions for the study area. Fig. 2 illustrates the combination of elevation (Fig. 2A) and geologic parent group (Fig. 2B) that were combined to develop the ecoregions across the watershed. Wilting point and field capacity were calculated from soil texture by ecoregion (Saxton et al. 1986). We used aboveground biomass as the primary calibration target for the model (Fig. 2C).

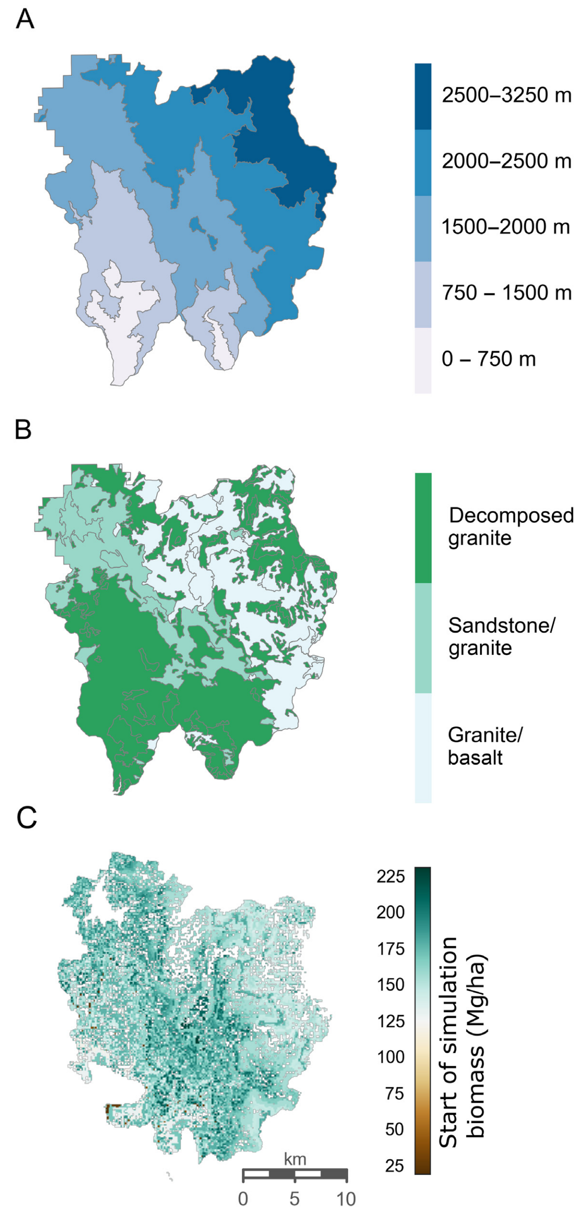

Fig. 2. We divided the Dinkey Creek watershed into five elevation bands (A) and three soil parent groups (B), ultimately to define eight ecoregions, resulting in a starting biomass distribution (C) that we validated against Forest Inventory and Analysis plots.

Ecosystem C dynamics

We used the Century Succession extension (Scheller et al. 2011*a*), which includes below-ground dynamics derived from the original CENTURY soil model (Parton et al. 1983), to model C dynamics across the landscape. The model accounts for aboveground C (AGC) accumulation through the growth and development of cohorts, coupled with C and nitrogen (N) cycling within the soil via decomposition. Cohort growth and decomposition at monthly time-steps is influenced by precipitation and temperature. We obtained Century-specific parameters for LANDIS ecoregions, tree functional groups, and tree species from the literature. Vegetation-specific parameter sets were developed elsewhere (Loudermilk et al. 2013, 2014). The Century extension uses a spin-up period equivalent to the oldest tree cohort to allow for stabilization of soil C prior to initiating simulations. We validated the model by comparing aboveground tree biomass following spin-up with tree biomass estimates we calculated from FIA data using allometric equations from Jenkins et al. (2003) and Chojnacky et al. (2014). Simulated biomass ranged from 19.8 to 231.8 Mg/ha with a mean value of 163.9 Mg/ha (Fig. 2C). Biomass estimates from FIA data ranged from 7.1 to 433.9 Mg/ha with a mean value of 187.0 Mg/ha. While the model did not capture the full range of variability in biomass across the landscape, the results indicate that the model is accounting for the biotic and abiotic constraints on growth.

Climate, wildfire, and fuels

LANDIS-II uses climate on an ecoregion-specific basis to govern the growth and reproduction of vegetation across the landscape. We used Daymet daily surface weather over a 1-km grid for the period 1980–2015, acquired via the USGS Geo Data Portal (http://cida.usgs.gov/gdp/, Thornton et al. 2012). We computed weighted area grid statistics on a per-ecoregion basis using the export service in the data portal. The LANDIS-II Century extension then converted these data to monthly means. At each time-step, the model randomly draws 1 yr of climate data from the distribution of years to provide monthly climate data for simulating vegetation growth and reproduction in a given simulation year.

The Dynamic Fire and Fuels extension simulates fire and fuel interactions as a function of a user-defined probability of ignition, fire size and fire weather distributions, coupled with topography and fuel availability (Sturtevant et al. 2009). The extension uses a distribution of weather attributes separate from those required by the Century extension to provide fire weather for a specific fire event. Wildfire is simulated stochastically, and when an ignition that results in a fire occurs, the extension draws the maximum fire size from the size distribution. The realized fire size is then constrained as a function of fuel availability and fire weather. We used CALFIRE data (Fire Perimeters v. 15.1, released 2016, http://frap.cdf.ca.gov/data/frapgisdata-sw-fireperimeters_download) to look at the historic distribution of fires across a portion of the southern Sierra (Fig. 3A), and used the fires over the period 1983 to 2014 to both determine the number of fires per year and derive the mean and standard deviation of fire size by fitting a lognormal distribution to fire perimeters for the same extent (Fig. 3B). These fire data included the 104,131-ha Rim Fire, which was larger than our study area. As a result, we set the maximum fire size equal to the size of our study area. We divided the landscape into three distinct fire regions to capture the variability in the weather distribution that occurs with elevation (Fig. 4A). Given the relatively low number of fires that occurred within the watershed, we held the number of ignitions and fire size distributions constant across all three fire regions.

To test our hypotheses about the influence of extreme fire weather, we developed two different fire weather distributions (contemporary and extreme) using meteorological data from remote access weather stations (RAWS). In both scenarios however, the climate data responsible for vegetation growth and reproduction were the same. For the contemporary fire weather distribution, we obtained data from three RAWS that were located across an elevation range that coincided with our fire regions. The RAWS were all within 25 km of the Dinkey Creek watershed and had meteorological records of 10–13 yr. For the extreme fire weather distribution, we obtained data from the Smith Peak RAWS from the year 2013. This station captured the weather conditions leading up to and during the Rim Fire. We used each weather distribution to compute the required inputs for the Dynamic Fire and Fuels extension, including fine fuel moisture code, build-up index, wind speed, wind direction, and fire weather index bin (Sturtevant et al. 2009, Fig. 4B–D). Each of these parameters, coupled with fuel type, determines the severity of a fire

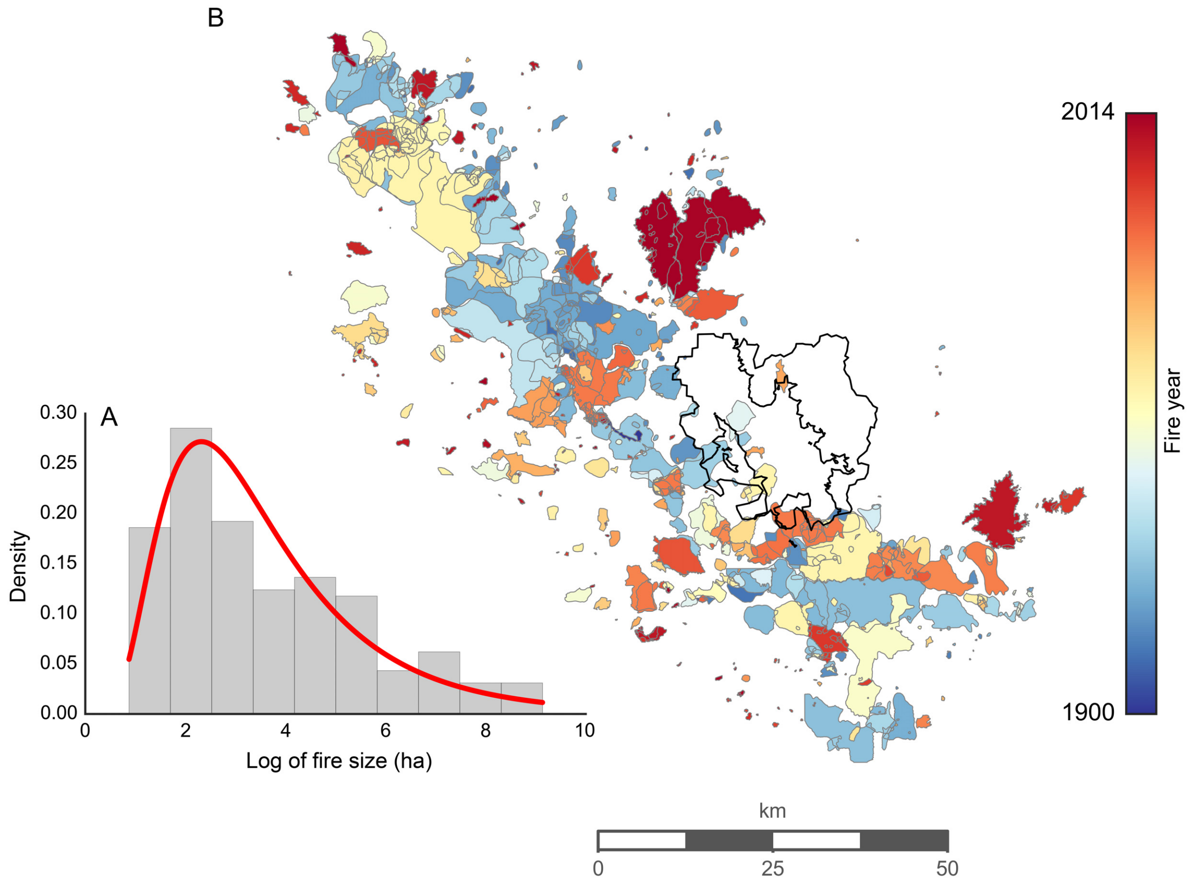

Fig. 3. We gathered historic fire extent data from CALFIRE across a region surrounding the Dinkey Creek watershed, outlined in black (A), and used data over the period 1983 to 2014 to generate a fire size distribution for wildfire in our modeling simulations (B).

event for each grid cell within the fire perimeter. The Dynamic Fire and Fuels extension defines severity as an index of potential mortality, with variation due to species fire tolerance. We held all other fire parameters constant between the two fire weather scenarios, including the fire size distribution and number of ignitions.

The Dynamic Fire and Fuels extension assigns each grid cell a fuel type based on the dominant biomass and age distribution of the vegetation, recent vegetation mortality, and post-disturbance or post-management information present at each time-step (Syphard et al. 2011). Each fuel type is user-defined and associated with fuels parameters that govern fire behavior (Sturtevant et al. 2009). We modified existing fuel parameterizations for species and age distributions in the Sierra Nevada developed by Syphard et al. (2011) and Loudermilk et al. (2014) to better represent the conditions within our study area. These modifications included adjusting parameters that influence fire spread and fire effects, such as canopy base height and build-up index, using local empirical data.

Forest management

We used the Leaf Biomass Harvest extension to simulate thinning and maintenance burning treatments. We divided the study area into management units based on forest type and dominant species, as informed by our initial communities layer and CALVEG (Existing Vegetation—CALVEG (2004) McClellan, CA: USDA Forest Service, Pacific Southwest Region. EvegTile03B_99_04_v2) forest type data for the Dinkey Creek watershed. We only applied treatments to

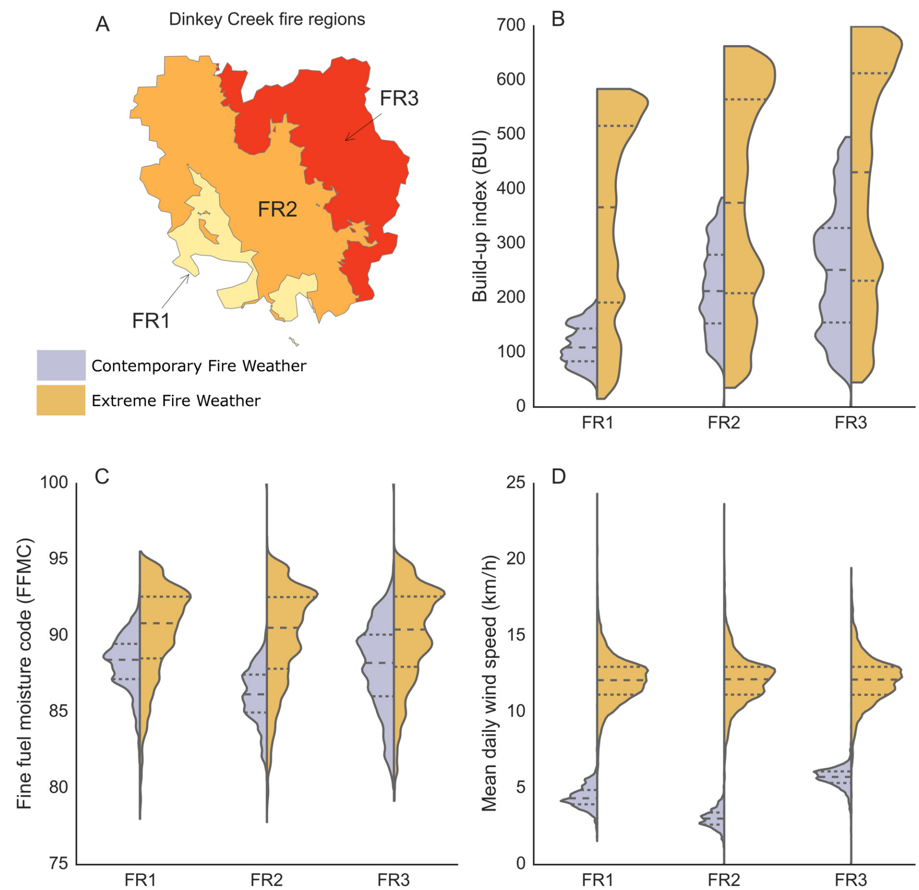

Fig. 4. We leveraged three fire regions across the greater Sierra National Forest developed by Liang et al. (*in press*) for the Dinkey Creek watershed (A) to describe how elevation affects the fuel build-up of fuels in between fire events (B), the interaction between moisture and fine fuels (C), and the wind speed (D), for each region under contemporary (purple) and extreme (tan) fire weather conditions.

forest types that have experienced a significant deviation from their historic mean fire return interval. For each of these forest types, ponderosa pine, pine-dominated mixed-conifer, fir-dominated mixed-conifer, and red fir, we developed forest type-specific thinning and maintenance burning prescriptions. For red fir forest, we excluded thinning as an ecologically appropriate treatment option and only simulated maintenance burning. This decision was based on the fact that the fuels profiles in higher-elevation, red fir forests may not merit thinning prior to reintroducing fire because low-severity fire may be sufficient to restore structural heterogeneity (Kane et al. 2014). The thinning treatments removed approximately 30% of the live tree biomass and included preferential harvest of the youngest cohorts first to reduce forest density, canopy continuity, and height to live crown, which are common objectives for reducing high-severity wildfire risk (North et al. 2009, Stephens et al. 2009). We designed the treatment rates such that the areas identified for mechanical treatment were only thinned once during the simulation and scheduled such that all thinning was completed during the first 10 yr of the simulation. Our goal was to rapidly attain a forest structure that could be maintained by regular surface fire, regardless of the potential operational constraints associated with the rate of treatment. We developed prescribed fire treatments using forest type-specific fire return intervals for ponderosa pine (15-year return interval), pine-dominated mixed-conifer (20-year return interval),

Table 1. Treatment description by forest type.

Forest type	Coverage across the simulation area (ha)	Mean area thinned over 10 yr (ha)	Mean C removal from thinning (Mg C/ha)	Target area treated with maintenance burning (ha/yr)	Target return interval (yr)
Oak woodland	4235	–	–	–	–
Ponderosa pine	9059	1533	21.4 (0.9)	567	15
Pine-dominated mixed-conifer	15,566	2784	30.1 (0.5)	695	20
Fir-dominated mixed-conifer	12,242	2656	28.7 (0.2)	424	25
Red fir	8104	–	–	178	40
Subalpine	4498	–	–	–	–

Notes: The coverage across the simulation area is the area of the watershed that each vegetation type occupies (some of which was not available for treatment in the model). The mean area thinned over 10 yr is the hectares of each forest type that was thinned during the simulation. The mean C removal (standard deviation) from thinning is the mean value across 50 replicate simulations. The per-year area target for maintenance burning is based on the area of the forest type and the target return interval.

fir-dominated mixed-conifer (25-year return interval), and red fir (40-year return interval). In the pine and mixed-conifer management units, the first entry of prescribed fire was applied 10 yr into the simulations, once the mechanical thinning was completed. Prescribed fire began at the start of the simulation for the red fir management unit. We applied prescribed fire to 100% of the area within each management unit over the simulation period, with area burned during each time-step being a function of the historic fire return interval for each forest type. Similar to our accelerated mechanical thinning treatment rate, the prescribed fire treatment rates were not designed to approximate current or planned rates of fire use, but were parameterized to simulate pre-suppression fire frequencies based on available fire reconstruction data and first-entry burns were implemented at an accelerated pace. Accounting for fire frequency variability among forest types, our maintenance burning treatment target was approximately 1864 ha/yr, equivalent to 46% of the treatable forest area per decade. The six management units and their specific treatment combinations, as well as treatment application intervals and rates by simulation, are described in Table 1. Total carbon removal and emission from combined thinning and maintenance burning ranged from 31 to 35 Mg C/ha in treated areas.

Simulation experiment

To investigate the interaction between treatments and fire weather, we simulated three management scenarios (no-management, thin-only, thin and maintenance burning) with two fire weather scenarios (contemporary and extreme), for a total of six scenarios. All other fire parameters were held constant resulting in a consistent number of fires and mean fire size by fire region across all simulations (Fig. 5).

We ran 50 replicates of each scenario for 100 yr, using annual time-steps. We assessed the interactions of fire weather and fuels treatment on landscape fire severity, C stocks and fluxes, and wildfire emissions. We calculated the mean and coefficient of variation (CV) for fire severity using the annual fire severity raster data for the 50 replicate simulations for each scenario. Our calculations only included grid cells for the years in which they burned. We calculated mean and 95% confidence intervals for AGC over the 100-year period using the 50 replicate simulations for each scenario. We compared distributions of the last 5 yr of the simulation period to test for statistical differences in AGC stocks between scenarios. We calculated mean annual NEE by taking the 100-year landscape average NEE by scenario across all 50 replicates. We calculated mean cumulative emissions from wildfire and prescribed fire over the 100-year period using the 50 replicates from each of the scenarios. We used analysis of variance and Tukey's honestly significant difference for mean separation following Bartlett's test for homoscedasticity. For comparisons where data were heteroscedastic, we employed Kruskal–Wallis tests with post hoc Dunn's comparisons. We conducted all model parameterization and output analyses, as well as figure generation using Python (Python Software Foundation. Python Language Reference, version 2.7. http://www.python.org).

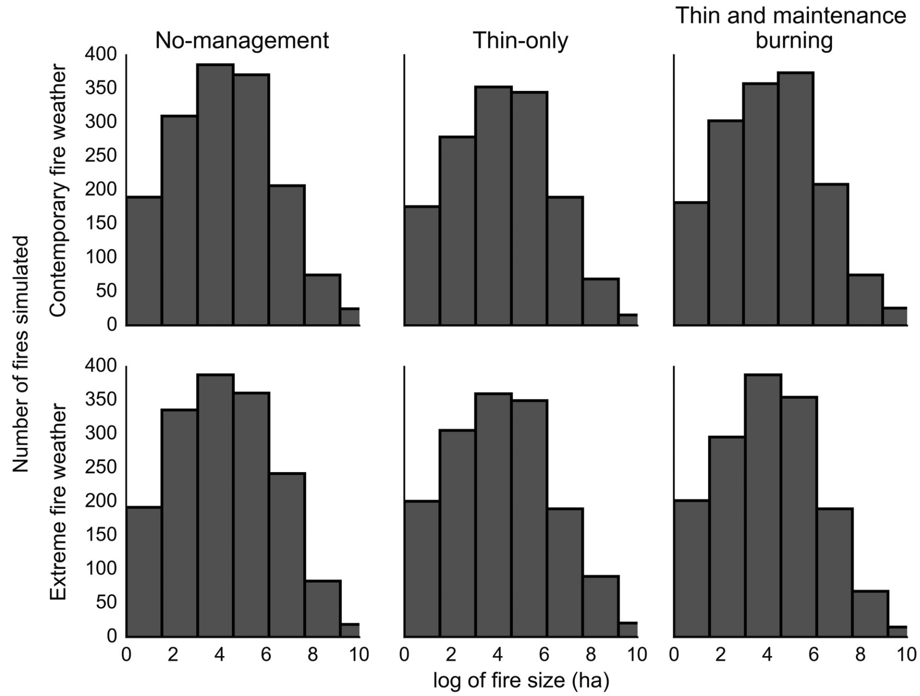

Fig. 5. Fire size distributions for both contemporary and extreme fire weather simulations across the watershed. Here, the size distribution is normalized with the natural log, and counts on the *y*-axis represent the total number of fires across 50 replicates of 100-year simulations per scenario.

Results

Fuels treatments under contemporary fire weather had little impact on mean fire severity across the watershed relative to the no-management scenario (<1% mean decrease for both thin-only and thin and maintenance burning treatments), due to the already low mean fire severity that resulted from the contemporary fire weather distribution (Fig. 6). The areas that showed the largest reduction in mean severity from management were areas of especially high biomass, but given that the majority of wildfires under contemporary fire weather had lower severity, these treatment effects had very little impact relative to the no-management scenario. Under contemporary fire weather, the CV of fire severity was generally low across management scenarios. Similar to the changes in mean fire severity following management, the greatest changes in the CV of fire severity were realized in high-biomass areas (Fig. 7). Relative to the no-management scenario, both management scenarios reduced fire size slightly, decreasing the average total number of hectares burned from 20,876 in the no-management to 18,756 in the thin-only and 19,106 in the thin and maintenance burning (Fig. 8).

The extreme fire weather scenarios showed much larger differences in mean wildfire severity across the landscape, with the thin-only scenario increasing mean fire severity by 1.7% and the thin and maintenance burning treatment decreasing mean fire severity across the watershed by 25% relative to no-management (Fig. 6). This resulted in the area that experienced high mean severity (mean fire severity >3) ranging from 19% in the no-management scenario and 17% in the thin-only scenario, to 1.6% in the thin and maintenance burning scenario. Similar to the contemporary fire weather scenario, fire severity in the no-management scenario under extreme fire weather conditions generally tracked the spatial distribution of biomass, with areas of increased C density typically having higher mean fire severity. Treatment of these areas with thinning alone had little effect on mean severity because of the relatively short duration of the single-entry thinning prescription in modifying

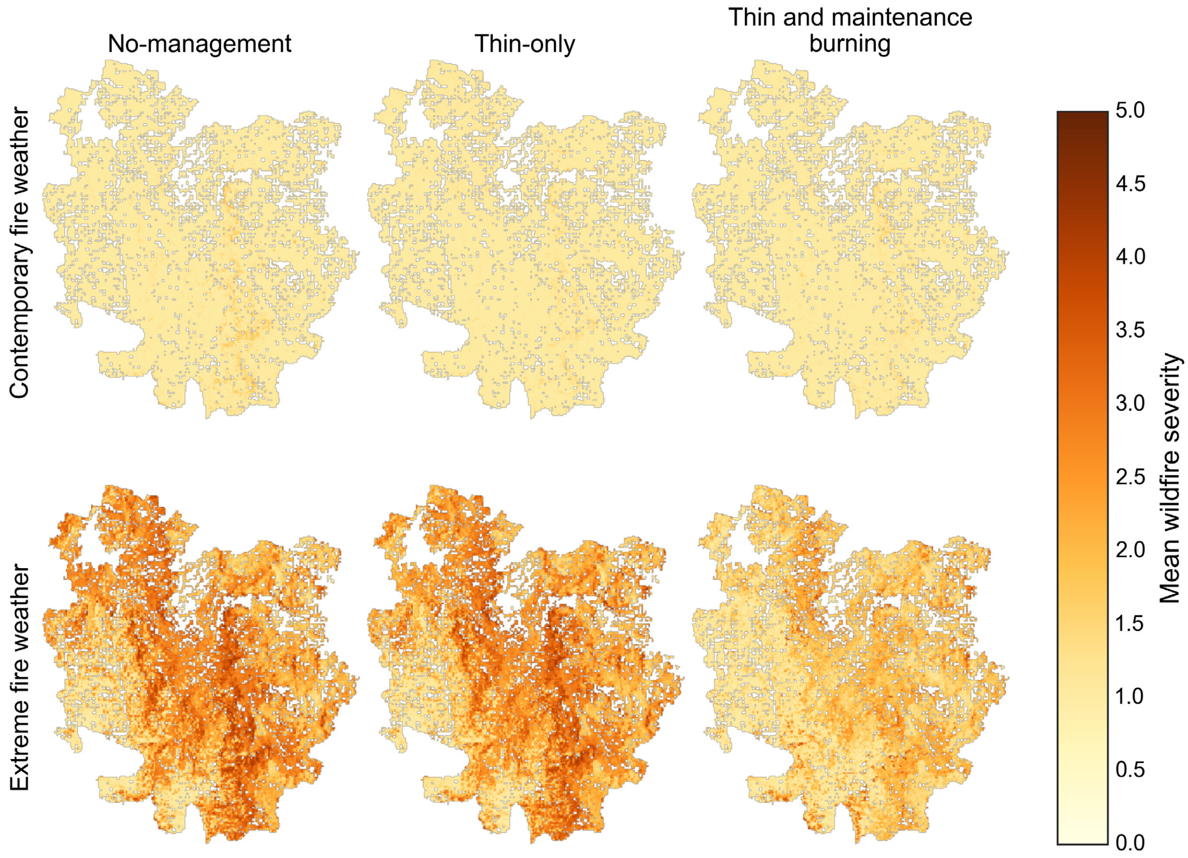

Fig. 6. Mean wildfire severity for the 50 replicates of 100-year simulations across the Dinkey Creek watershed. Fire severity ranges from 0 to 5, where 0 indicates no fires occurred, 1–3 are low- and mixed-severity fires, and 4–5 are fires that resulted in high mortality and crown fire.

forest structure and the increase in shrub connectivity resulting from increased light availability due to opening of the canopy. When the thinning treatments were followed by regular maintenance burning, tree regeneration and shrub growth were reduced, consequently reducing mean fire severity. Under extreme fire weather, the CV of fire severity increased over the contemporary values for all management scenarios, indicating that a larger range of fire severities occurred across the majority of the landscape (Fig. 7). In the context of mean fire severity (Fig. 6) under extreme fire weather, the CVs for no-management and thin-only indicate that large portions of the landscape consistently experienced higher-severity wildfire. However, in the thin and maintenance burning scenario under extreme fire weather, the mean severity and CV were consistently low across most of the landscape, with higher CV occurring in the highest biomass areas. The thin and maintenance burning result indicates that the timing between stochastic wildfire events and time since prescribed fire is an important characteristic for moderating fire behavior. In addition to the increase in mean fire severity, area burned under extreme fire weather increased slightly over area burned under contemporary fire weather in the no-management (1001 ha) and thin-only (3490 ha). However, in the thin and maintenance burn, there was little difference in area burned between contemporary and extreme fire weather and surface fires (severity 1 and severity 2) burned the majority of the landscape (Fig. 8). Similar to the trends seen with mean severity, the increased light availability following the opening of the canopy during thinning and subsequent increase in shrub fuels continuity

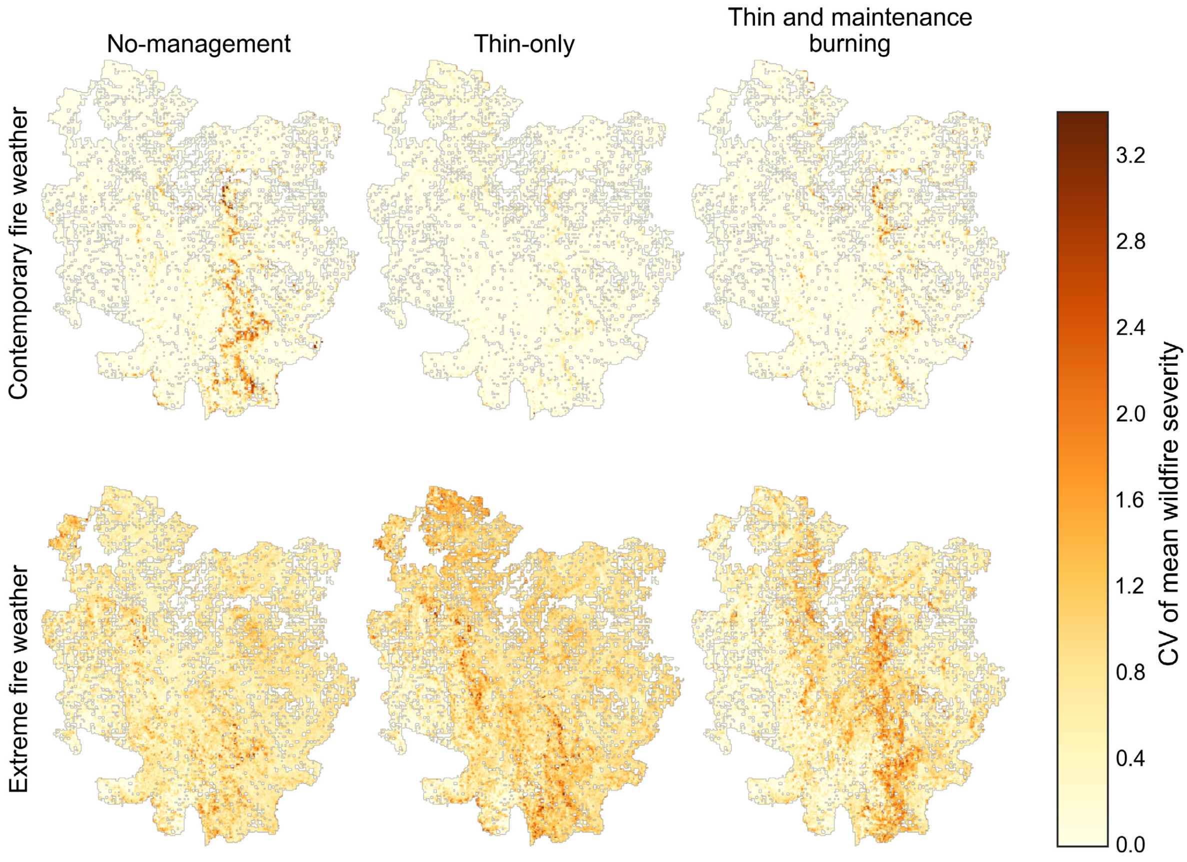

Fig. 7. Coefficient of variation (CV) of the simulated wildfire severity across the Dinkey Creek watershed.

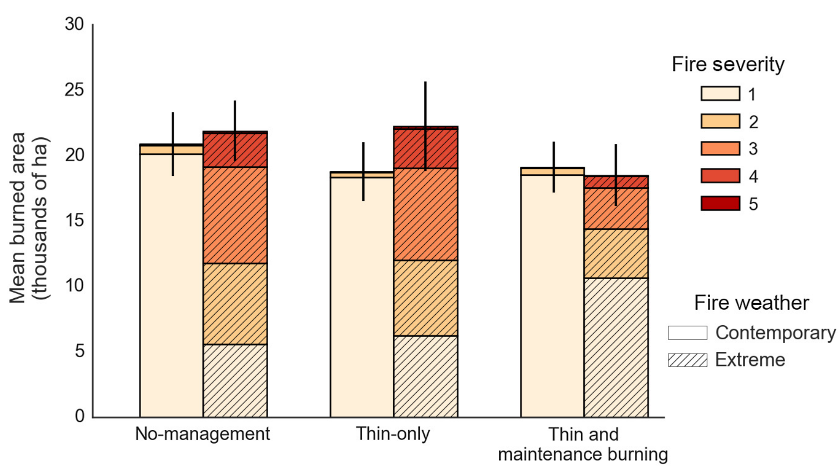

Fig. 8. Mean burned area by severity class under management and fire weather scenarios. Bars represent the total area burned in thousands of hectares on average over the 100-year simulations by fire severity class. Solid bars represent contemporary fire weather simulations, and hashed bars represent the extreme fire weather simulations. Error bars represent ±1 standard error of the mean total burned area.

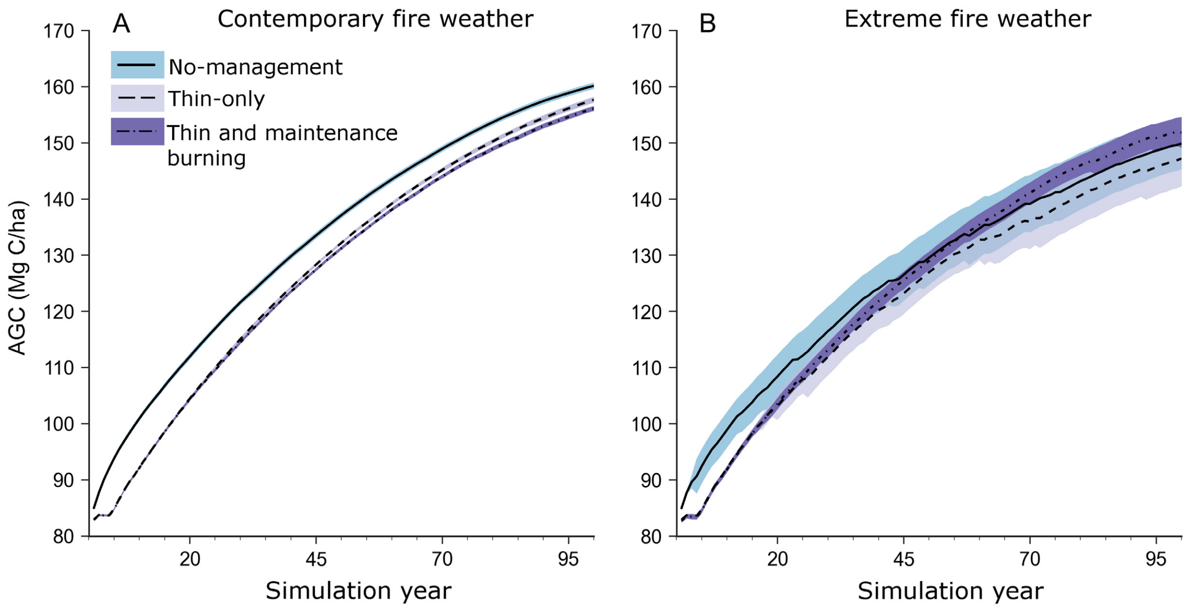

Fig. 9. Aboveground carbon (AGC) over the course of the 100 yr of simulation for three management plans: no-management (light blue), thin-only (light purple), and thinning with maintenance burning (dark purple), under contemporary (A) and extreme (B) fire weather conditions. Time series shown are the mean of 50 replicate simulations, with a 95% confidence interval.

actually resulted in slightly larger fires in the thin-only scenario relative to the no-management. Subsequent entry with prescribed fire reduced the total hectares burned to contemporary fire weather levels.

Treatment and its effects on fire severity under the two different fire weather scenarios resulted in altered landscape C dynamics. We had hypothesized that treatments would only yield an increase in landscape C storage when they reduced fire severity relative to no-management. The low mean severities across management scenarios under contemporary fire weather resulted in significantly lower AGC at the landscape scale for both the thin-only and thin and maintenance burning treatments (Fig. 9A). During the last 5 yr of simulation, AGC was significantly higher ($P < 0.005$) in the no-management scenario (159.7 Mg C/ha) than in either the thin-only (156.9 Mg C/ha) or the thin and maintenance burning (155 Mg C/ha). Under extreme fire weather, there were no significant differences in AGC between the no-management (149.3 Mg C/ha), thin-only scenario (146.5 Mg C/ha), and thin and maintenance burning scenarios (151.5 Mg C/ha, Fig. 9B). However, the thin and maintenance burning scenario significantly reduced variance in end-of-simulation AGC compared to no-management and thin-only scenarios ($P < 0.001$). The lack of an impact of the thin-only prescription on the variance of the end-of-simulation AGC was in part driven by the influence of shrubs and their competition for light with tree seedlings and contribution to surface fuel continuity. While shrubs in this system do not make a significant contribution to AGC or total ecosystem carbon (Wiechmann et al. 2015), they can increase continuity of the fuels layer when tree canopy cover is reduced, by resprouting and quickly reestablishing following fire.

We had hypothesized that the thin and maintenance burning scenarios would result in a stronger C sink than either of the other scenarios. Under contemporary fire weather, we found no treatment differences in NEE across the landscape (Fig. 10). However, under extreme fire weather, we found significant decreases in the strength of the C sink for both the no-management and thin-only management scenarios. However, the C sink in the thin and maintenance burning scenario was not significantly different from the outcome for this treatment under contemporary fire weather

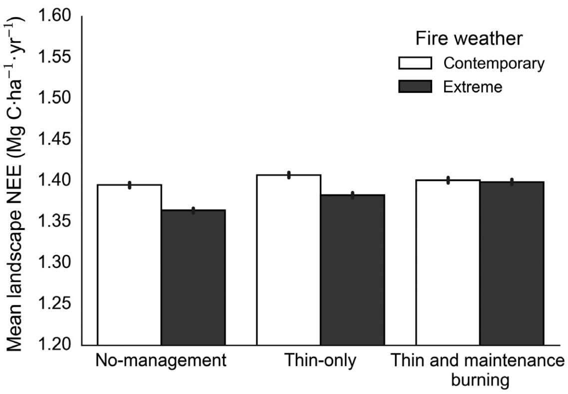

Fig. 10. Landscape mean net ecosystem exchange (NEE) of carbon over the 100-year simulation period. Error bars represent the 95% confidence interval around the mean generated from the 50 replicate simulations.

(Fig. 10), indicating that the interaction between fire weather and fuels is an important determinant of carbon exchange between the forest and atmosphere.

Wildfire-driven C emissions varied by treatment and fire weather severity (Fig. 11). Under contemporary fire weather, mean cumulative wildfire emissions were similar across all three treatment types (no-management = 0.9 Mg C/ha, thin-only = 0.7 Mg C/ha, and thin and maintenance burning = 0.8 Mg C/ha), with the added emissions from prescribed fire resulting in a significantly higher cumulative carbon emission in the thin and maintenance burning scenario (3.27 Mg C/ha, $P < 0.001$). Consistent with our hypothesis, extreme fire weather coupled with the thin and maintenance burning prescription resulted in significantly lower cumulative wildfire carbon emissions compared to no-management and thin-only scenarios (no-management = 27.6 Mg C/ha, thin-only = 28.4 Mg C/ha, thin and maintenance burning = 11.4 Mg C/ha, $P < 0.001$) even with the inclusion of the emissions from prescribed fire adding an additional 2.5 Mg C/ha (total emissions = 13.9 Mg C/ha, $P < 0.001$). Further, under extreme fire weather, the variance of cumulative carbon emission from wildfire was significantly lower in the thin and maintenance burning treatment relative to the no-management and thin-only scenarios ($P < 0.001$).

Discussion

Reductions in burn severity following management are well documented in this region (Stephens et al. 2009, North and Hurteau 2011), but under our contemporary fire weather simulations we saw little change in mean fire severity between management scenarios across the landscape (Fig. 6). We also found little difference in the CV of fire severity between management scenarios (Fig. 7), indicating that the majority of simulated fires had low severity (Fig. 8). This resulted from the rarity of wildfire (roughly a 1 in 125 chance of wildfire occurring in our simulations), coupled with the relatively benign weather conditions in our contemporary fire weather simulations (Fig. 4).

When we accounted for the increasing frequency of extreme fire weather, the thin and maintenance burning scenario had a large reduction (>25%) in mean fire severity across the landscape, decreasing the portion of the landscape that burned at high severity by an order of magnitude compared to no-management. The CV of fire severity for the thin and maintenance burning scenario demonstrates that mean severity was consistently lower across much of the landscape, with high-biomass areas having the most variability between fire events. Our CV of fire severity results for the thin and maintenance burning scenario suggests that even in areas where significant reductions in fire severity are possible through thinning and maintenance burning, the occasional crown killing fire is still possible if weather conditions allow.

Interestingly, thinning alone proved to be inadequate for modifying fire behavior under extreme fire weather conditions. These disparate results between the thin-only and thin and maintenance burning treatments are primarily due to the development of the understory shrub layer following the reduction in canopy cover from thinning and the increased flammability of the shrub layer under extreme fire weather, as evidenced by the CV of fire severity for the thin-only (Fig. 7). Previous empirical work in mixed-conifer forest in the southern Sierra found that 10 yr following thinning the amount of shrub C increased by 100–200% (Wiechmann et al. 2015). Subsequent prescribed burning simulated in our

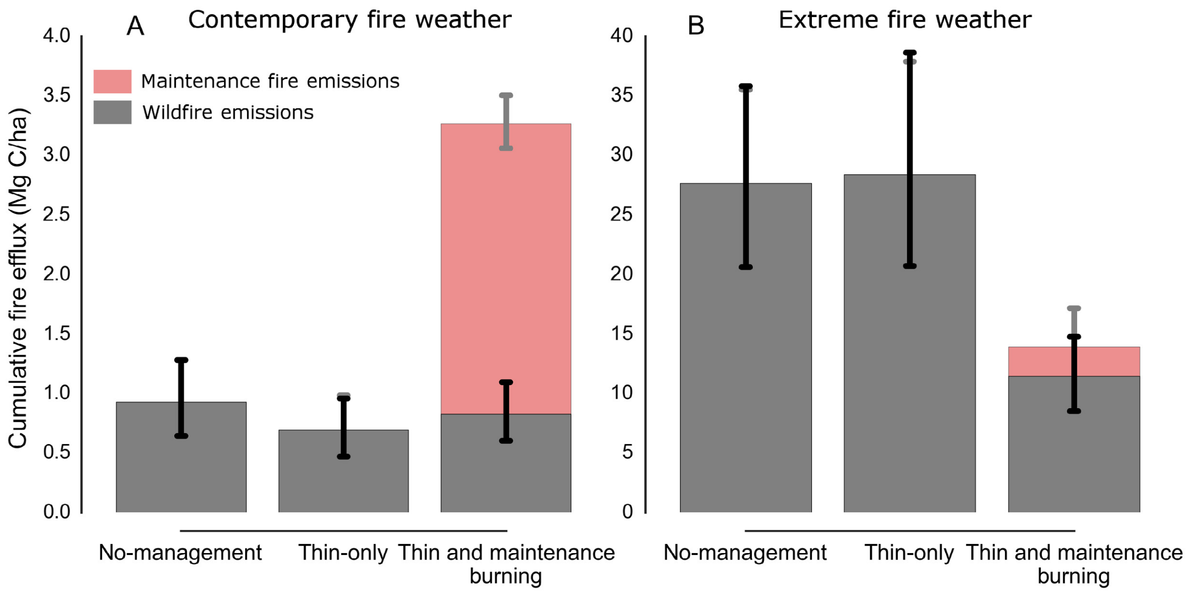

Fig. 11. Cumulative C emissions from fire following 100 yr of simulation for contemporary (A) and extreme (B) fire weather. Gray bars represent emission from wildfire, whereas the red bar on the thin and maintenance burn scenario adds the emissions generated from prescribed burning. Error bars represent the 95% confidence interval around the mean generated from 50 replicates. Note the *y*-axis scale varies between panels.

thin and maintenance burning scenario helped reduce shrub biomass and connectivity, which reduced fire severity under the extreme weather conditions that allow the shrub layer to carry fire. Specifically, the interaction between high air temperatures and low relative humidity in our extreme fire weather scenario resulted in greatly reduced fuel moisture and subsequently an increase in surface fuel build-up (Fig. 4). This change in fuels coupled with the greatly increased maximum wind speeds allowed fires to carry faster across any given fuel type. As a result, the interaction of forest structure and fuel loads with extreme fire weather altered the proportion of the landscape that was impacted by different fire severity classes (Fig. 8).

While management options exist for moderating wildfire behavior and reducing the risk of high-severity wildfire, they require upfront C reductions. We found that thinning alone reduced AGC by 21–30 Mg C/ha within managed areas. When prescribed fire was applied repeatedly, this reduced AGC an additional 3–11 Mg C/ha over the course of the simulation period in areas treated with prescribed fire. Previous research has demonstrated that these upfront C reductions can yield a range of outcomes, from a net reduction to a net increase in C, over time (Campbell et al. 2012, Hurteau et al. 2016). Under contemporary fire weather, the upfront C reductions from treatment yielded end-of-simulation landscape C values that were lower than the no-management scenario because of the low frequency of high-severity wildfires (Fig. 8). Recent significant increases in the frequency of extreme fire weather (Collins 2014) signify that the potential for this outcome is becoming less likely. When we accounted for the increasing frequency of extreme fire weather by using weather data from one of the largest recorded wildfires in California's history in our extreme fire weather simulations, we did not find significant differences in end-of-simulation AGC between management scenarios. However, the variance in AGC across the simulation replicates was significantly reduced for the thin and maintenance burning scenario, consistent with previous work in the Sierra Nevada that found increased C stability in fire-maintained forest (Earles et al. 2014).

Previous research has shown that the influence of management to moderate fire behavior on forest C dynamics varies by forest type. In xeric systems, such as the southwestern United States, treated forest C stocks can surpass those of an unmanaged landscape because of the relatively high probability of wildfire and low productivity of the system (Hurteau et al. 2016). In more productive forest types with a lower probability of wildfire, such as Douglas-fir forests in the Pacific Northwestern United States, the costs of treatment yield lower landscape C than foregoing management (Mitchell et al. 2009, Laflower et al. 2016). Our AGC results indicate that in the Sierra Nevada, the C outcome of treatment is sensitive to the fire weather distribution and its influence on fire effects.

As demonstrated by previous research, forest C loss through combustion varies as a function of fire severity, and can impact forest C balance for years to decades (Meigs et al. 2009, Meigs et al. 2011, Dore et al. 2012). Similarly, our results demonstrate that extreme fire weather and the area affected by severe wildfire can influence landscape NEE and cumulative fire emissions (Figs. 10, 11). Under contemporary fire weather, we found no difference in NEE or cumulative wildfire emissions between management scenarios until we accounted for the cumulative prescribed fire emissions in the thin and maintenance burning scenario. The addition of regular burning caused significantly higher cumulative fire emissions. Under extreme fire weather, management scenario differences in mean landscape NEE and cumulative wildfire emissions were significantly different. As we had hypothesized, the thin and maintenance burning scenario had higher mean landscape NEE and lower cumulative emissions, even after accounting for emissions from prescribed burning (Figs. 10, 11B). The net effect of thinning followed by regular maintenance burning is an overall reduction in tree mortality when wildfire occurs, leading to increased C uptake. The effects of treatment in moderating wildfire-induced tree mortality are consistent with empirical research in the Sierra that found high mortality in untreated stands and mortality concentrated in smaller trees in treated stands (North and Hurteau 2011). While restoring forest structure and fire regimes may initially diminish C stocks, by moderating fire behavior these efforts reduce fire-driven reductions in forest C uptake under increasingly common severe fire weather.

This study tested hypotheses about forest management practices in the context of stochastic wildfire. Our results represent potential future outcomes that are dependent on the weather distributions we used in the two fire weather scenarios. To capture the variability within each fire weather distribution, we leveraged replication and model stochasticity to create an ensemble of model outputs, from which we generated means and confidence intervals. This approach does not capture the changing frequency of extreme fire weather, which could increase the variability in fire severity. Furthermore, we did not account for the influence of projected climate on forest C dynamics, which could alter the forest growth response. However, previous research demonstrated that restoring surface fire confers C stock stability under increasing drought frequency (Earles et al. 2014).

Furthermore, our results must be considered in the context of our simulated treatment rates. Our initial rates of thinning and first-entry burning were far more aggressive than those proposed or implemented in the Dinkey Creek Collaborative Forest Landscape Restoration Project (CFLRP). Our objective in using this accelerated rate of initial treatment was to quickly achieve ecologically appropriate fire regimes for our range of forest types. Once these initial treatments were implemented, our annual maintenance burning targets (Table 1) were similar to Dinkey Creek CFLRP planned rates. Given the significant area in need of fire restoration in the Sierra Nevada, coupled with increasing fire weather severity and operational limitations (North et al. 2012, Collins 2014), managing natural fire ignitions to achieve heterogeneous fire effects presents an opportunity for moving fire-suppressed forests toward a more ecologically resilient condition (Stephens et al. 2016). Our results, in terms of both the mean and CV of fire severity, suggest that under the current distribution of fire weather, there are opportunities for managing natural ignitions to meet management objectives.

Given that fire severity in the Sierra Nevada has increased as a result of long-term fire exclusion and given the higher likelihood of a subsequent high-severity fire following an initial high-severity fire, the potential exists for a transition toward a lower C state system with ongoing climate change and increasing area burned (Miller et al. 2009, Hurteau and Brooks 2011, Coppoletta et al. 2016; Liang et al., *in press*). Our results suggest that capitalizing on contemporary fire weather to accomplish restoring fire regimes provides an increase in forest C stability when wildfire burns under extreme fire weather.

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Literature Cited

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

Ager, A. A., N. M. Vaillant, and A. McMahan. 2013. Restoration of fire in managed forests: a model to prioritize landscapes and analyze tradeoffs. Ecosphere 4:1–19.

Beaty, R. M., and A. H. Taylor. 2008. Fire history and the structure and dynamics of a mixed conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. Forest Ecology and Management 255:707–719.

Bonan, G. B. 2008. Forests and climate change: forcings, feedbacks, and the climate benefits of forests. Science 320:1444–1449.

Campbell, J. L., and A. A. Ager. 2013. Forest wildfire, fuel reduction treatments, and landscape carbon stocks: a sensitivity analysis. Journal of Environmental Management 121:124–132.

Campbell, J. L., M. E. Harmon, and S. R. Mitchell. 2012. Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions? Frontiers in Ecology and the Environment 10:83–90.

Caprio, A. C. and T. W. Swetnam. 1993. Historical Fire regimes along an elevation gradient on the west slope of the Sierra Nevada, California. Pages 173–179 *in* J. K. Brown, R. W. Mutch, C. W. Spoon, and R. H. Wakimoto, technical coordinators. Proceedings: Symposium on Fire in Wilderness and Park Management: Past Lessons and Future Opportunities, Missoula, MT, March 30–April 1, 1993. INT-GTR-320. USDA Forest Service, Intermountain Research Station, Ogden, Utah, USA.

Chojnacky, D. C., L. S. Heath, and J. C. Jenkins. 2014. Updated generalized biomass equations for North American tree species. Forestry 87:129–151.

Collins, B. M. 2014. Fire weather and large fire potential in the northern Sierra Nevada. Agricultural and Forest Meteorology 189–190:30–35.

Collins, B. M., H. S. 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.

Coppoletta, M., K. E. Merriam, and B. M. Collins. 2016. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecological Applications 26:686–699.

Dore, S., M. Montes-Helu, S. C. Hart, B. A. Hungate, G. W. Koch, J. B. Moon, A. J. Finkral, and T. E. Kolb. 2012. Recovery of ponderosa pine ecosystem carbon and water fluxes from thinning and stand-replacing fire. Global Change Biology 18: 3171–3185.

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.

Finkral, A. J., and A. M. Evans. 2008. The effects of a thinning treatment on carbon stocks in a northern Arizona ponderosa pine forest. Forest Ecology and Management 255:2743–2750.

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.

Hessburg, P. F., J. K. Agee, and J. F. Franklin. 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management 211:117–139.

Hurteau, M. D. 2013. Effects of wildland fire management on forest carbon stores. Pages 359–380 *in* D. G. Brown, D. T. Robinson, N. H. F. French, and B. C. Reed, editors. Land use and the carbon cycle: science applications in human environment interactions. Cambridge University Press, Cambridge, UK.

Hurteau, M. D., J. B. Bradford, P. Z. Fulé, A. H. Taylor, and K. L. Martin. 2014. Climate change, fire management, and ecological services in the southwestern US. Forest Ecology and Management 327: 280–289.

Hurteau, M. D., and M. L. Brooks. 2011. Short- and long-term effects of Fire on carbon in US dry temperate forest systems. BioScience 61:139–146.

Hurteau, M. D., G. W. Koch, and B. A. Hungate. 2008. Carbon protection and fire risk reduction: toward a full accounting of forest carbon offsets. Frontiers in Ecology and the Environment 6:493–498.

Hurteau, M. D., S. Liang, K. L. Martin, M. P. North, G. W. Koch, and B. A. Hungate. 2016. Restoring forest structure and process stabilizes forest carbon in wildfire-prone southwestern ponderosa pine forests. Ecological Applications 26:382–391.

Jenkins, J. C., D. C. Chojnacky, L. S. Heath, and R. A. Birdsey. 2003. National-scale biomass estimators for United States tree species. Forest Science 49: 12–35.

Kane, V. R., M. P. North, J. A. Lutz, D. J. Churchill, S. L. Roberts, D. F. Smith, R. J. McGaughey, J. T. Kane, and M. L. Brooks. 2014. Assessing fire effects on forest spatial structure using a fusion of Landsat and airborne LiDAR data in Yosemite National Park. Remote Sensing of Environment 151:89–101.

Kretchun, A. M., E. L. Loudermilk, R. M. Scheller, M. D. Hurteau, and S. Belmecheri. 2016. Climate and bark beetle effects on forest productivity—linking dendroecology with forest landscape modeling. Canadian Journal of Forest Research 46:1026–1034.

Laflower, D. M., M. D. Hurteau, G. W. Koch, M. P. North, and B. A. Hungate. 2016. Climate-driven changes in forest succession and the influence of management on forest carbon dynamics in the Puget Lowlands of Washington State, USA. Forest Ecology and Management 362:194–204.

Liang, S., M. D. Hurteau, and A. L. Westerling. *In press*. Response of Sierra Nevada forests to projected climate-wildfire interactions. Global Change Biology. https://doi.org/10.1111/gcb.13544

Loudermilk, E. L., R. M. Scheller, P. J. Weisberg, J. Yang, T. E. Dilts, S. L. Karam, and C. Skinner. 2013. Carbon dynamics in the future forest: the importance of long-term successional legacy and climate-fire interactions. Global Change Biology 19:3502–3515.

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.

Marlon, J. R., et al. 2012. Long-term perspective on wildfires in the western USA. Proceedings of the National Academy of Sciences USA 109:E535–E543.

Martin, K. L., M. D. Hurteau, B. A. Hungate, G. W. Koch, and M. P. North. 2015. Carbon tradeoffs of restoration and provision of endangered species habitat in a fire-maintained forest. Ecosystems 18:76–88.

McGinnis, T. W., J. E. Keeley, S. L. Stephens, and G. B. Roller. 2010. Fuel buildup and potential fire behavior after stand-replacing fires, logging fire-killed trees and herbicide shrub removal in Sierra Nevada forests. Forest Ecology and Management 260:22–35.

Meigs, G. W., D. C. Donato, J. L. Campbell, J. G. Martin, and B. E. Law. 2009. Forest fire impacts on carbon uptake, storage, and emission: the role of burn severity in the Eastern Cascades, Oregon. Ecosystems 12:1246–1267.

Meigs, G. W., D. P. Turner, W. D. Ritts, Z. Yang, and B. E. Law. 2011. Landscape-scale simulation of heterogeneous fire effects on pyrogenic carbon emissions, tree mortality, and net ecosystem production. Ecosystems 14:758–775.

Millar, C. I., and N. L. Stephenson. 2015. Temperate forest health in an era of emerging megadisturbance. Science 349:823–826.

Miller, J. D., H. D. Safford, M. Crimmins, and A. E. Thode. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. Ecosystems 12:16–32.

Mitchell, R. J., J. K. Hiers, J. Obrien, and G. Starr. 2009. Ecological forestry in the southeast: understanding the ecology of fuels. Journal of Forestry 107: 391–397.

Moritz, M. A., M.-A. Parisien, E. Batllori, M. A. Krawchuk, J. Van Dorn, D. J. Ganz, and K. Hayhoe. 2012. Climate change and disruptions to global fire activity. Ecosphere 3:49.

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

North, M. P., and M. D. Hurteau. 2011. High-severity wildfire effects on carbon stocks and emissions in fuels treated and untreated forest. Forest Ecology and Management 261:1115–1120.

North, M., M. Hurteau, and J. Innes. 2009. Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions. Ecological Applications 19:1385–1396.

Parton, W. J., et al. 1993. Observations and modeling of biomass and soil organic matter dynamics for the grassland biome worldwide. Global Biogeochemical Cycles 7:785–803.

Saxton, K. E., W. J. Rawls, J. S. Romberger, and R. I. Papendick. 1986. Estimating generalized soil-water characteristics from texture. Soil Science Society of America Journal 50:1031–1036.

Scheller, R. M., J. B. Domingo, B. R. Sturtevant, J. S. Williams, A. Rudy, E. J. Gustafson, and D. J. Mladenoff. 2007. Design, development, and application of LANDIS-II, a spatial landscape simulation model with flexible temporal and spatial resolution. Ecological Modelling 201:409–419.

Scheller, R. M., D. Hua, P. V. Bolstad, R. A. Birdsey, and D. J. Mladenoff. 2011*a*. The effects of forest harvest intensity in combination with wind disturbance on carbon dynamics in Lake States Mesic Forests. Ecological Modelling 222:144–153.

Scheller, R. M., and D. J. Mladenoff. 2008. Simulated effects of climate change, fragmentation, and interspecific competition on tree species migration in northern Wisconsin, USA. Climate Research 36: 191–202.

Scheller, R. M., S. van Tuyl, K. L. Clark, J. Hom, and I. La Puma. 2011*b*. Carbon sequestration in the New Jersey Pine Barrens under different scenarios of fire management. Ecosystems 14:987–1004.

Scholl, A. E., and A. H. Taylor. 2006. Regeneration patterns in old-growth red fir-western white pine forests in the northern Sierra Nevada, Lake Tahoe, USA. Forest Ecology and Management 235:143–154.

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. 2013. Soil Survey Geographic (SSURGO) Database. http://sdmdataaccess.nrcs.usda.gov/

Stephens, S. L., R. E. Martin, and N. E. Clinton. 2007. Prehistoric fire area and emissions from California's forests, woodlands, shrublands, and grasslands. Forest Ecology and Management 251:205–216.

Stephens, S. L., J. D. McIver, R. E. J. Boerner, C. J. Fettig, J. B. Fontaine, B. R. Hartsough, P. L. Kennedy, and D. W. Schwilk. 2012. The effects of forest fuel-reduction treatments in the United States. BioScience 62:549–560.

Stephens, S. L., J. J. Moghaddas, B. R. Hartsough, E. E. Y. Moghaddas, and N. E. Clinton. 2009. Fuel treatment effects on stand-level carbon pools, treatment-related emissions, and fire risk in a Sierra Nevada mixed-conifer forest. Canadian Journal of Forest Research 39:1538–1547.

Stephens, S. L., B. M. Collins, E. Biber, and P. Z. Fule. 2016. U.S. Federal fire and forest policy: emphasizing resilience in dry forests. Ecosphere 7:e01584.

Sturtevant, B. R., R. M. Scheller, B. R. Miranda, D. Shinneman, and A. Syphard. 2009. Simulating dynamic and mixed-severity fire regimes: a process-based fire extension for LANDIS-II. Ecological Modelling 220:3380–3393.

Syphard, A. D., R. M. Scheller, B. C. Ward, W. D. Spencer, and J. R. Strittholt. 2011. Simulating landscape-scale effects of fuels treatments in the Sierra Nevada, California, USA. International Journal of Wildland Fire 20:364–383.

Thornton, P. E., M. M. Thornton, B. W. Mayer, Y. Wei, R. Devarakonda, R. S. Vose, and R. B. Cook. 2012. Daymet: Daily Surface Weather Data on a 1-km Grid for North America, Version 3. ORNL DAAC, Oak Ridge, Tennessee, USA.

van der Werf, G. R., J. T. Randerson, L. Giglio, G. J. Collatz, M. Mu, P. S. Kasibhatla, D. C. Morton, R. S. Defries, Y. Jin, and T. T. Van Leeuwen. 2010. Global fire emissions and the contribution of deforestation, savanna, forest, agricultural, and peat fires (1997–2009). Atmospheric Chemistry and Physics 10:11707–11735.

Westerling, A. L. 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society of London B: Biological Sciences 371:20150178.

Westerling, A. L., B. P. Bryant, H. K. Preisler, T. P. Holmes, H. G. Hidalgo, T. Das, and S. R. Shrestha. 2011*a*. Climate change and growth scenarios for California wildfire. Climatic Change 109(Suppl 1): S445–S463.

Westerling, A. L., M. G. Turner, E. A. H. Smithwick, W. H. Romme, and M. G. Ryan. 2011*b*. Continued warming could transform Greater Yellowstone fire regimes by mid-21st century. Proceedings of the National Academy of Sciences USA 108:13165–13170.

Wiechmann, M. L., M. D. Hurteau, M. P. North, G. W. Koch, and L. Jerabkova. 2015. The carbon balance of reducing wildfire risk and restoring process: an analysis of 10-year post-treatment carbon dynamics in a mixed-conifer forest. Climatic Change 132: 709–719.

Wiedinmyer, C., S. K. Akagi, R. J. Yokelson, L. K. Emmons, J. A. Al-Saadi, J. J. Orlando, and A. J. Soja. 2011. The Fire INventory from NCAR (FINN) – a high resolution global model to estimate the emissions from open burning. Geoscientific Model Development 3:2439–2476.

Wiedinmyer, C., and M. D. Hurteau. 2010. Prescribed fire as a means of reducing forest carbon emissions in the western United States. Environmental Science and Technology 44:1926–1932.

Xu, C., G. Z. Gertner, and R. M. Scheller. 2009. Uncertainties in the response of a forest landscape to global climatic change. Global Change Biology 15: 116–131.