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Research, part of a Special Feature on Adaptation in Fire-Prone Landscapes: Interactions of Policies, Management, Wildfire, and Social Networks in Oregon, USA

Effects of accelerated wildfire on future fire regimes and implications for the United States federal fire policy

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ABSTRACT. Wildland fire suppression practices in the western United States are being widely scrutinized by policymakers and scientists as costs escalate and large fires increasingly affect social and ecological values. One potential solution is to change current fire suppression tactics to intentionally increase the area burned under conditions when risks are acceptable to managers and fires can be used to achieve long-term restoration goals in fire adapted forests. We conducted experiments with the Envision landscape model to simulate increased levels of wildfire over a 50-year period on a 1.2 million ha landscape in the eastern Cascades of Oregon, USA. We hypothesized that at some level of burned area fuels would limit the growth of new fires, and fire effects on the composition and structure of forests would eventually reduce future fire intensity and severity. We found that doubling current rates of wildfire resulted in detectable feedbacks in area burned and fire intensity. Area burned in a given simulation year was reduced about 18% per unit area burned in the prior five years averaged across all scenarios. The reduction in area burned was accompanied by substantially lower fire severity, and vegetation shifted to open forest and grass-shrub conditions at the expense of old growth habitat. Negative fire feedbacks were slightly moderated by longer-term positive feedbacks, in which the effect of prior area burned diminished during the simulation. We discuss trade-offs between managing fuels with wildfire versus prescribed fire and mechanical fuel treatments from a social and policy standpoint. The study provides a useful modeling framework to consider the potential value of fire feedbacks as part of overall land management strategies to build fire resilient landscapes and reduce wildfire risk to communities in the western U.S. The results are also relevant to prior climate-wildfire studies that did not consider fire feedbacks in projections of future wildfire activity.

Key Words: *Envision; forest landscape disturbance modeling; forest restoration; wildfire feedbacks; wildfire simulation; wildfire suppression policy*

INTRODUCTION

Policies and planning efforts in the U.S. aimed at curbing wildfire impacts to social and ecological values increasingly recognize that current suppression policies are not financially sustainable and not desirable from an ecological standpoint (North et al. 2015, USDA OIG 2016). Spiraling suppression costs are eroding agency budgets allocated to restoration and conservation programs, and the effectiveness of suppression efforts to reduce the growth of extreme, long duration wildfire events is increasingly questioned (Calkin et al. 2015). The long-term effects of fire suppression policies and practices have led to widespread densification of fire-frequent conifer forests and changes in fire regimes across much of the western U.S. (Arno and Brown 1991, Noss et al. 2006, Collins et al. 2013). Accelerated restoration programs (USDA FS 2012) are finding success in local contexts (USDA FS 2015*a*), but have not arrested the upward trend in burned area and risks to socioeconomic values. Changing the current trajectory in area burned by uncharacteristic fire in fire adapted forests, i.e., those with undesirable ecological consequences, will require more substantial reduction in fuels over wide areas to reduce the wildfire deficit throughout the western U.S. and change large scale wildfire behavior.

Newer U.S. federal wildfire policies have recognized that current mechanical fuel treatment programs alone are insufficient, and that safe and effective use of wildfire is a core part of long-term strategies to create fire adapted communities, fire resilient landscapes, and effective response to wildfire incidents (USDA/USDI 2014). However, a major implementation challenge is defining and mapping the optimal mix of different long-term strategies to achieve risk management goals on dynamic, fire-prone landscapes that are fragmented by ownership, ecological conditions, and attitudes toward fire (Fig. 12 in Ager et al. 2016). For instance, establishing a common agreement among land managers, landowners, and collaborative landscape planning groups (Jakes et al. 2007, Butler et al. 2015) concerning the spatial allocation of fire management strategies is a complicated process. In particular, managing natural ignitions versus traditional fuel management using mechanical thinning and underburning poses many challenges from a risk governance standpoint. Traditional fuel treatment methods can change the behavior of large fires and facilitate suppression and containment (Kalies and Yocom Kent 2016), particularly in areas treated with prescribed fire (Finney et al. 2007, Moghaddas et al. 2010, Syphard et al. 2011*a*, Stephens et al. 2012). However, mechanical fuel treatment programs are expensive and are constrained by administrative, financial, and operational factors (North et al. 2015). The scale of existing programs and associated investments would need to be increased by several orders of magnitude to treat the backlog of forests that have undergone densification and fuel buildup due to fire exclusion practices (Haugo et al. 2015).

By contrast, managing wildfires for restoration in fire-adapted conifer forests can be inexpensive (depending on suppression tactics), effective, and ecologically beneficial, but can carry high uncertainty and risk for human safety. Policy analyses must

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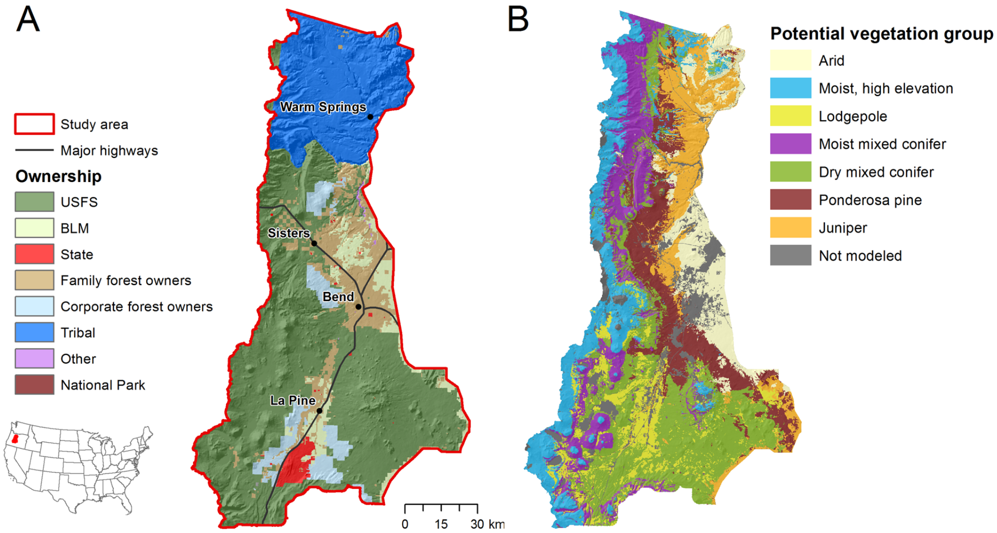
Fig. 1. Map of the study area showing (A) ownerships, towns, and major highways and (B) potential vegetation groups derived from Halofsky et al. (2014). BLM = Bureau of Land Management, USFS = U.S. Forest Service.



account for and mitigate the increased uncertainty and risks from fire events (Hill 2000, Wonkka et al. 2015, Hmielowski et al. 2016), which include unwanted aesthetics in amenity dependent communities, smoke production, and related health impacts. Numerous recent fires in the western U.S. have been managed in part or in whole for ecological benefit (henceforth "restoration fires"), including over 12 fires in 2017 totaling more than 60,000 ha of burned area (Forest Service Fire and Aviation Management, *personal communication*). Both modeling and empirical studies support these actions by showing that wild and prescribed fires can limit growth and lower severity of future fires and losses (Collins et al. 2009, Arkle et al. 2012, North et al. 2012, Houtman et al. 2013, Hoff et al. 2014, Parks et al. 2014, 2015*a*, Price et al. 2015) and facilitate suppression efforts (Moghaddas and Craggs 2007, Syphard et al. 2011*b*, Cochrane et al. 2012, Thompson et al. 2016). However, despite these and other studies, including landscape simulation research in which alternative fire management studies are simulated over time (Scheller and Mladenoff 2007, Spies et al. 2014), research is nonexistent on how landscape fire regimes and suppression budgets might change following long-term changes in fire policy. Thus the timing, pace, and scale of increased wildfire to actually achieve restoration goals are not known.

To address this gap, we use the agent-based Envision model (Bolte et al. 2004, Guzy et al. 2008) to examine the effects of increasing area burned on future fire regimes on a large 1.2 million ha multiowner landscape in the eastern Cascades of Oregon, USA. The area contains expansive areas of dry, fire adapted conifer forests that are the target of U.S. Forest Service restoration programs to improve fire resiliency and reduce wildfire impacts to local communities. We simulated four scenarios in which area burned was increased incrementally to mimic policies that leverage natural fires to reduce fuel loadings and restore presettlement fire regimes. We hypothesized that at some level of fire activity, fuels would begin limiting the growth of fires in subsequent years and reduce area burned and fire intensity because the consumption of fuels by wildfire would exceed accretion by forest growth and succession. We were specifically interested in evidence for tipping points (Adams 2013) and other discontinuities in fire feedbacks, and the overall leverage (Price et al. 2015) of fire to reduce future fire. We also hypothesized that increased fire would lead to impacts on ecological values including habitat for species dependent on specific types of old growth forest.

METHODS

Study area

The 1.2 million ha study area is located in central Oregon (Fig. 1) and includes public lands managed by the Deschutes National Forest (DNF), Bureau of Land Management (BLM), state of Oregon, National Park Service (NPS), and the Confederated Tribes of Warm Springs. Privately owned family lands and corporate timberlands, numerous small, private inholdings, and extensive wildland urban interface (WUI) are present on the east side of the study area (Table 1). The DNF is partitioned into about 30 different land management designations (e.g., general forest, scenic areas, recreation, wildlife, wilderness) according to the land and resource management plan (USDA FS 1990). Approximately 46% of the area within the DNF is in land designations that are available for forest and fuel management activities, with the unavailable lands located primarily within wilderness and recreational areas on the eastern edge of the forest.

The physiographic gradients, conifer forests (Fig. 1B), climate, and management resemble many of the western U.S. national forests and are described in detail elsewhere (Spies et al. 2014). Forest species include lodgepole pine (*Pinus contorta*), ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), and mountain hemlock (*Tsuga *mertensiana*). In general, the cooler, wet subalpine forests are located in the west, and semiarid juniper (*Juniperus occidentalis*) woodlands and arid shrublands to the east (Fig. 1B). About 24% of the study area contains dry mixed conifer forest, with lesser amounts of high elevation forest (15%), ponderosa pine (13%), and wet mixed conifer (13%) forest. Juniper and lodgepole pine forests combined cover about 18% of the study area, with the remaining lands consisting of arid shrub steppe and nonvegetated areas (17%). The area is noted for extensive contiguous stands of low-density ponderosa pine old growth that historically were maintained with periodic natural fire (Merschel et al. 2014). Much of the dry forest area receives extensive prescribed fire treatments by the DNF, although substantial areas with multilayer forests remain prone to uncharacteristically high intensity wildfire. The area has substantial fire activity, with on average 372 ignitions per year (1992-2013) that burn an average of 11,423 ha annually. Recent large fires include the B&B Complex in 2003 (36,733 ha), Pole Creek in 2012 (10,844 ha), and Sunnyside Turnoff in 2013 (21,448 ha).

Table 1. Ownership types and corresponding proportion of the study area used in the simulation experiment.

Ownership	Study area (%)
Federal	61
Tribal	21
Corporate forests	6
Family forests	4
State forests	2
Wildland urban interface (WUI)	7

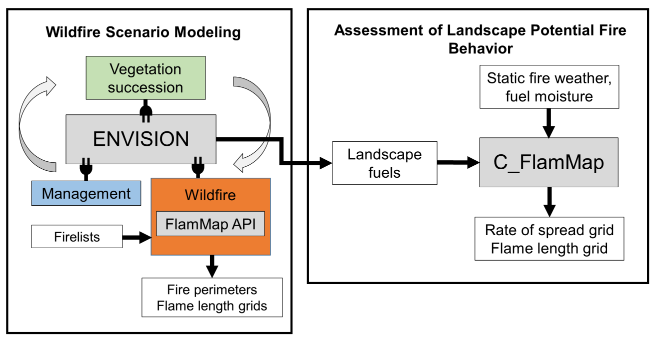
Envision overview

Envision is a landscape and agent-based modeling platform that simulates landscape change over time with a plug-in architecture allowing the incorporation of submodels for landscape processes, such as vegetation succession, forest management, and wildfire (Fig. 2). The model and its application have been described in a number of papers (Guzy et al. 2008, Hulse et al. 2009, 2016, Barros et al. 2017, Spies et al. 2017) and we only briefly described them with an emphasis on the wildfire submodel. For additional details concerning the development and testing of the wildfire submodel see Ager et al. (2017) and Ager, Barros, Day et al. (*unpublished manuscript*).

Vegetation succession

Vegetation succession within Envision is simulated with a state and transition submodel that classifies the landscape into a discrete set of vegetation states, each state having a set of deterministic and probabilistic transitions that describe the vegetation trajectory over time in response to succession, disturbance, and management. The states were attributed to spatially explicit individual decision units (IDU) with size ranging between one and eight ha and delineated according to vegetation and administrative boundaries (Spies et al. 2017). The states and transitions we implemented were originally developed as part of the Integrated Landscape Assessment Project (ILAP; Halofsky et al. 2014) and modified to represent specific forest management activities and wildfire effects. Each IDU was attributed with a vegetation class (henceforth vegclass) that represented a unique combination of potential vegetation type (PVT), tree size, forest canopy cover, and canopy layering. There were a total of 39 potential vegetation types (Appendix 1, Table A1.1). Tree size was represented by nine classes ranging from barren to stands of giant trees (> 76.2 cm dbh). Canopy cover was represented by five classes that ranged from none to high (> 60%), including a postdisturbance class. Forest canopy was classified as none, single, or multilayered. The structural stage attributes for describing a vegclass are further described in Appendix 1, Table A1.2 (see also Spies et al. 2017).

Fig. 2. Overview flow chart of wildfire scenario modeling showing the major "plugins" or submodels coupled to the Envision modeling framework. Wildfire events are simulated annually (left) using the FlamMap Application Programming Interface (API) as described in the text. Landscape potential fire behavior (flame length, spread rate) is calculated (right) using annual Envision outputs on landscape conditions that are processed through a command line version of FlamMap (C_FlamMap) as described in the text.



Each vegclass represented a vegetation state that can transition to a different state based on succession, management, and wildfire severity. State changes in vegetation were modeled as both probabilistic and deterministic transitions. Deterministic transitions were determined by an age threshold, meaning that once a specific age was reached, a stand would transition to a new successional state (Hemstrom et al. 2007). Probabilistic transitions reflected alternative succession pathways (e.g., a change in dominant species or development of seral species) and transition probabilities were determined by experts within the Forest Service and calibrated with the Forest Vegetation Simulator (FVS; Dixon 2002, Burscu et al. 2014). Transitions associated with management and wildfire were modeled as deterministic processes, through the management and wildfire submodels.

Forest management

Forest management activities were modeled in Envision based on extensive survey data and interviews with private landowners and public land managers, conducted as part of the Forests, People, Fire project (Spies et al. 2014). These management activities included various types of commercial and noncommercial harvesting, prescribed fire, and mowing and grinding (Appendix 2, Table A2.1). Management activities were simulated on the DNF and areas managed by the BLM following the Northwest Forest Plan and DNF land management plan (Appendix 2, Fig. A2.1).

Specifically, treatments were not allowed in areas deemed unsuitable for commercial timber production (low productivity, not operable), or removed from the scheduled timber program because of biodiversity conservation and amenity protection (e.g., wilderness, recreation areas, critical habitat, scenic areas). The remaining 61% of the area was available for treatment, henceforth, treatable area.

Management activities were allocated to IDUs based on preference scores that considered both biophysical and forest stand information (see Barros et al. 2017). Individual decision units were selected for treatment in decreasing order of preference until the annual area treatment target was met for each treatment activity. Modeling management activities required grouping treatment units (IDUs) into project areas to replicate the spatial grain of operational implementation on national forests. For this purpose the "expand" function was created in Envision to build treatment blocks that approximated the size and distribution of historical management activities.

In our simulations, we specifically modeled a scenario that called for treating a total of 8500 ha per year, representing an annual rate of 0.7% of the study area, or 2.3% of treatable land on the DNF. The annual treatment area was distributed among the treatment types as follows: 50% for mechanical thinning, 30% for mowing and grinding, and 20% for application of prescribed fire (some of which had previously been thinned). The effects of increasing management activity on wildfire dynamics are reported in Barros et al. (2017).

Wildfire

We created a wildfire simulation submodel within Envision (Fig. 2, left) by building an application programming interface to the FlamMap.DLL and Nodespread.DLL Dynamic Link Libraries developed at Alturas Solutions, Missoula, Montana (Brittain 2017). The resulting application programming interface (FlamMap API) shares the same code libraries as the FlamMap program (Finney 2006) and a number of wildfire decision support systems. The system has been extensively tested by the U.S. and international fire research community (Scott and Burgan 2005, Andrews 2007, Ager et al. 2011, 2014, Finney et al. 2011, Noonan-Wright et al. 2011, Kalabokidis et al. 2014, Salis et al. 2015, Oliveira et al. 2016). In-depth description of the fire model was beyond the scope of this journal and is described in detail in Ager, Barros, Day, et al., *unpublished manuscript*. A second command line program external to Envision (C_FlamMap, Fig. 2, right) was used to post process Envision landscape conditions and predict potential fire behavior for the entire study area (versus predicting and simulating discrete wildfire events). It was also built from FlamMap.DLL and Nodespread.DLL (Brittain 2017).

In each simulation year, Envision calls the wildfire submodel to prepare inputs by first translating the IDU conditions into surface and canopy fuels then writing a binary gridded (90 x 90 m) input file for the wildfire submodel. The conversion process uses a grid template that contains topographic variables that remain unaltered through the simulation. The wildfire submodel then reads information on the fire ignitions from a firelist generated from a spatiotemporal ignition prediction model, including fire weather conditions and burn period. The wildfire submodel is executed to simulate all fires predicted for the current simulation year, and the resulting fire perimeter and gridded flame lengths are written to files. The flame length grids are overlaid with IDU polygons and the average flame length for each affected polygon is calculated and used to interpret fire effects.

Daily wildfire probability, ignition location, cause (human or natural), and fire size were predicted by a spatiotemporal ignition prediction model (Preisler et al. 2004, Preisler and Ager 2013) described in detail in Ager et al. (2017) and Ager, Barros, Day, et al., *unpublished manuscript*. The model uses empirically derived relationships between energy release component (ERC) and historical fire size and ignition location data (11,618 ignitions between 1992-2009) obtained from the spatial wildfire database of the U.S. (Short 2014). Energy release component is an index in the national fire danger rating system (Bradshaw et al. 1984) used for fuel moisture. Previous studies have successfully used ERC to predict wildfire occurrence and size at continental scales (Finney et al. 2011). The fire prediction system used ignition and ERC data for a 3.32 million ha region within central Oregon (henceforth region) that encompassed the study area (Ager et al. 2017). Historical daily ERC data were downloaded from the Remote Automated Weather (RAWS) USA Climate Archive (WRCC 2014) for 25 remote stations within the region and included data from 1961-2011 depending on the station. Variability in ERC values among the stations within the study area was not sufficient to warrant separate fire prediction models for the areas around each station, hence we averaged ERC values over all stations by day of year.

Fuel moisture files for each fuel size class (1-hr, 10-hr, 100-hr, 1000-hr; Scott and Burgan 2005), as well as live herbaceous and woody components, were derived from historical (1987-2011) average fuel moisture values for each fuel class and for each value of ERC used in the simulations. Fuel moisture files were created prior to running the simulation and read by the wildfire submodel as each fire was simulated.

Wind direction was generated by randomly selecting from historical gust directions (1994-2011) from the Lave Butte RAWS station based on day-of-year of the predicted fire. Wind speed was based on gust values derived from the same weather data, but was restricted to days in the historical record in which area burned exceeded 500 ha to capture days in the historical record when fires actively spread. Wind gust speed was sampled from a gust speed probability distribution generated from analysis of the Lava Butte RAWS data.

The Nodespread.DLL fire spread algorithm requires burn period rather than fire size (Finney 2002), therefore fire size was translated from hectares to minutes by generating a fire size-burn period distribution using the wildfire submodel. Random ignition point locations (100) were simulated in the study area with burn periods ranging from 30 to 8000 min, with wind speed, azimuth, and ERC fixed at 18 mph, 220 degrees, and 60, respectively (Ager, Barros, Day, et al., *unpublished manuscript*). With these data, we derived a second-order polynomial linear regression model that was used to predict burn period for each fire as a function of modeled fire size (Ager, Barros, Day, et al., *unpublished manuscript*). Preliminary examination of the relationship showed calibration procedures were needed to replicate historical fire size distributions. Specifically, if the fire size was under predicted in Envision because ignitions landed in nonburnable areas, a fire size adjustment was made to randomly relocate ignitions within a 5 km radius of the original location. This was done up to five times for any ignition that did not reach 80% of its predicted size. The XY location of the fire that best matched the predicted size was recorded in the output firelist. When the simulated fire size in Envision was overestimated (greater than 1.5 times the predicted size) the burn period was reduced proportionally to the difference between the predicted and simulated fire size (Ager, Barros, Day, et al., *unpublished manuscript*).

The spatiotemporal ignition prediction model was written in R (R Core Team 2014) and executed prior to an Envision simulation to generate firelist text files that predicted daily fire occurrence and size from the model. Additional parameters associated with each ignition in the firelist are day-of-year, ERC, fire weather parameters (wind speed and azimuth), burn probability, burn period, fire cause (natural or human), ignition location (XY coordinate), and fuel moisture conditions. Note that the spatiotemporal ignition prediction model generated a stream of ERCs based on an autoregressive model of historical ERCs, and thus each execution resulted in a unique firelist and 50-year fire simulation.

Fire effects

Fire effects were modeled using gridded flame length outputs for each fire perimeter generated by the wildfire submodel. Flame length is often used as a proxy for describing fire intensity in the field (NWCG 2013). A fire effects lookup table translated flame lengths into disturbance types for each vegclass affected by fire. Flame length was translated into three fire disturbances: (1) low-intensity fires that do not cause enough tree mortality to change the vegclass, but reduce fuels accumulation; (2) mixed severity fires that may change the vegclass through the mortality of smaller trees and/or less fire resistant species; and (3) stand-replacing fires that kill all trees, returning the IDU to either a grass-forb state, or when sprouting species are present, to a young state. The translation of flame length to fire severity relied on FVS (Reinhardt and Crookston 2003). In this process, we used the approach of Ager et al. (2010) in which representative tree lists for each vegclass were exposed to simulated fires of increasing flame length in 0.33 m intervals and the flame length interval that resulted in more than 20% and less than 80% tree mortality in the stand was used to establish the lower and upper flame length thresholds for mixed severity fire. Flame lengths above and below the mixed severity threshold were then used to classify fires as stand-replacing and low severity, respectively.

Fuel dynamics pre- and postdisturbance

Surface and canopy fuels consisted of the standard five fuel variables used by FlamMap5 and related wildfire simulation models (Finney 2006, Ager et al. 2011, Finney et al. 2011). Surface fuels were represented by the fuel models of Scott and Burgan (2005). Surface fuel models for the IDUs in the DNF were assigned based on the majority representation in the forest's fuel model layer. Outside the forest, we used the LANDFIRE 2008 rapid refresh FBFM40 layer (LANDFIRE 2013). Canopy fuels were described by canopy bulk density, canopy cover, canopy base height, and total stand height, and determined using the average value of each variable for representative stands for each vegclass with the Fire and Fuels Extension (FFE) of FVS.

Changes in fuel structure that were not accompanied by changes in vegclass (e.g., tree size or canopy) were accomplished by assigning fuel-model variants based on disturbance type (Appendix 2, Table A2.2). The fuel model variant remained unchanged until a set number of years passed (time-in-variant) or a vegclass deterministic or probabilistic transition occurred. The impact of management activities on vegclasses was estimated based on expert opinion (Platt 2014, Kline et al. 2017) and stand modeling with FVS. For example, tree removal (i.e., thinning, clear-cut) triggered transitions to vegclasses that reflected lower tree density, larger tree size, lower canopy closure, and reduced canopy layers, depending on the intensity of the management activity (Appendix 2, Table A2.1). For management actions that resulted in changes in surface fuel attributes only (e.g., fuels mastication, prescribed fire, and surface fire), the vegclass remained unchanged.

Simulations

We used the wildfire submodel within Envision to simulate 50-year scenarios where wildfire activity was first simulated at contemporary levels (1992-2009), and then incrementally increased while maintaining forest and fuel management at current levels (Barros et al. 2017). The four increased levels of wildfire activity were achieved by multiplying the burn period of each ignition by 2X, 3X, 4X, and 10X. For each of the five scenarios we simulated 15 replicates and varied only the burn period. A number of other fire simulation parameters could have been manipulated to achieve an increase in area burned (fuel moisture, wind speed) although changes in these parameters could potentially also increase fire intensity as well.

Fire feedbacks

We used the simulated fire perimeter data to examine the self-limiting properties of wildfires over time. We analyzed total area burned per year by fitting a generalized additive model (GAM; Wood 2011) using seven explanatory variables: ERC, wind speed, burn period, the cumulative area burned in prior years, year of simulation (1-50), average flame length, and wildfire scenario (1X, 2X, 3X, 4X, 10X). We considered four alternative time lags for cumulative area burned in prior years: 1-5 years, 6-10 years, 11-20 years, and more than 20 years. The model was run using the Mixed GAM Computation Vehicle (mgcv) package (Wood 2006) in R (R Core Team 2014). The resulting model was used to estimate the fractional change in area burned in a given simulation year in response to prior area burned, expressed as a proportion of the study area. We also examined how the effect of prior area burned varied over time. We estimated the combined effects of year and cumulative area burned on area burned in a given year, and created smoothed contour plots. Outputs from the statistical modeling were also used to estimate the wildfire leverage, meaning the unit reduction in area burned in a given year resulting from one unit of antecedent area burned. The concept of leverage has been used to quantify the effect of prescribed fire on subsequent unplanned fires (Price et al. 2012, 2015).

To examine changes in fire intensity and spread rate for the entire study area, we processed landscapes generated by Envision with C_FlamMap to calculate potential flame length and spread rates under static fuel moisture and weather (Fig. 2, right). C_FlamMap (Brittain 2017) is a command line version of

Table 2. Wildfire response metrics per wildfire scenario over 50 simulated years. Increased levels of wildfire activity in each scenario were achieved by multiplying the burn period of each ignition over contemporary levels by the multiplier indicated.

	Wildfire scenario				
Wildfire response metric	1X	2X	3X	4X	10X
Total area burned (ha)	3,350,622	8,702,019	13,650,813	18,170,496	38,509,509
Total area burned per replicate (ha)	223,375	580,135	910,054	1,211,366	2,567,301
Average annual area burned per replicate (ha)	4467	11,603	18,201	24,227	51,346
Average fire size (ha)	325	847	1333	1782	3825
Simulated increase in area burned over contemporary (%)	0	253	407	542	1150
Study area burned per replicate year^{-1} (%)	0.4	1.0	1.6	2.1	4.4
FRI (year)[†]	250	100	63	48	23
Leverage[‡]	0.01	0.6	0.15	0.21	0.43

[†]FRI = fire rotation interval, the time required to burn the entire study area landscape.
[‡]Leverage corresponds to the slope of a linear regression where the independent variable is annual area burned and the predictor is cumulative area burned in the previous five years.

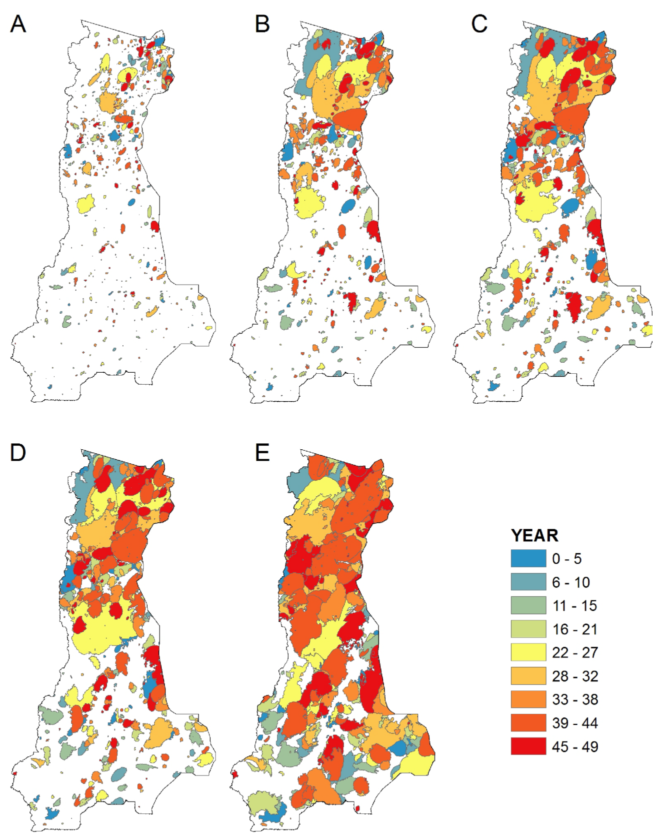
FlamMap5 and calculates independent potential fire behavior for each pixel assuming static weather conditions. We processed 3750 simulations (15 replicates x 5 wildfire scenarios x 50 years) with C_FlamMap to generate flame length (m) and spread rate (m min^{-1}) grids at 90 m resolution. We used 97th percentile weather conditions to represent extreme weather conditions consistent with large fire growth in the area (Ager et al. 2007). The outputs were used to calculate and plot average flame length and average rate of spread over the entire forested area (and 15 replicates), per year and fire scenario.

Finally, to understand how increasing levels of fire affected fire sensitive components of biological diversity, we assessed changes in high suitability habitat for the northern spotted owl (NSO). The NSO is listed under the *Endangered Species Act* in the Pacific Northwest and large reserves in the study area are dedicated to maintaining and growing habitat that consists of dense, multilayered, older mixed-conifer forests. The habitat model was based on vegetation type, canopy cover, and tree size characteristics and was developed specifically for central Oregon using owl occurrence data as reported by Spies et al. (2017).

RESULTS

Analysis of area burned among the wildfire scenarios showed that increasing the wildfire activity from 1X to 10X resulted in an increase of average annual area burned from 4467 (0.4% of study area) to 51,436 ha (4.4% of study area; Table 2). Note that because of the nonlinear behavior of fire spread, the increase in burned area relative to contemporary wildfire is not proportional to the burn period multipliers (Appendix 3, Fig. A3.2), e.g., increasing the burn period by a factor of 4 led to an increase in area burned by a factor of 5.4 (Table 2). In the contemporary scenario, the fire rotation interval (time to burn the study area) was 250 years versus 23 years for the 10X scenario (Table 2). The effect of increasing burn period was especially apparent in the arid, juniper woodland and dry forest areas compared to moist mixed conifer and subalpine forests (Fig.1). The 10X wildfire scenario resulted in fire perimeters that eventually covered almost the entire study area (Fig. 3E). Annual variability in area burned was greater for scenarios with higher levels of simulated fire (e.g., 4X and 10X) and mean fire size increased substantially from an average of 325 ha to 3732 ha for the 1X and 10X scenarios, respectively (Table 2; Fig. 4).

Fig. 3. Simulated fire perimeters in the study area from one 50-year Envision simulation for five wildfire scenarios: 1X (A), 2X (B), 3X (C), 4X (D), and 10X (E). Increased levels of wildfire activity in each scenario were achieved by multiplying the burn period of each ignition over contemporary levels by the multiplier indicated.



Specific instances in which fire growth was limited by prior fires during the simulation were readily apparent in the simulation outputs. For the purpose of illustrating fire interactions, we identified two fire perimeters in an Envision simulation in which

Fig. 4. Mean fire size by year of simulation for 15 replicate runs for each wildfire scenario (1X, 2X, 3X, 4X, 10X, and see Table 2 for total area burned per year in each scenario). Increased levels of wildfire activity in each scenario were achieved by multiplying the burn period of each ignition over contemporary levels by the multiplier indicated.

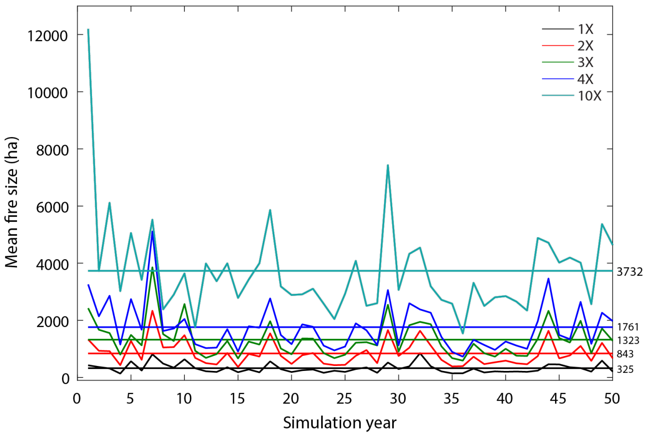
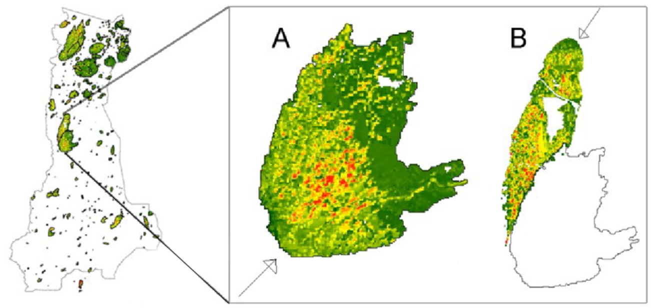


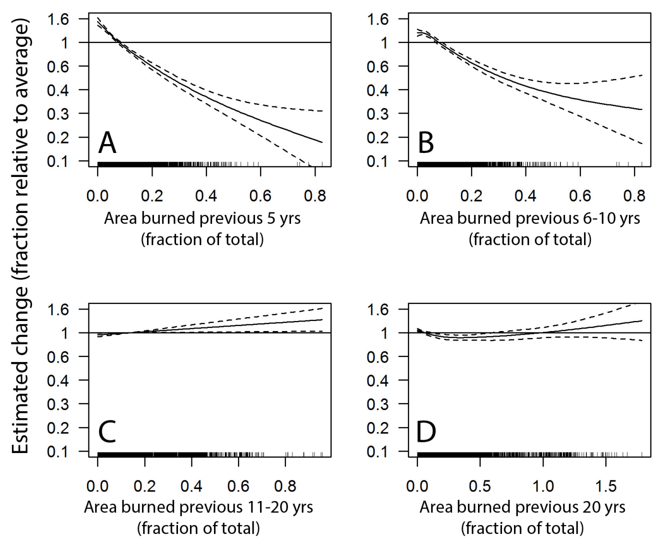
Fig. 5. Example of fire feedbacks in the study area in which a simulated fire (A) in simulation year 25 limited the growth of a subsequent fire in year 28 (B). Pixels represent flame length from a low of 1 (green) to high of 54 m (red). Arrows indicate direction of fire spread. Fire in A was 14,657 ha and simulated at ERC = 80 and wind speed = 18 mph. Fire in B was 8609 ha at ERC = 81 and wind speed = 16 mph. Fire B without A burned 12,286 ha.



a simulated fire encountered the perimeter of a fire that had burned the prior year (Fig. 5). These two ignitions were separated by three years and the first fire reduced the area of the second fire by 30%. In the 50-year simulation, the cumulative effect of these interactions over space and time showed that the effect of prior area burned on wildfire size in the current simulation year depended on the time window between the fires (Fig. 6). Fires burning in the past five years reduced area burned in the current simulation year up to 80% under the extreme case when 60% of the study area was burned (Fig. 6A). Fires burning in the past 10 years reduced the area burned in a given simulation year a maximum of 35%. Diminishing returns in terms of the effect of prior area burned on subsequent fires in a given year were observed when about 35-45% of the landscape was burned in the previous 10 years (Fig. 6A, B). Wildfires more than 10 years prior had no discernible effect on the area burned in a given simulation year (Fig. 6C, D). Over all the scenarios, we estimated the average reduction (or leverage; Price et al. 2012, 2015) of area burned in the prior 5 years on area burned in a given simulation year was about 0.18 (i.e., 18% reduction per unit area burned), and varied from 0.01 for the 1X scenario to 0.4 for the 10X.

The effect of prior area burned on subsequent area burned varied over time and generally followed the annual rate at which wildfires burned the study area (Fig. 7). For example, when 50% of the study area was burned over the first 10 years of the simulation (5% per year, 20 year fire rotation), about 0.7% of the study area was predicted to burn in any subsequent year. When the same area was burned over the prior 50 years (1% per year or fire rotation of 100 years), the rate of burning in year 50 was estimated between 2.2% and 2.7%, or about 3 times higher (Fig. 7). We observed similar results for rates of spread confirming that observed effects on burned area were caused by modification of fuels from prior area burned (Appendix 3, Fig. A3.1). Specific thresholds for discontinuities in fire feedbacks were not observed,

Fig. 6. Change in average area burned in any given year as a function of area burned in the previous 5 years (A), 6 to 10 years (B), 11 to 20 years (C), and more than 20 years (D).

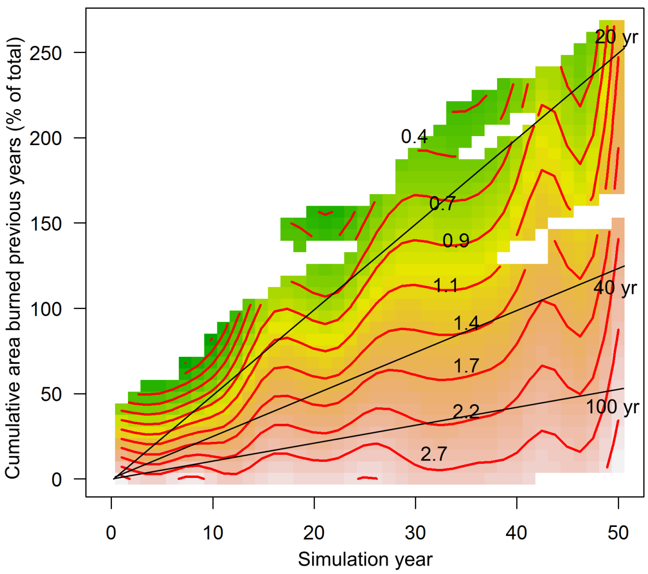


but the analysis did reveal cyclical trends in fire feedbacks with increasing area burned. The cycles exhibited a periodicity of about 10 years (Fig. 7), corresponding to the regrowth of fuels as determined by the vegetation succession submodel.

Although contours of the estimated effect (% burning in current year, shown in Fig. 7) more or less paralleled the rate of burning (or fire rotation interval, Fig. 7; Table 2), there was a general tendency for the effect to diminish over time, meaning a higher rate of burning was required to achieve the same effect in later versus earlier simulation years. For instance, at a 2.5% rate of annual burning (fire rotation interval of 40 years) after 20 years

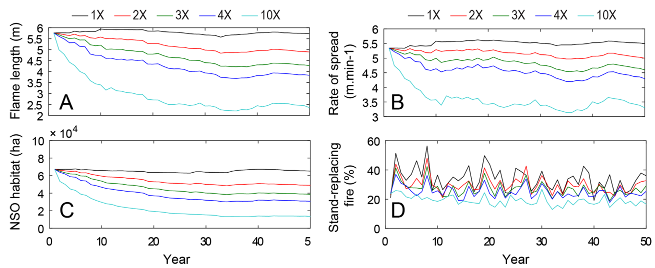
(Fig. 7, X = year 20 , Y = 50%), the estimated rate of burning for subsequent years is estimated at ca. 1.7% per year. At the same rate of burning over a longer time frame (Fig. 7, X = year 40, Y = 100%), the estimated rate of burning for subsequent years is about 1.4% or a 18% decrease in the annual area burned. A decreasing effect of prior burned area on the area burned in a given simulation year is a positive feedback of fire on fire, and although this longer-term effect was minor compared to the short-term effects of prior burned area (Fig. 5A), it was observed for a wide range of fire rotation intervals in the simulation outputs.

Fig. 7. Contour plot of expected area burned in the current year (% total) conditional on simulation year and cumulative area burned in all previous years. Black lines show reference fire rotation intervals, or number of years to burn an area equal to the entire study area. The average area burned per year for all simulations was 2.2% or a fire rotation of about 45 years. Contours show a 10-year periodicity that corresponds to the modeled vegetation regrowth after fires.



Modeled outputs of potential fire behavior for the study area as a whole obtained from C_FlamMap simulations showed how fire intensity and spread rate changed with increasing levels of fire activity (Fig. 8A, B). As noted earlier, these results represent potential fire behavior if the entire study area burned under constant weather conditions, versus simulated fire events within Envision. Trends in potential fire behavior over time measure broad-scale changes in surface and canopy fuels. Substantial reductions in potential flame length were observed as fire activity was increased (Fig. 8A). For instance, average flame length decreased in the 10X scenario from 6 m in year 1 to 2.3 m in year 50 (62% reduction) with most of the change occurring in the first 10 years of simulation. A slightly upward trend in flame length was observed over time for the contemporary 1X fire scenario in the first 10 years of the simulation (Fig. 8A). Fire intensity decreased slightly for the 2X scenario and more sharply for the higher levels of burning suggesting that reducing the upward trend in the contemporary scenario would require at least twice the current area burned. Changes in potential rate of spread over time mirrored flame length although the effect of increasing fire was less pronounced. For the 10X scenario, rate of spread was reduced from 5.4 m min^{-1} to 3.2 m min^{-1} at the end of the simulation time (40% reduction over 50 years) with most of the reduction observed in the first 10 years of the simulation (Fig. 8B).

Fig. 8. Trend in (A) potential flame length, (B) potential rate of spread, (C) northern spotted owl habitat (NSO), and (D) stand-replacing fire by simulation year and wildfire scenario (1X, 2X, 3X, 4X, and 10X). Increased levels of wildfire activity in each scenario were achieved by multiplying the burn period of each ignition over contemporary levels by the multiplier indicated.



Northern spotted owl habitat (NSO) declined slightly (2000 ha, 3%) under the 1X fire scenario (Fig. 8C). Substantial reduction was observed for the 10X scenario in which NSO habitat was reduced by 36,474 ha (80.13%). Intermediate fire scenarios also resulted in substantial decline in habitat area, and for all scenarios much of the loss occurred in the first 20 years. In the 10X wildfire scenario, most of the decline occurred in the first 10 years of the simulation. The decline was spread more widely over the simulation time in the other fire scenarios.

Fire severity among the scenarios was measured as the percentage stand replacing fire in forested areas (i.e., high severity), versus mixed and low severity (Fig. 8D). There was a slight trend over time toward less stand-replacing fire in all the scenarios. However, the differences among scenarios were larger than the downward trend over time. Stand replacing fire relative to total area burned averaged 34% for the 1X scenario, compared to 19% for the 10X or a 44% reduction.

DISCUSSION

Our study examined the effects of increased fire on future fire regimes and forest structure on a 1.2 million ha landscape in central Oregon. We examined whether contemporary (prior 20 years) fire regimes are stationary and what magnitude of burning would be required to observe fire feedbacks under current forest management programs. Negative fire feedbacks were defined as the reduction in area burned by wildfire events in a given year resulting from encounters with prior wildfire perimeters. We found that feedbacks paralleled the rate of burning during the 50-year simulation and that increasing wildfire reduced landscape flammability as measured by spread rate, flame length, and area burned. On average, we found that area burned in a given simulation year was reduced by about 18% per unit area burned within the prior five years averaged across all scenarios. At the highest fire treatment (10X burn period, 11.5X area burned), the study area burned with a fire rotation interval of 23 years, versus 250 years under the simulated contemporary rate of burning.

Averaged over the different vegetation types, area burned more than 10 years prior did not significantly affect the area burned in a given simulation, a result that is within the range of longevity (2 to 23 years) reported in a recent review (Prichard et al. 2017). Negative fire feedbacks were slightly moderated by longer-term positive feedbacks, in which the effect of prior area burned diminished during the simulation.

The broader interpretation of the results is in part conditional on the plausibility of the accelerated fire scenarios, which could materialize by multiple pathways including: (1) a warming climate (McKenzie et al. 2004, McKenzie and Littell 2017); (2) a change in suppression practices (USDA/USDI 2009); and (3) increased human ignitions. McKenzie et al. (2004) used statistical relationships between climate and wildfire activity to estimate that a moderate warming scenario could result in a two to fivefold increase in annual area burned in the western U.S. However, this and related studies assumed fuels would be available at future rates of burning and thus fuel-mediated fire feedbacks were not factored into the estimates (see McKenzie and Littell 2017). Although climate-fire statistical studies have provided evidence that fire regimes in and around the study area are strongly driven by climate (not fuel limited), our study suggests that under a warmer climate, higher rates of burning would be moderated by negative fire feedbacks. We also showed that future fire regimes would be characterized by predominately low intensity wildfire (Fig. 8A). Achieving our wildfire scenarios through changes in current suppression practices is a social and operational question that is difficult to assess. Current federal wildfire policy provides for many options to respond to wildfire ranging from full suppression to passive monitoring (USDA-USDI 2009), but risk considerations, including potential impacts to socioeconomic values located on national forests have generally lead to a full suppression response on the vast majority of wildfires in this region. The potential for human ignitions to accelerate wildfire activity in the study area deserves further consideration because they account for about half of the historical area burned in the past 20 years. Human ignitions have effectively extended the fire season into the spring and fall, and probably contribute more to area burned by low severity fire than natural ignitions (Fig. 4 in Ager et al. 2017). Our Envision model does include projections of population growth within the study area, but we did not model how this change affected ignitions as done elsewhere (Prestemon et al. 2016).

A number of empirical studies have shown reduced wildfire spread and severity in recently burned areas (Teske et al. 2012, Haire et al. 2013, Prichard and Kennedy 2014, Parks et al. 2015*b*, Holsinger et al. 2016). Despite these and other studies of fire on fire feedbacks (reviewed in Prichard et al. 2017), a mechanistic typology to disentangle underlying processes that generate both positive and negative feedbacks could help organize existing knowledge and the design of future studies. For instance, Pritchard et al. (2017) tabulated four metrics that describe potential outcomes from fire-on-fire interactions that could be further expanded by considering spatial mechanisms by which these effects are manifested. Specifically, ignitions can occur in recently burned areas and fail to spread, thus changing patterns of fire occurrence (Parks et al. 2015*c*). Ignitions in recently burned areas with decreased fuel loadings can result in fires with lower spread rates and intensity (Safford et al. 2009, Prichard et al. 2010). Finally, fires can ignite in unburned areas and spread to burned areas (Finney et al. 2005). Previous studies have not been able to distinguish the relative contributions of these different spatial interactions because of lack of data, small sample sizes, or a combination of the two. In general empirical data sets have insufficient information to understand and analyze the simultaneous effects of past fires on spread rate, intensity, and burned area over long periods of time, hence the utility of a simulation framework. As an example, we observed short-term (< 10 years) negative feedbacks stemmed from fires encountering recently burned area (Fig. 3E, Fig. 5) whereas longer-term positive feedbacks resulted from fire-induced accelerated succession in which the landscape changed from relatively stable mature vegetation states to younger successional stages that have rapid transitions to increasingly flammable conditions. Positive fire feedbacks are not reported for temperate forests in the review by Pritchard et al. (2017). Our future work will include analyses to determine the relative effects of the different mechanisms that contribute to fire feedbacks, both positive and negative, observed within the study area.

We examined the ecological impact of our fire scenarios using habitat for northern spotted owl (NSO) and area burned by high severity fire. Large reductions in NSO habitat were predicted by the model under scenarios with accelerated wildfire (Fig. 8C). Habitat for the NSO is sensitive to fire, owing to the fact that requirements for high levels of canopy closure and multistory conditions translate to high crown fuel loadings and ladder fuels, and thus the potential for high severity fire. Prior simulation studies suggest that even low intensity fire can modify stand structure in NSO habitat to make it unsuitable (Ager et al. 2007, Kerns and Ager 2007). In the companion study of Spies et al. (2017), wildfire was the major driver of habitat loss although contemporary fire levels slightly increased habitat over time as observed in our study. Dense older forests also provide habitat for other species of interest including the northern goshawk (*Accipter gentilis*) and Pacific marten (*Martes caurina*) (Spies et al. 2017). Northern spotted owls are a federally listed species (ESA 1973), and as pointed out by Spies et al. (2017), their habitat preservation is an ecological and social driver of federal forest management at the expense of other ecological and socioeconomic values (e.g., open old growth forests that are more resilient to fire and drought). In this study, a clear trade-off existed between increased fire activity to restore fire resilient forests and the conservation of NSO habitat. For instance, U.S. Forest Service managers are increasingly focused on managing for historical disturbance regimes and ranges of variability (Haugo et al. 2015). Although areas of dense multilayered forests were preserved under historical fire regimes because of their topographic positions and climate (Camp et al. 1997), accelerated wildfire scenarios we simulated burned through these areas. Refining the simulations to increase the frequency of lower severity fire specifically could lead to the development of forest structure and composition that is resilient to fire and climate change (Hessburg et al. 2016). Although Davis et al. (2016) found that NSO habitat in the eastern Cascades of Oregon actually increased by 13% between 1993 and 2012, a single large fire within the study area could reverse this trend. Spies et al. (2017) found that NSO nesting habitat declined under management scenarios compared to those without management, although contemporary wildfire levels resulted in higher habitat loss than did management.

The study complements previous forest and fire modeling studies, although comparisons are difficult. Barros et al. (2017) examined increasing levels of fuel management and found that, compared to no action, current forest management policy on federal lands led to reductions in area burned up to 25% over a period of 50 years. They also found that tripling the current amount of area treated would reduce burned area under simulated conditions up to 40%, and the likelihood of a fire > 10,000 ha by threefold. The "leverage" (Price et al. 2015) we found of prior area burned on current year fire activity was less than typically reported for fuel treatments (Finney et al. 2007). We suspect that leverage from fuel management is more efficient because fuel treatments are dispersed and provide a higher chance of encountering a subsequent fire compared to a single fire footprint. Empirical studies have shown that management policies allowing fires to burn in wilderness areas or use of prescribed fire have eventually led to self-limiting fire in specific fire-on-fire events (Finney et al. 2005, Boer et al. 2009, Price et al. 2015). Other simulation studies (e.g., Loudermilk et al. 2014) did not report landscape fire feedbacks, presumably because fires rarely intersected prior fire perimeters. Price et al. (2012) found zero leverage in southern California in predominantly grassland-shrubland systems likely due to low encounter rates (intersection of wildfire with previously burned areas), and noted higher leverage in forests and savannas in Australia (0.1 and 0.3, respectively). In a subsequent global analysis (Price et al. 2015), the highest leverage was documented in Portugal (0.9). This high rate was partially explained by likely spatial bias, in which the complex, fragmented landscape constrains fire spread and leads to regular reburning.

Exploring future trade-offs associated with a fire management strategy that relies on both restoration fires and mechanical fuel management requires a robust landscape and management simulation model. Specifically, national forests could benefit from mapping landscape-scale synergies between mechanical forest fuel treatments and restoration wildfires to meet ecologic, socioeconomic, and fire resiliency goals of federal forest restoration programs. Optimal investment levels likely exist for respective management emphases on mechanical fuel management, prescribed fire, and restoration wildfires. For instance, fuel management programs can be used to build low hazard wildfire containers in which fires can burn at low intensity and be contained with low-cost suppression activities. Much of the landscape fuel management research has explored optimal fuel management strategies to specifically reduce fire spread (Loehle 1999, Finney 2007, Lehmkuhl et al. 2007, Parisien et al. 2007, Konoshima et al. 2008, Wei et al. 2008, Kim et al. 2009), versus creating fire adapted (e.g., low hazard) landscapes that can be maintained as resilient landscapes with periodic fire. However, treating landscapes to restore natural fire is appropriate in fire adapted forests, whereas fuel break strategies aimed at fire exclusion and protecting fire sensitive values (e.g., WUI or habitat for dense forest species) serve a purpose in high severity fire regimes. Current fuel management projects on western U.S. national forests are difficult to interpret with respect to long-term fire management goals (exclusion versus acceleration) most likely because they are motivated by wide ranging objectives including economic values, fire ecology, current wildfire exposure, and stakeholder involvement in the planning process (Butler et al. 2015, Kalies and Yocom Kent 2016).

Ultimately the development of policies to address socioeconomic and ecological losses from large-scale natural disturbances is a complex problem that requires the integration of both social and biophysical risk systems (Corotis and Hammel 2010, Fuchs et al. 2011). Reducing the area of high-severity fire through fuel management or wildfire to create fire resilient forests comes with a number of trade-offs in terms of addressing the production of other ecosystem services and socioeconomic demands from national forests. Fire suppression costs, which are currently over 50% of the USDA Forest Service annual budget (USDA FS 2015*b*), can potentially be reduced by restorative fire, however, water quality, wildlife, recreation, and visual amenities will be affected in the short run if natural ignitions are allowed to burn as natural fires develop fire resilient forests. Fire interactions are highly uncertain in space and time, and thus scheduling wildfire as a way to treat fuels is a complicated approach compared to fuel treatments, especially within highly fragmented landscapes with respect to ownership, development, and disparate land management objectives (Charnley et al. 2017). In addition, increased smoke production from restoration wildfires has serious health implications and degrades amenity values in rural communities that depend on them for economic sustainability (Liu et al. 2015, Schweizer and Cisneros 2017). An equally large challenge will be managing the social and political risk facing fire managers if fires escape and result in economic losses from either restoration wildfires or prescribed fire (Hill 2000, Ryan et al. 2013). Despite improvements in the technology and tools to predict the spread of wildfires during an active fire incident (Noonan-Wright et al. 2011), high uncertainty and risk for managers during wildfire events will remain a barrier to using wildfires to manage fuels, especially near the urban interface and other high risk areas.

Our future work with Envision will use detailed decision criteria to select fires based on seasonality and location of ignitions (Figs. 3-5 in Ager et al. 2017) and thus more accurately represent operational practices aimed at increasing the area burned by restoration wildfires. By amplifying specific wildfire events, socioeconomic losses and suppression costs can be minimized while reducing fuels in key areas that can spawn future high severity fires. In a broader context, the agent-based Envision policy modeling system can also be used to investigate many other social and biophysical aspects of wildfires, and contribute to disentangling the potential effects of climate, succession, and management on future fire regimes. Future research with the model can potentially provide insights into the temporal scale mismatches (Cumming et al. 2006) between short- and long-term wildfire risk management that contribute to fragmented wildfire risk governance systems (Steelman 2016).

Responses to this article can be read online at:
http://www.ecologyandsociety.org/issues/responses.php/9680

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Appendix 1. Vegetation Succession.

Table A1.1. Potential vegetation type (PVT) descriptions and management group by each of the Vegetation Dynamics Development Tool (VDDT) modeling regions in the study area.

VDDT modeling region	PVT description	PVT management group
Oregon Blue Mountains	Grand fir - cool, moist	Moist mixed conifer
Oregon Blue Mountains	Subalpine fir - cold, dry	Moist, high elevation, other
Oregon Blue Mountains	Subalpine woodland	Moist, high elevation, other
Oregon Blue Mountains	Ponderosa pine - dry, with juniper	Ponderosa pine
Oregon Blue Mountains	Ponderosa pine - xeric	Ponderosa pine
Oregon Blue Mountains	Mountain hemlock - cold, dry	Moist, high elevation, other
Oregon Blue Mountains	Mixed conifer - cold, dry	Dry mixed conifer
Southeast Oregon	Mixed conifer - cold, dry	Dry mixed conifer
Southeast Oregon	Mixed conifer - dry (pumice soils)	Dry mixed conifer
Southeast Oregon	Ponderosa pine - dry (residual soils)	Ponderosa pine
Southeast Oregon	Ponderosa pine - xeric	Ponderosa pine
Oregon East Cascades	Western hemlock - wet	Moist, high elevation, other
Oregon East Cascades	Western hemlock - intermediate	Moist, high elevation, other
Oregon East Cascades	Western hemlock - cold	Moist, high elevation, other
Oregon East Cascades	Pacific silver fir - warm	Moist, high elevation, other
Oregon East Cascades	Pacific silver fir - intermediate	Moist, high elevation, other
Oregon East Cascades	Mountain hemlock - intermediate	Moist, high elevation, other
Oregon East Cascades	Mixed conifer - moist	Moist mixed conifer
Oregon East Cascades	Oregon white oak / Ponderosa pine	Ponderosa pine
Oregon East Cascades	Subalpine parkland	Moist, high elevation, other
Oregon East Cascades	Shasta red fir - dry	Moist, high elevation, other
Oregon East Cascades	Mixed conifer - dry (pumice soils)	Dry mixed conifer
Oregon East Cascades	Lodgepole pine - wet	Lodgepole
Oregon East Cascades	Lodgepole pine - dry	Lodgepole
Oregon East Cascades	Ponderosa pine - dry (residual soils)	Ponderosa pine
Oregon East Cascades	Mixed conifer - dry	Dry mixed conifer
Oregon East Cascades	Mixed conifer - cold, dry	Dry mixed conifer
Oregon East Cascades	Mountain hemlock	Moist, high elevation, other
Oregon East Cascades	Ponderosa pine - xeric	Ponderosa pine
Oregon East Cascades	Ponderosa pine - Lodgepole pine	Dry mixed conifer

Table A1.1. Contd.

VDDT modeling region	PVT description	PVT management group
Southwest Oregon	Subalpine parkland	Moist, high elevation, other
Southwest Oregon	Mountain hemlock - cold, dry	Moist, high elevation, other
Southwest Oregon	Shasta red fir - moist	Moist, high elevation, other
Southwest Oregon	White fir - cool	Moist mixed conifer
Southwest Oregon	White fir - intermediate	Dry mixed conifer
Southwest Oregon	Douglas-fir - moist	Dry mixed conifer
Southwest Oregon	Douglas-fir - dry	Dry mixed conifer
Southwest Oregon	Oregon white oak	Moist, high elevation, other
Southwest Oregon	Ponderosa pine - dry, with juniper	Ponderosa pine

Table A1.2. Structural stage attributes assigned to each of 39 potential vegetation types (PVT, Table A1.1) to define 565 unique vegclasses, thus each vegclass includes a combination of PVT, tree size, canopy cover and layering.

Structural stage attribute	Class
Size (dbh)	Barren
	Meadow
	Shrubs
	Seedling/sapling
	Pole (0.13-0.25 m)
	Small tree (0.25-0.38 m)
	Medium tree (0.38-0.51 m)
	Large tree (0.51-0.76 m)
	Giant tree (>0.76 m)
Canopy cover	Low (open, 10-40%)
	Medium (40-60%)
	High (closed, >60%)
	Post-disturbance
Layering	None
	Single
	Multi

Appendix 2. Management, Wildfire and Fuels.

Table A2.1. Effect of fire severity and management action on tree size, canopy cover, canopy layering and surface fuels.

Fire severity/management activity	Effect of disturbance
Surface fire (includes prescribed fire)	Reduces surface fuels; reduces multi-layer states to a single layer for some vegetation states
Mixed-severity fire	Reduces surface fuels; reduces multi-layer states to a single layer; decreases canopy cover by one or two classes
Stand-replacing fire	Reduces surface fuels and no canopy layers remain; decreases canopy cover to none or low; trees are killed with transition to grass-forb or shrub vegetation states
Mowing and grinding	Eliminates shrub layers and increases surface fuels
Pre-commercial thinning	Increases surface fuels; decreases high canopy cover to moderate or low cover
Thin from below	Increases surface fuels; generally reduces multi-layer states to single layer; decreases high canopy cover to moderate
Partial harvest	Increases surface fuels; generally reduces multi-layer states to single layer; generally decreases canopy cover by one class in high and moderate states
Partial harvest – heavy	Increases surface fuels; reduces multi-layer states to single layer; decreases canopy cover by one or two classes; reduces tree size by one class
Regeneration harvest	Increases surface fuels and no canopy layers remain; decreases canopy cover to none or low; trees are removed with transition to grass-forb or shrub vegetation states
Post-fire salvage of dead trees	No effect in canopy cover or layering. Increases surface fuels.

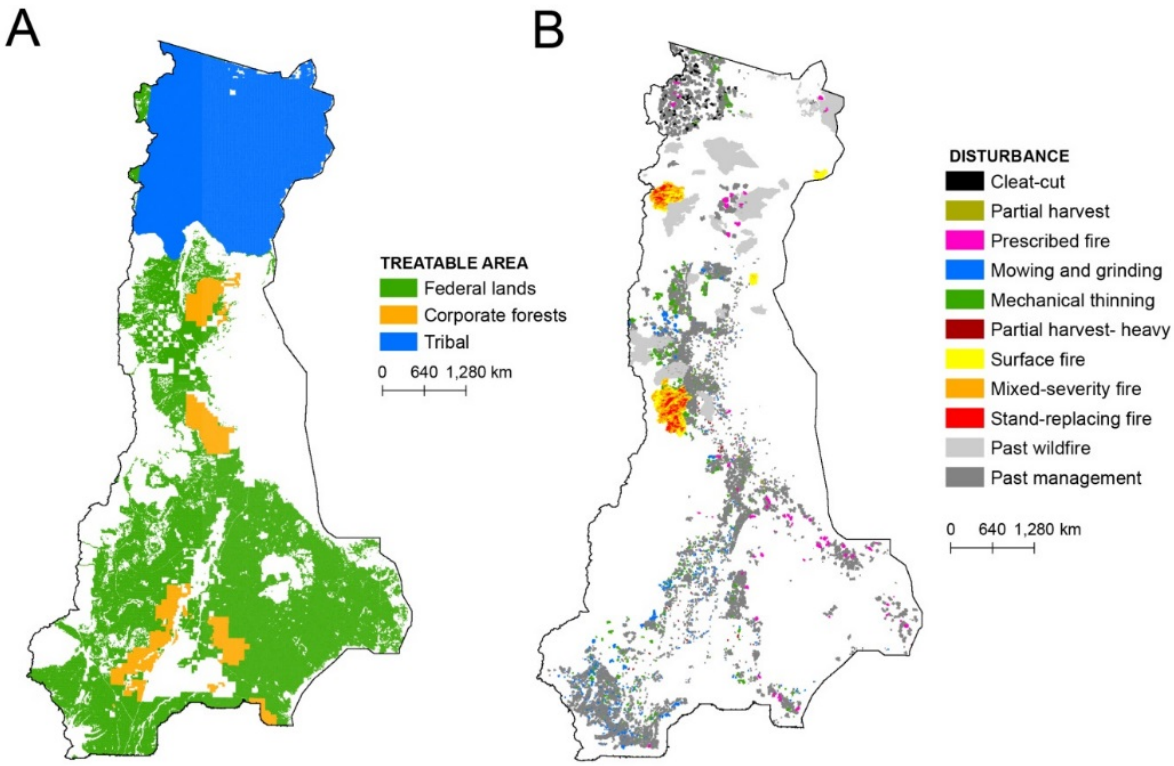

Fig. A2.1. Maps of A) treatable area and B) disturbance type (management and wildfire) at year 1 of the simulation (initial conditions) in the north sub-study area. Treatable area in federal lands corresponds to all forested lands excluding areas classified as wilderness and areas excluded from timber production due to biodiversity, conservation and amenity protection.

Table A2.2. Description of fuel model variants.

Fuel model variant	Description of variant and how it is applied	Time in variant (years)
1	Baseline fuel model for a vegclass	Remains the same until a disturbance or vegetation transition occurs
2	Assigned after a non-lethal surface fire in forested and non-forested (arid) vegclasses	5
3	Assigned after a mixed-severity fire in forested vegclasses	10
4	Assigned after a stand-replacing fire in forested vegclasses	10
5	Assigned after mowing/mastication treatments in forested vegclasses	5
6	Assigned after thinning treatments/partial harvests in forested vegclasses	5

Table A2.3. Fuel model codes assigned to post-disturbance conditions. All models are described in Scott and Burgan (2005) with exception of MAST, a custom fuel model for masticated fuel beds.

Baseline	Surface fire or prescribed fire	Mixed severity fire	Stand-replacing fire	Mastication	Thinning
Until transition/ disturbance	10 years	10 years	10 years	5 years	5 years
NB3	NB3	NB3	NB3	NB3	NB3
NB8	NB8	NB8	NB8	NB8	NB8
GR1	TL1	GR1	TL1	GR1	GR1
GR2	TL2	GR2	TL1	GR2	GR2
GR3	TL2	GR2	TL1	GR2	GR3
GS1	TL2	GS1	TL1	MAST	TL5
GS2	TL2	GR2	TL1	MAST	TL5
SH1	TL2	GS1	TL1	MAST	TL5
SH2	TL2	GS2	TL1	MAST	TL5
TU1	TL2	GR2	TL1	MAST	TL5
TU4	TL1	TL1	TL1	MAST	TL5
TU5	TL1	TL1	TL1	MAST	TL5
TL1	TL1	TL1	TL1	MAST	TL5
TL2	TL1	TL1	TL1	MAST	TL5
TL3	TL1	TL1	TL1	MAST	TL5
TL4	TL1	TL1	TL1	MAST	TL5
TL5	TL1	TL1	TL1	MAST	TL5
TL6	TL1	TL1	TL1	MAST	TL5
TL7	TL1	TL1	TL1	MAST	TL5
TL8	TL1	TL1	TL1	MAST	TL5
TL9	TL1	TL1	TL1	MAST	TL5

LITERATURE CITED

Scott, J. H. and R. E. Burgan. 2005. Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. Gen. Tech. Rep. RMRS-GTR-153, USDA Forest Service, Rocky Mountain Research Station.

Appendix 3. Additional Fire Perimeter Analysis.

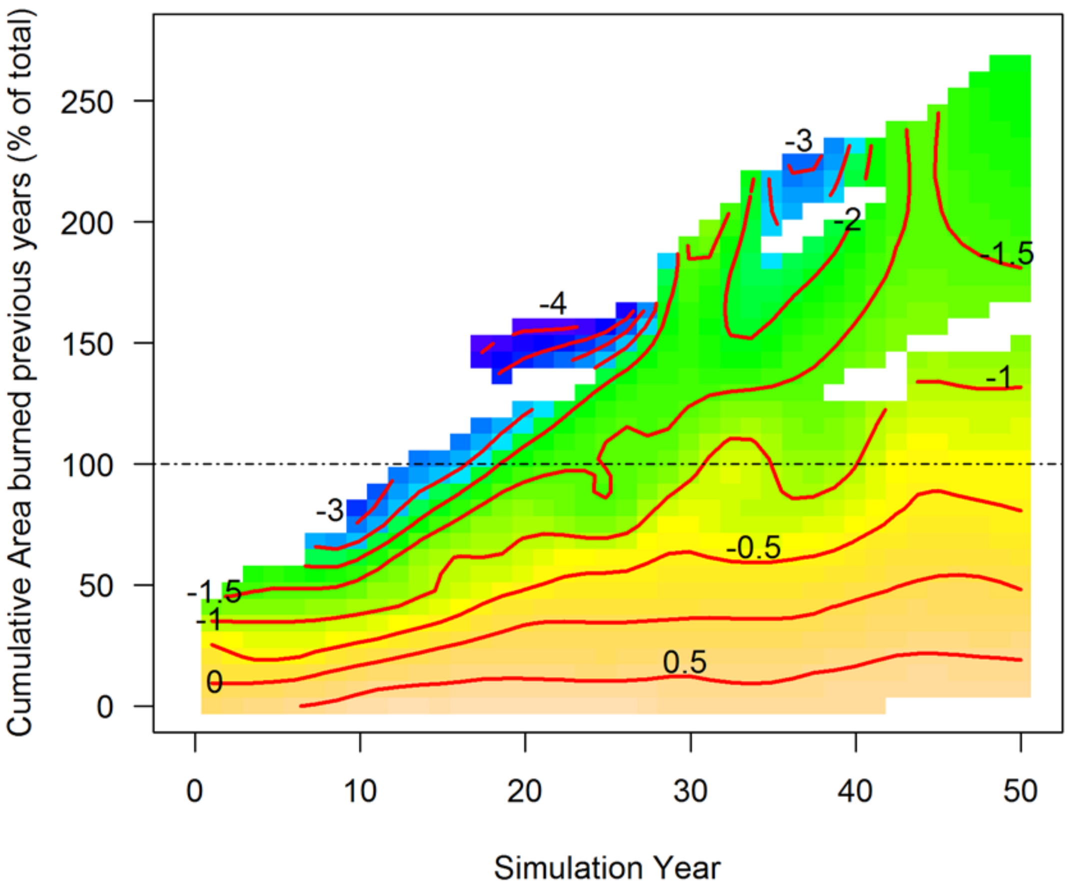

Fig. A3.1. Contour plot of change in average rate of spread (m min^{-1}) relative to the mean rate of spread as a function of simulation year and percentage of study area previously burned. The dashed line indicates the level at which an area equivalent to the whole study area (100%) was burned in previous years. Contour = 0 represents no change relative to average rate of spread.

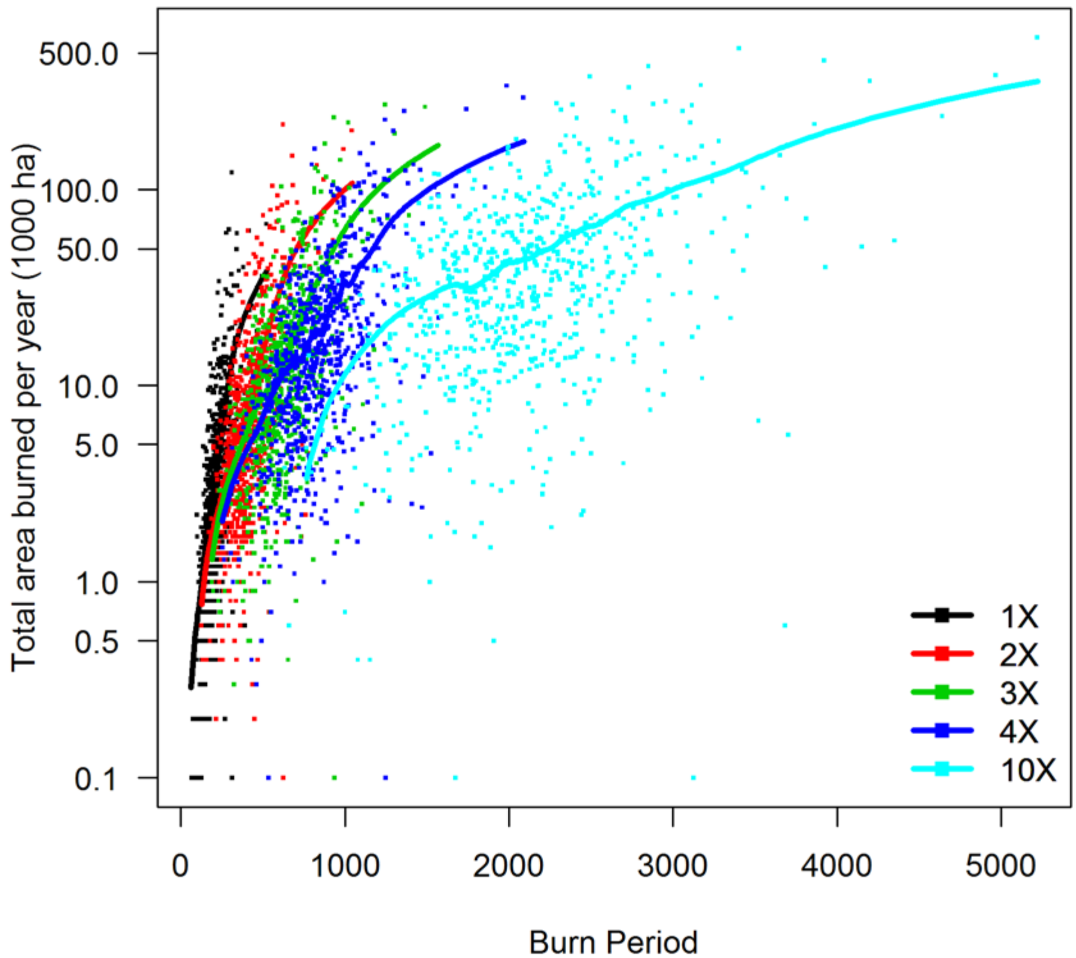

Fig. A3.2. Total area burned (1000 ha) as a function of burn period (minutes) for wildfire scenario (1X, 2X, 3X, 4X and 10X). Increased levels of wildfire activity in each scenario were achieved by multiplying the burn period of each ignition over contemporary levels by the multiplier indicated.