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RESEARCH ARTICLE

Contrasting effects of future wildfire and forest management scenarios on a fire excluded western US landscape

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Abstract
Context Restoring wide areas of fire excluded western US landscapes to fuel limited, fire resilient systems where fires self-regulate and burn with low or mixed severity will require expanded use of both prescribed and natural fire, coupled with strategic mechanical fuels management. However, optimal admixtures of fire and fuel management to set landscapes on trajectories to improve fire resilience and conserve carbon are not well understood.
Objectives To understand the effect of accelerating restoration and fuel management in response to potential future fire regimes on a large fire excluded mixed-owner forest landscape.
Methods We simulated 50-year wildfire and active forest management scenarios on a multi-owner landscape in southcentral Oregon, crossed in a factorial design with a range of wildfire and forest management intensities.
Results Wildfire was more efficient at reducing potential high-severity fire, whereas restoration treatments created patches of fire resilient old forest, especially on federally managed land. With some exceptions, both disturbances reduced aboveground carbon over time, although the magnitude varied among the combinations of fire and active management intensities. We observed interactive effects from specific combinations of fire and management in landscape response metrics compared to stand-alone disturbances.
Conclusions Fire and active management have similar landscape outcomes for some but not all restoration objectives, and active management will be required under future predicted fire regimes to conserve and create fire resilient old forest. Achieving widespread fire resilient forest structure will be limited by divergent landowner management behaviors on mixed-owner landscapes.

Keywords Envision · Fire severity · Forest landscape modeling · Wildfire resilience · Forest carbon

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Introduction

Regional droughts, land-use change, fire exclusion policies, and increasing frequency of human-caused ignitions are all contributing factors to recent trends in wildfire area burned and loss (Bowman et al.

2020). Climate change studies predict area burned in the western United States will increase 2 to 3 times based on both statistical and process models (McKenzie and Littell 2017; Kim et al. 2018). Governance systems are adapting to extreme fires as the new norm (Fischer et al. 2016; Abrams 2019) and a myriad of federal, state, and local community programs are being implemented and expanded to mitigate large fire impacts on both public and private lands (NFPA 2015; USDA Forest Service 2018). Federal and private forest restoration and fuel treatment programs on fire excluded forests continue to expand in scale (USDA Forest Service 2012, 2015), although assessments suggest that an increase in these activities by several orders of magnitude is required to reduce the backlog of forests that have undergone densification and fuel buildup (Merschel et al. 2018; Stephens et al. 2021).

Current federal and, to a lesser degree, private forest management practices combine concepts and principles of both ecological restoration and fuels management to achieve multiple socioecological objectives including restoring broad ecological resiliency to disturbance, and reducing risk to critical values including developed areas (Stephens et al. 2021). Forest restoration and fuel management are widely perceived to have many other tangential benefits to resource management objectives including wildlife habitat, watershed health, and providing raw materials to local wood products facilities (Kalies et al. 2016). Although the goals of fuel management and restoration activities can diverge in some aspects (Stephens et al. 2021), treatments on federal lands are highly synchronized in space and time with the broad goal of improving fire resiliency, forest health, and risk reduction. The coordination allows for improved economic and operational efficiencies, where for instance, commercial thinning activities to reduce overstocking can generate revenue that subsidizes mechanical surface fuels treatments and prescribed burning. By contrast, private land management in fire prone areas is focused on commercial and non-commercial harvests that reduce crown fuels and potential high severity fire without surface fuel reduction treatments (Charnley et al. 2017).

Federal fire management also recognizes a co-equal role for wildfire as an ecological restoration tool via policies that allow managing natural ignitions for restoration (i.e., resource objective fire, (Huffman et al. 2020)) on fire deficient landscapes (Parks et al. 2015). Resource objective (RO) fire under current policy consists of conscious decisions during the management of a fire event to allow or direct burning in particular areas. Empirical studies have shown multiple restoration benefits of RO fire, including improved understory conditions, reduction in fuels, and increased landscape fuel heterogeneity that confers fire resiliency compared to fire excluded landscapes (Hessburg et al. 2016; Churchill et al. 2017). RO fire under proper burning conditions has been an especially effective alternative to mechanical fuel management to reduce fire severity in large wilderness areas and national parks where mechanical treatments are not permitted (Huffman et al. 2020). However, the bulk of RO fire application and research concerning its effects comes from these specific areas in relatively few ecoregions (Collins et al. 2007; Collins and Stephens 2007; Fule and Laughlin 2007; Miller et al. 2012) where three decades of fire use policies have created patchy, fuel limited conditions and fires frequently encounter prior burned areas. Expansion of RO fire outside of protected areas is widely advocated, although wildfires burning on mixed owner, peri-urban, fire excluded landscapes brings substantial risk to people, property, and fire managers (Bowman and Johnston 2005; Black et al. 2008). Burning fire-excluded landscapes prior to mechanical treatments can create large patches of high severity fire that inhibit forest regeneration (Kemp et al. 2016; Owen et al. 2017) and catalyze transition to grass and shrub types (Kane et al. 2013), including non-native plants (Kerns et al. 2020). Large stand-replacing wildfires can also homogenize landscape fuels and set the stage for future large fire events, a trajectory that can be prevented by proactive fine-scale disturbance (small fires or management) that increases spatial heterogeneity in fuels (Peterson 2002; Stevens-Rumann et al. 2016).

Precisely how to expand the use of fire in synchrony with ecological restoration and fuel management to reduce the extent of high severity fire is a significant knowledge gap. Landscape modeling studies in dry pine forests suggest that treating landscapes to restore fire resilient conditions, followed by maintenance burning will substantially improve fire resiliency and carbon storage over time (Hurteau 2017; Krofcheck et al. 2017a; Liang et al. 2018; McCauley et al. 2019). However, other studies suggest natural ignitions occurring in the right place at the right time to achieve restoration objectives can be rare (Barros et al. 2018). Significant challenges also exist in the socio-political dimension since public trust and consensus among landowners to manage fire regimes with stochastic wildfire in multi-owner commercial forests may well conflict with both landowner objectives (Charnley et al. 2015) and statutes that govern state and local jurisdictions that mandate aggressive wildfire suppression to protect private land and property.

Given these broad challenges and uncertain future fire regimes, experiments are needed to understand what intensities of forest and fuel management are required to prepare and maintain fire excluded landscapes for amplified fire regimes that might materialize from changes in climate (Abatzoglou et al. 2017), fire management policy that includes wider use of RO fire (Huffman et al. 2020), or a combination of the two. We used the agent-based forest landscape succession and disturbance model Envision (Bolte 2018) to simulate a fully factorial design of multiple, plausible future fire and active management scenarios on a 2 million ha, multi-owner landscape in southcentral Oregon, USA. We modeled land manager preferences with respect to specific practices for federal and private industrial forests (Charnley et al. 2017; Spies et al. 2017) and examined landscape conditions over 50 years as measured by key restoration metrics, including stand structure, aboveground carbon, and potential high severity fire within the study area. We asked four interrelated questions: (1) What are the outcomes from different combinations of active forest management and wildfire disturbance over the 50-year simulation period? (2) Do specific combinations of management and fire accelerate landscape change? (3) What are the relative efficiencies of the two disturbances (change per area disturbed) and do specific combinations exhibit synergistic effects on landscape response? (4) How do private versus federal forest management practices affect the outcomes of each of the above?

Methods

Study area

The 2,073,112 ha study area was composed of public and private lands in southcentral Oregon (Fig. 1). Federal lands comprised 52% of the area and included the Fremont-Winema National Forest (FWNF) and land managed by the Bureau of Land Management (BLM). The FWNF covers over 1 million ha, 90% of which is available for forest management activities. The remaining regions of the FWNF (10%) include scenic and wilderness areas, recreation areas, and protected habitat. Public lands are interwoven with land managed by five major private industrial forest owners that collectively account for 17% of the study area. We defined major industrial landowners as corporate landowners owning more than 10,000 ha individually, hereafter private. The state of Oregon manages 1% of the study area, and private non-corporate and homeowners account for 11% and 14% of the study area, respectively (Fig. 1A). Vegetation (Halofsky et al. 2014a) varies east to west from arid lands (2%) and juniper (*Juniperus*, spp.) woodlands (15%) to fire-frequent ponderosa pine (*Pinus ponderosa*) forests (55%). Alpine forests (6%) are found at higher elevations and along the western boundary of the study area. Lodgepole pine (*Pinus contorta*) and unburnable lands correspond to 5% and 16% of the study area (Fig. 1B; Online Resource 1 Table A1), respectively. Conifer forests are widely overstocked as a result of fire exclusion with extensive areas of dense stands prone to high severity fire (Merschel et al. 2014; Spies et al. 2017). Overstocked conditions exist in both alpine and shrub steppe ecotypes, where in the former case whitebark pine forests (*Pinus albicaulis*) are being thinned to reduce competition from heterospecific conifers, and in the latter juniper that have encroached shrub steppe are being removed (Miller and Rose 1999).

Recent fire history data (Short 2017) showed that during the period 1992–2015 the study area averaged 268 ignitions year^{-1}, burning an average of 1930 ha year^{-1}. This equates to ca. 0.1% percent of the study area burned annually on average, or a fire return interval of 500 years. The maximum fire size was 37,626 ha during this period. Lightning was the primary source of ignitions (68%) and accounted for 79% of the area burned. Recent large fires include

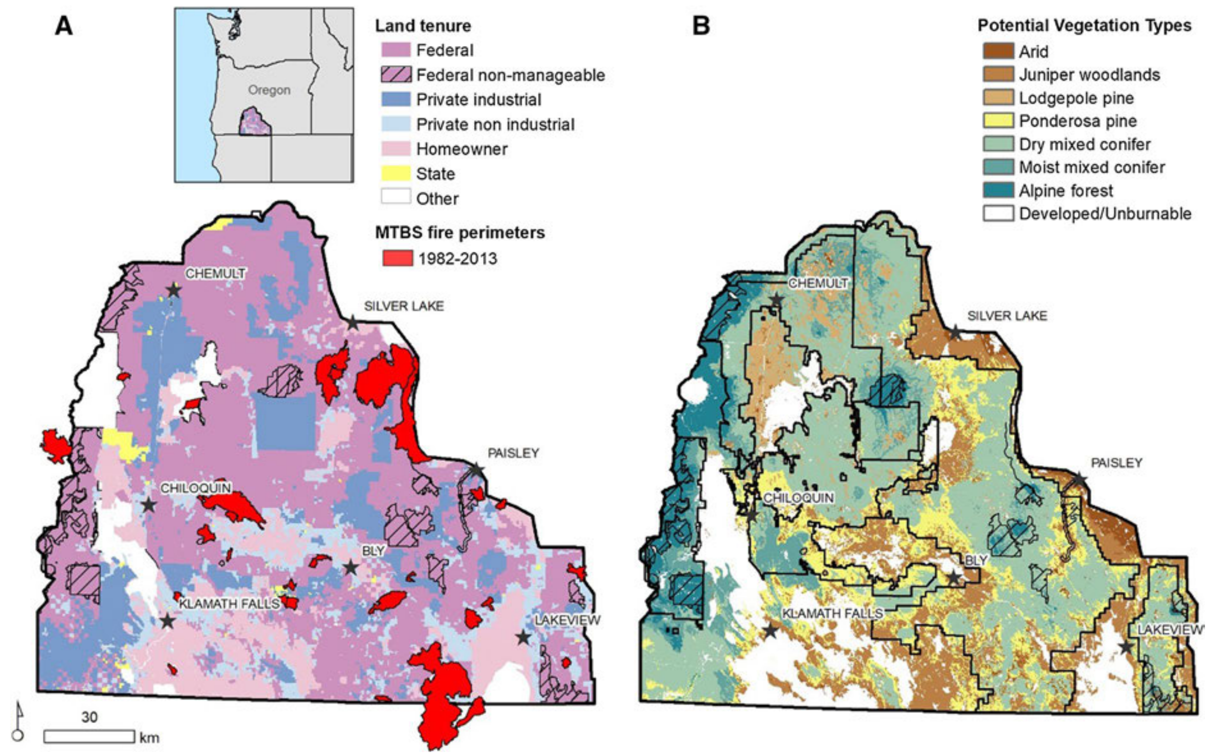


Fig. 1 Map of the study area showing (**A**) major land tenures and fire perimeters from 1982 to 2013 (MTBS 2017), and (**B**) potential natural vegetation types according to Halofsky et al. (2014b). Arid lands are defined as areas where tree cover is < 10% and contain shrub and herbaceous vegetation

the 2002 Toolbox (22,176 ha), 2012 Barry Point (37,626 ha), 2018 Watson Creek (23,901 ha; Fig. 1A) and most recently the 2021 Bootleg fire (167,425 ha).

In terms of natural fire regimes, 43% of the study area burns with a 0- to 35-year frequency, and 26% with a 35–200 year frequency, all with low to mixed severity fire (Landfire 2017). An additional 21% burns with a 35- to 200-year frequency with stand replacing severity (Online Resource 1 Table A2). Using the midpoint of the fire regime classes to estimate natural rates of burning from the fire regime classification, we estimate a pre-settlement fire return interval of 70 years (15,067 ha per year) or about 7–8 times the rate of contemporary burning. These statistics do not include the more recent Bootleg fire that burned 8% of the study area in July of 2021 after the study was completed.

Envision overview

Envision is an agent-based modeling framework that integrates spatiotemporal modeling of wildfire with forest management and vegetation succession submodels (Spies et al. 2017). The model and its submodels have been described in detail in several prior publications (Yospin et al. 2015; Hulse et al. 2016; Spies et al. 2017; Barros et al. 2017; Ager et al. 2018). Envision runs on an annual time step at the spatial scale of individual decision units (IDUs)—polygons on the landscape ranging from 0.1 to 10 ha. IDUs are defined by potential natural vegetation, current vegetation, and ownership. Current vegetation was defined by average tree size, canopy cover (CC) and the number of canopy layers (CL) (henceforth vegclass). In contrast, potential natural vegetation was determined from the ILAP project and consisted of 39 ecotypes. The combination of these two determines the vegetation's current state, potential successional pathways, eligibility for management, effects

Table 1 Effect of fire severity and management on average tree size, canopy cover, layering, and fuels in state-transition vegetation classes

Fire severity/management activity	Effect of disturbance/management activity			
	Size class	Canopy cover	Number of canopy layers	Fuels[a]
Surface fire (includes prescribed fire)	Change does not trigger state transition	Change does not trigger state transition	Reduces multi-layered states to single layer for some vegetation states	Reduces surface fuels; transitions to fuel model 181 or 182
Mixed-severity fire	Change does not trigger state transition	Decreases cover by one or two classes (e.g., high to moderate, moderate to low, moderate to open)	Reduces multi-layered states to single layer	Reduces surface fuels; post-fire fuel model depends on pre-fire state
Stand-replacing fire	Trees are killed; transition to grass-forb or shrub vegetation state	Decreases tree cover to none or low	No canopy layers remain	Reduces surface fuels; transitions to fuel model 181
Mowing and grinding	Change does not trigger state transition	Change does not trigger state transition	Change does not trigger state transition	Eliminates shrub layers and increases surface fuel; transition to custom model for masticated fuel beds
Pre-commercial thinning	Change does not trigger state transition	Decreases high cover states to low or moderate cover	Generally no change	Increases surface fuels; transitions to fuel model 185
Thin from below	Change does not trigger state transition	Decreases high cover states to moderate cover	Generally reduces multi-layered states to single layer	Increases surface fuels; transitions to fuel model 185
Partial harvest	Primarily no change	Generally decreases cover by one class in high and moderate cover states	Generally reduces multi-layered states to single layer	Increases surface fuels; transitions to fuel model 185
Partial harvest—heavy	Reduces size class by one (e.g., large to medium)	Decreases cover by 1–2 classes (e.g., high to moderate)	Reduces multi-layered states to single layer	Increases surface fuels; transitions to fuel model 185
Regeneration harvest	Trees are removed; transitions to grass-forb or shrub state	Decreases tree cover to none or low	No canopy layers remain	Increases surface fuels; transitions to fuel model 185
Post-fire salvage of dead trees	Change does not trigger state transition	Change does not trigger state transition	Change does not trigger state transition	Increases surface fuels; transitions to fuel model 185

Table modified from Spies et al. (2017)

[a]Fuel model 181 = low load compact conifer litter; 185 = high load conifer litter

of wildfire disturbance, and secondarily fire behavior via vegclass-specific fuel models (see below). Envision uses a probabilistic state and transition framework to model changes to each IDU based on prior work in the region and the Forests People Fire Project (Spies et al. 2017) (Table 1). In any given simulation year, depending on the vegclass and submodels described below, some IDUs will burn, some will be managed, some will transition into a new vegclass and there will be no change in many. At the end of each annual time step, vegclass attributes and fuel models (see below) are updated to reflect any processes that occurred in the IDU and will carry over to the following simulation year.

Envision includes submodels to simulate forest growth, succession and management, wildfire, and wildfire effects, each described below.

Existing vegetation and fuels data

Initial forest conditions reflected vegetation structure and disturbance around year 2012. Initial vegetation states (i.e., vegclasses) were identified from the gradient nearest neighborhood (GNN) dataset (Ohmann et al. 2011). The GNN data were created for Oregon and Washington states by imputing forest inventory plots to a 30 × 30 m grid using remotely sensed and topographic data (Ohmann et al. 2011). The inventory data were processed with the Forest Vegetation Simulator (FVS, Reinhardt and Crookston 2003) and summarized to determine the majority vegclass per IDU. Fuel models used in fire spread calculations for the FWNF were obtained from the forest's fuel model layer and LANDFIRE. We ran Envision without wildfire for five years to update succession and forest management to 2012, the initial simulation year. We used fire perimeters and the wildfire severity classification from the Monitoring Trends in Burn Severity (MTBS 2017) database to correct conditions for historical fires in the study area between 2006 and 2012. The resulting initial conditions layer, which was representative of the landscape around 2012, was dominated by open-canopy forest and small to medium trees. Approximately 60% of the study area had less than 40% canopy cover, and 30% was comprised of trees in small-size classes (DBH ≤ 38 cm). Areas dominated by medium (38 < DBH ≤ 51 cm) and large trees (DBH > 51 cm) covered 21% and 19% of the study area, respectively.

Succession and management submodel

Forest management included mechanical thinning, harvest, fuels mastication and prescribed fire. These actions differ in basal area removal and post-treatment forest structure and fuels (Spies et al. 2017). The management submodel was parametrized using ca. 50 interviews with local federal and private industrial land managers (Charnley et al. 2017), regarding the type of treatments, current application rate, and appropriate vegclass (Online Resource 1 Table A3). A key difference among the ownerships in management practice is the use of prescribed fire and retention of large trees on federal land (Online Resource 1 Table A3). Envision selects and prioritizes IDUs for treatment based on user-input criteria specific to each ownership and treatment type (Online Resource 1 Table A3). The model applies treatments to higher-ranking IDUs until the annual area (or volume) target for each action is met or if no additional IDUs match the treatment criteria.

Forest succession is modeled through a state-and-transition model that includes both probabilistic and deterministic (based on time) rules that simulate succession trajectories over the course of 50 years for the vegclasses in the study area. The state-and-transition model was adapted from existing models for Oregon (Hemstrom et al. 2004). Probabilistic rules can include, for example, the development of understory layers that will change a stand from a single to multilayer vegclass while keeping all other attributes the same. Deterministic transitions are based on time, reflect how growth affects the attributes that characterize an IDU (size, cover, and number of layers), and are based on empirical knowledge of potential vegetation types.

Wildfire submodel

The wildfire submodel in Envision uses the minimum travel time (MTT) fire spread algorithm (Finney 2002) and associated crown fire models as implemented in the FlamMap application interface (henceforth FlamMap API) (Brittain 2015; Finney 2006). Space considerations preclude detailed description and the reader is referred to Ager et al. (2018) for additional details. In each simulation year Envision calls the wildfire submodel by translating the IDU conditions to surface and canopy fuels then writing a binary gridded (90×90 m) input file for the FlamMap API. The wildfire submodel ingests fire ignitions generated from a prediction system (described below), including fire weather and burn period. The MTT fire spread algorithm simulates fire as a function of time and the method does not permit simulation of specific sizes other than using time to achieve approximate area burned per fire (Finney 2002). The MTT and associated fire behavior algorithms simulate all fires predicted for the current simulation year, and fire perimeters and gridded flame length are output. Since there are multiple grid cells per IDU polygon, then the average flame length for each affected polygon is calculated and used to interpret fire effects (see fire effects section below). The FlamMap API as implemented in Envision also models a potential flame length that would be realized if each IDU burned independently under an assumed fire weather scenario. This process is typically referred to as a "static" fire simulation in FlamMap. Potential flame length was modeled each simulation year after surface and canopy fuels were updated for succession, wildfire, and management effects. Based on prior fire modeling in the study area (Online Resource 2 Fig. B2, Table B2), we used a constant wind speed = 29 km h^{-1} and wind azimuth = 220° for the static runs with fuel moistures 5%, 6%, 9%, 50% and 82% for 1-h, 10-h, 100-h, live herbaceous and woody fuels, respectively.

Daily wildfire probability, ignition location, cause (human or natural), and fire size are predicted by a spatiotemporal model (Ager et al. 2017) (R Core Team 2014) executed prior to simulations to generate daily fire occurrence and size. The model uses daily energy release component (ERC, Finney et al. 2011; Preisler et al. 2016; Holden et al. 2019) to predict fire size and ignition probability based on a sample of 11,618 ignitions (1992-2011) within a 3.32 million ha region surrounding the study area (Ager et al. 2017). Daily ERC data were obtained from the Remote Automated Weather (RAWS) USA Climate Archive (Western Regional Climate Center 2014) for 1961–2011. The fire prediction system generates a stream of ERCs based on an autoregressive model of historical ERCs. Thus each execution resulted in a unique fire list for a unique 50-year fire scenario. The resulting firelist contains attributes for day-of-year, ERC, fire weather parameters (wind speed and azimuth), ignition probability, burn period, fire cause (natural or human), ignition location (XY coordinate), and fuel moisture conditions. Details on fuel moisture, wind speed and direction and burn period can be found in Online Resource 2 and Ager et al. (2018).

Fire effects

The fire effects submodel in Envision (Ager et al. 2018) classifies fire severity at the IDU level based on the IDU's potential flame length in a fire simulation (i.e., to assess landscape-level fire effects). Threshold flame lengths that triggered a change in the vegclass were obtained from modeling plots (n = 25,000) representing each vegclass in FVS. FVS incorporates species specific tree mortality functions initially developed as part of the First Order Fire Effects Model (FOFEM). FOFEM uses logistic regression to predict probability of mortality based on scorch height and crown fraction burned. The plots representing each vegclass were subjected to a range of flame lengths and the resulting tree mortality was used to identify the fire thresholds associated with three classes of fire severity (low severity surface, mixed-severity, and stand-replacing high-severity). The tree mortality levels for each severity class were ≤ 20% for low, 20–80% for mixed, and > 80% for high (Spies et al. 2017). This process explicitly accounted for species-specific mortality rates for each vegclass in the model and provided for a range of fire severities and resulting state changes among different vegclasses, including transient forest to shrub and grass transitions in the case of high severity fire. We note that one limitation is that the model does not provide for permanent type changes from forest to grass and shrub observed and reported (Coop et al. 2020) in some recent extreme fire events.

Model validation

The Envision model underwent extensive validation as part of prior research in the Forests People Fire Project as described elsewhere (Spies et al. 2017; Ager et al. 2018) and summarized in Online Resource 3. Submodels were derived or calibrated from pre-existing models that have seen extensive and long-term application in the research community (Dixon 2002; Barbour et al. 2007; Finney et al. 2011). The wildfire submodel in particular underwent extensive

Table 2 Management constraints and preference assumptions by landowner and activity

Landowner	Action	Constraints	Preferences[b]
Federal	Thinning	Primarily thinning from below, removing 20% of the stand volume *only* in stands not thinned in the last 14 yrs, with mean DBH > 25 cm and multistory closed canopies (> 60%) and slopes < 30%[a]	Ponderosa pine > dry mixed-conifer > lodgepole pine > basal area greater than 20.67 m^2 ha^{-1} > med or high canopy closure stands > within WUI > dry mixed conifer or ponderosa pine with high potential for high-severity fire > dry mixed conifer or ponderosa pine with high potential for mod-severity fire > moist mixed conifer
	Prescribed fire	Same constraints as thinning except on single story stands and only in dry mixed conifer or ponderosa pine; possible only 9 yrs after previous prescribed burns	Thinning happened within last 4 yrs > ponderosa pine with single-layered low closure canopies > dry mixed conifer with single-layered, low closure canopies > dry mixed conifer or ponderosa pine with high potential for mod-severity fire > fuel model is high-load dry[c]
	Mastication	Same constraints as thinning except only fuel models that are high load dry climate shrubs, very high load climate shrubs or very high load dry climate timber-shrub (Scott and Burgan 2005); possible only 9 yrs after last disturbance	Within the WUI > within 400 m of a major road > last fire was 20 or more years ago > fuel model is high-load dry[c] > fuel model is moderate load dry climate shrub
	Salvage harvesting	Possible only 1–3 yrs after stand-replacing fire and meeting criteria for thinning, removes 50% of the remaining standing sawtimber volume	Within 366 m of a road > Within area designated as potential habitat for northern spotted owl
Private Industrial	Commercial harvesting	Partial harvest, removing 75% of the stand volume in any forest type with mean DBH > 25 cm; possible only > 19 yrs since last harvest	Basal area greater than 23 m^2 ha^{-1} > slope less than 30%
	Salvage harvesting	Possible only 1–3 yrs after stand-replacing fire on slopes < 30% in any stand of commercial timber species, removes 50% of the remaining standing sawtimber volume	Distance to road less than 366 m

[a] Stands designated as wilderness or deemed unsuitable for commercial timber, and nesting habitat for spotted owls are also excluded; and at a tree level, no trees > 53.34 cm DBH will be removed, however this is not built into the model explicitly, it is assumed those trees would be in the remaining 80% of uncut volume

[b] See also Spies et al. (2017) Appendix Table A6.1

[c] High load fuel model = fuel model is high-load dry climate shrub, very high load dry climate shrub, very high load dry climate timber-shrub or moderate load dry climate shrub

calibration and testing as part of prior research in the study area (Ager et al. 2018). The underlying MTT algorithm is a core component of federal wildfire planning systems (Andrews 2007; Finney et al. 2011). The management activities in the vegetation model were also developed as part of workshops with land managers within the study area and collection of data describing management activities on the respective ownerships (Charnley et al. 2017). Results from the Envision model have been used in numerous workshops in the region covered by the study area to explore management scenarios (OSU 2014).

Experimental design

We combined wildfire and active forest management into nine scenarios where each disturbance was simulated at three different levels (1X, 3X and 5X) in a fully factorial design (Online Resource 1 Table A4), with the addition of fire-only and active forest management scenarios (n = 15 scenarios). Management was amplified by increasing the area treated rather than treatment intensity since treatments are based on well-developed silvicultural and fuel management practices in the study landscape. Contemporary area burned was modeled based on 1992–2009 fire activity in the study area (Ager et al. 2018). Contemporary fire was amplified by multiplying the burn period for each ignition by a factor of three and five. In the same study area, Barros et al. (2021) found amplification of future area burned was driven by fire size and not frequency. Note that this method does not generate exactly 3X and 5X burned area since fire perimeters expand non-linearly over burn periods. Several other fire simulation parameters could have been manipulated to increase area burned (fuel moisture, wind speed, spatiotemporal ignition distribution) although changes in these parameters can also alter fire intensity and complicate interpretation. All scenarios shared the same ignitions and burning conditions (wind speed, wind direction, and ERC on ignition day). Replicates for each scenario (n = 10) varied in terms of ignition location, day of year, and fire size, according to a spatiotemporal ignition model described above. Variation among replicates also materialized from probabilistic vegetation state transitions, fire-on-fire feedbacks, and fire-treatment interactions.

Contemporary levels of active forest management were based on business as usual determined from interviews with federal and private managers (Charnley et al. 2017)(Table 2) and used to develop a forest management plan (Table 2, Online Resource 1 Table A3). The plan specified treatment area targets (thin, harvest, underburn) per year, thus treatment area and not intensity was increased. Area treated in any given year can be lower than the specified targets when the conditions do not exist for a particular treatment (e.g., wildfire burns stands before they are scheduled for treatment). Thus, the model incorporated dynamic feedbacks from all combinations of fire and active management over time.

Response metrics

We studied five response variables: (1) annual area burned by wildfire; (2) area of resilient forest structure; (3) area of potential high-severity fire; (4) aboveground carbon; and (5) forest structure (tree size and density). Annual area burned by wildfire by scenario and replicate was calculated from fire perimeter shapefiles generated by Envision for each simulated fire. Resilient forest structure (henceforth resilient forest) was defined as the area of ponderosa pine, dry mixed or moist mixed-conifer forest with large trees (DBH > 51 cm) and open canopies (CC ≤ 40%); or with moderate-size trees (38 < DBH ≤ 51 cm), a single canopy layer, and moderate canopy cover (40% < CC ≤ 60%) (Merschel et al. 2014; Spies et al. 2017). Both empirical and wildfire simulation studies support this broad definition of forest conditions that are resilient to wildfire, i.e., surface and canopy fuels that burn with low severity fire. Resilient forest excluded alpine and subalpine forests and stands dominated by lodgepole pine and juniper. Potential high-severity fire was calculated by applying the flame length measured in the static fire behavior simulation to fire severity thresholds for each vegclass (see Fire effects section above). The area of all IDUs that was predicted to burn at high severity (versus low or mixed) in future fires (i.e., potential) was then tabulated by scenario and year. Carbon (Mg ha^{-1}) was the mass of live and dead aboveground carbon using regression models from approximately 25,000 inventory plots that were used to represent the different vegclass categories (Spies et al. 2017,

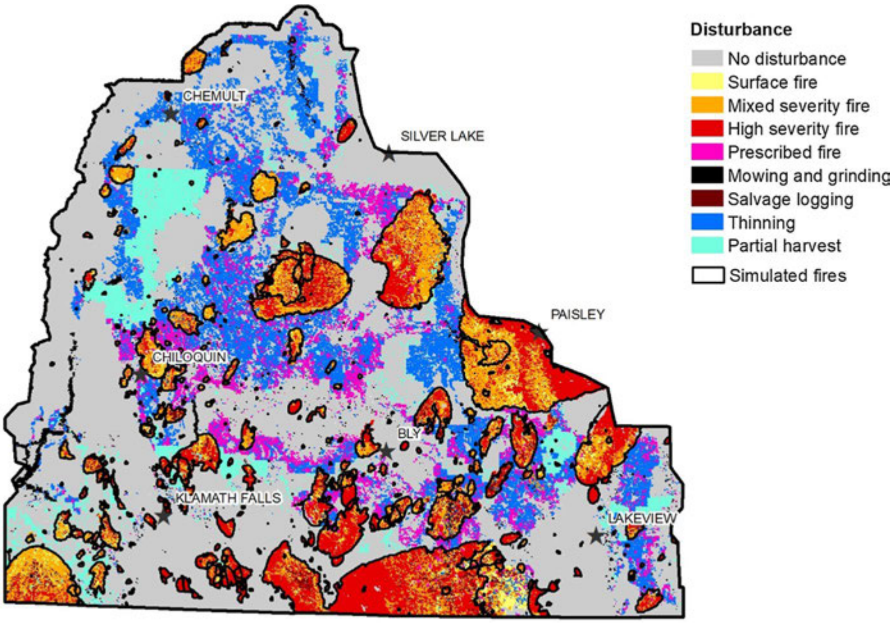


Fig. 2 Simulated active management activities and wildfire over a 50-year period for one replicate of the M1F1 scenario, which targeted contemporary levels of active management and wildfire activity. See Fig. A1 in Online Resource 1 for an example of treatments in a typical simulation year

as reported in Appendix 9). The plots were processed with FVS to obtain mean aboveground carbon for each vegclass. Forest structure was measured in terms of tree diameter and canopy cover. Three size classes were: (1) early successional forest defined as grasses, shrubs and trees < 12 cm DBH; (2) small-tree forest defined as trees > 12 cm DBH and ≤ 37 cm DBH; and (3) large-tree forest defined as trees > 50 cm DBH. Three canopy cover classes were defined as: (1) open canopy (CC < 40%), (2) medium-canopy (40% ≤ CC ≤ 60%) and (3) closed-canopy (CC > 60%) forest.

Analysis

For each scenario we examined area treated and burned among the ten replicate runs with boxplots. For three of the five response variables (resilient forest, potential high severity fire and carbon) we examined the relative effects of wildfire versus active forest management alone and in combinations using generalized additive models (GAM, Wood 2006a) with cumulative area treated, cumulative area burned, and simulation year as explanatory variables (R Core Team 2019). The procedure utilized semi-parametric spline functions to estimate the relationships between the dependent variable and the various explanatories. Observations consisted of annual values for each response metric and replicate combination (n = 7500). The GAM models to estimate the effects of area treated and burned for the response variables were:

$$\mathit{resilient\ forest} \sim a1 + g_{11}(\mathit{Wildfire}) + g_{12}(\mathit{Management}) + g_{13}(\mathit{Year}) + \tau$$

$$\mathit{potential\ high\ severity\ fire} \sim a2 + g_{21}(\mathit{Wildfire}) + g_{22}(\mathit{Management}) + g_{23}(\mathit{Year}) + \tau$$

$$\mathit{aboveground\ carbon} \sim a3 + g_{31}(\mathit{Wildfire}) + g_{32}(\mathit{Management}) + g_{33}(\mathit{Year}) + \tau$$

where *Wildfire* = cumulative area burned at each time step as a percentage of the study area

Management = cumulative area managed at each time step as a percentage of the study area

Year = simulation year, and τ = Gaussian error.

The intercepts, $a1 \ldots a3$, are the overall average levels of the corresponding dependent variables. The functions, g, are non-parametric smooth spline functions (Wood 2006b) included in the model to account for potential nonlinear effects of year, cumulative area previously burned, and area treated.

Results

Area burned and treated

Simulated wildfires and treatment units exhibited patterns of shape, size, and distribution consistent with historical observations and current management practices (Fig. 2). Area treated for scenarios without fire (M1, M3 and M5) varied little among replicates (Fig. 3A) since in the absence of wildfire the treatment locations and targets were the same across space and time. When scenarios included wildfire, variability in area treated among replicates increased and the median value of area treated declined with increasing levels of fire (Fig. 3A), indicating that wildfire competed with forested area targeted for active management. Area burned varied mostly among different fire scenarios, although there were differences among replicates and active management levels (Fig. 3B). Under the contemporary level of wildfire (F1 scenarios), variability in area burned among replicates was relatively small and minimally affected by management level, while at higher fire levels, variability among replicates increased and median area burned decreased as management increased. The largest fire observed in the simulations was 1.2 million ha in the F5M5 scenario, over 250 times the mean fire size across all scenarios. The reduction in area burned per area treated varied among the fire scenarios, with F5 scenarios resulting in the largest treatment effect (Online Resource 1 Fig. A5). The treatment leverage (*sensu* Price et al. 2015) was estimated at − 0.04 for the F1 scenarios, and − 0.29 for the M5 scenarios (Online Resource 1 Table A5). For instance, under the M5F5 scenario, treating 100 ha resulted in a reduction in area burned by wildfire of 29 ha.

The amount and type of active management activities allocated in the simulations differed substantially between federal versus private lands (Table 3), reflecting the divergent practices determined from landowner surveys (Charnley et al. 2017)(Table 2). For all scenarios, thinning and prescribed fire were the predominant federal treatments over the 50-year simulation, with lesser amounts of mastication (Table 3). By contrast, on private forests, nearly all treatments available for selection were commercial harvest with a small percentage of salvage logging (Table 3).

Table 3 Percentage of the land base treated by treatment type for two of the simulated scenarios for private and federal lands with contemporary levels of management and wildfire activity (M1F1) and 5X contemporary (M5F5)

Treatment type	Percent of land base			
	M1F1		M5F5	
	Federal	Private industrial	Federal	Private industrial
Thinning	35.4	0	45.3	0
Prescribed fire	15.8	0	72.3	0
Mastication	3.3	0	16.2	0
Harvest	1.0	20.7	3.1	79.2
Salvage	0.8	3.3	1.2	5.9

Resilient forest

The change in area of resilient forest in response to active management and wildfire was rapid and highly variable among scenarios (Fig. 4A; Online Resource 1 Table A5). All scenarios led to increases in fire resilient forest, except F1 (fire at contemporary levels with no management) where there was a 20% reduction by the end of the simulation. On federal lands, resilient forest increased the most (15 to 35%) in scenarios with high levels of active management regardless of fire level (Fig. 5A). Scenarios that excluded active management (F1, F3, F5) led to relatively low area of resilient forest. On private lands, resilient forest declined over time for all but four scenarios, with only minor increases (1–4%) (Fig. 5B). Under contemporary active management levels and amplified wildfire (M1F3, M1F5), the amount of resilient forest slowly and steadily increased over time and never plateaued whereas high management scenarios showed relatively rapid increases in the first 20 years of the simulation (particularly M5 and M5F1). The effect of active management versus fire was evident in the results of the GAM modeling (vertical contour lines, Fig. 6A), where the largest area of resilient forest (> 14%, blue region on the contour plot) was associated with multiple combinations of management and wildfire ranging between 60 and 100% of the study

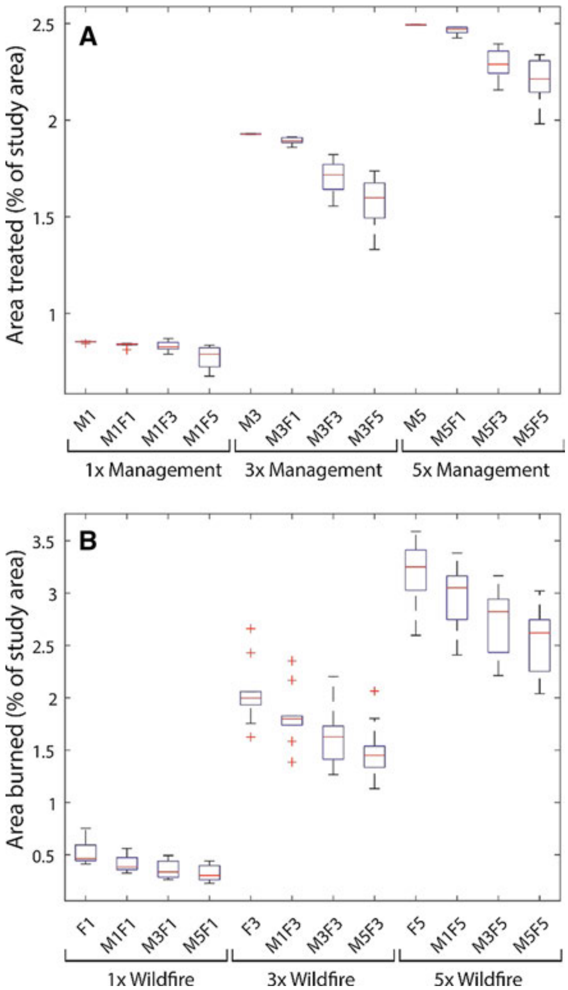


Fig. 3 Annual area treated (**A**) and annual area burned (**B**) as a percentage of the study area for each of the 15 scenarios. Individual observations are averaged across replicates. The central line within each box indicates the median value, and the bottom and top edges of the box indicate the 25th and 75th percentiles. Replicate outliers plotted as a plus symbol. Note that M0 (no active management) is not shown for area treated (**A**) and F0 (no fire) is not shown for area burned (**B**). *F* = wildfire scenario, *M* = active management scenario; 1, 3 and 5 indicate 1X, 3X and 5X the contemporary levels of each scenario. Note the ordering of the *x*-axis varies in each panel

area treated, combined with 40 to 100% of the study area burned.

Potential high-severity fire

The change in potential high severity fire was also highly variable among the scenarios in terms of rate and magnitude over the simulation period (Fig. 4B; Online Resource 1 Table A5). Scenarios ranked by overall change in high severity fire (Fig. 4B) showed that combined active management and fire led to the greatest reduction. On federal lands, area of potential high severity fire was reduced for all scenarios except F1 (Fig. 5C). Several large wildfires in the early decades of the simulation led to a large reduction in area of potential high severity fire (Fig. 5). Conditions that contributed to high severity fire on private lands increased under all scenarios with high levels of management and low levels of fire (Fig. 5D). For example, under M1 and M3, potential high severity fire on private lands increased from 39% to close to 55% (Fig. 5D), while the opposite trend was seen on federal lands (Fig. 5C). For the study area as a whole, wildfire was more efficient at reducing potential high severity fire compared to active management (horizontal contour lines, Fig. 6B), with a reduction by 50% when 100% of the landscape was burned with wildfire over the 50-year simulation.

Aboveground carbon

There were only two scenarios that resulted in an increase in carbon (M1 and F1) and one scenario led to no change (M1F1; Fig. 4C; Online Resource 1 Table A5). All other scenarios resulted in a carbon loss during the simulation period (Fig. 4C). Rate of loss was rapid for most scenarios and the trends were bifurcated into two groups (Fig. 7). On federal lands carbon increased only in the lowest management and fire scenarios (M1 and F1, Fig. 7A), and showed the largest decrease, from 55 to about 33 Mg ha^{-1} for the M3F5 scenario. Some scenarios generated similar levels of change in carbon despite differing levels of area treated or burned by wildfire (e.g., F3, M5, Fig. 8). Results from the GAM model showed an equal effect between the two disturbances on carbon loss (Fig. 6C), and a reduction in carbon of about 0.31 Mg ha^{-1} for each 1% of the study area disturbed by either fire or active management. Under high fire scenarios, management moderated carbon loss compared to low or no fire (Fig. 8).

Changes over time in forest structure

As expected, increasing wildfire led to an expansion of early successional forest, mostly at the expense of

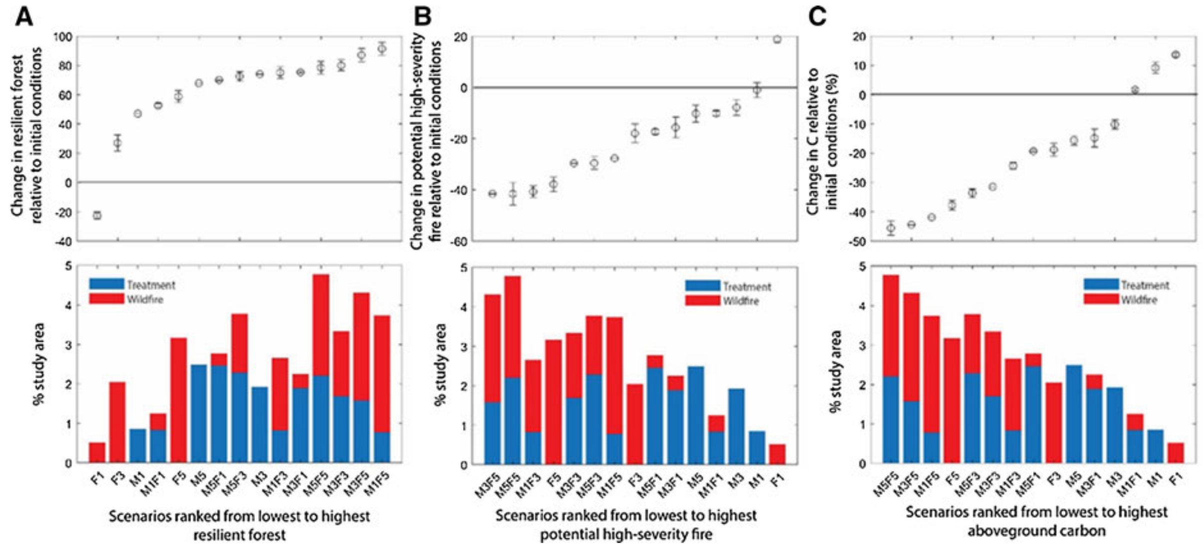


Fig. 4 Effects of different combinations of fire and active management on the change in (**A**) fire resilient forest area, (**B**) area of potential high-severity fire, and (**C**) aboveground carbon. Top panels show the average change in each landscape metric as a percentage of the initial conditions ± standard error. Bottom panels show the corresponding percentage of the study area that was either burned by wildfire or treated in each scenario each year. Within each panel, scenarios are sorted from the lowest to highest amount of change for the given response metric. For example, under the F1 scenario 0.5% of the study area was burned by wildfire annually without any area treated, which resulted in a 20% decrease in resilient forest area over the course of 50 years. *F* = wildfire scenario, *M* = active management scenario; 1, 3 and 5 indicate 1X, 3X and 5X the contemporary levels of each scenario

the small-tree structure class (Fig. 9A, B). Scenarios with high levels of active management, either combined with wildfire or as a stand-alone disturbance, had little impact on the area in early successional vegclasses and small-tree forests. High levels of forest management led to increases in mature forest, particularly under management-only scenarios and combined scenarios where fire was at contemporary levels (Fig. 9C). In scenarios with amplified wildfire, mature, large-tree forests also increased (e.g., from 18.5 to 19.5% under M1F3, representing a nearly 20,000 ha increase), albeit less than under scenarios with increased management. More importantly, the area of mature trees did not decline by the end of the 50-year simulations under any of the scenarios—not even under F5. This suggests that tree mortality under amplified wildfire scenarios remained low, and that management strategies aimed at maintaining resilience can be effective even under amplified wildfire scenarios, at least on federal lands (see Fig. 9C).

Increasing wildfire and active management relative to contemporary levels increased open forest over the 50-year simulations (Fig. 9D). The increase in open stands was negatively correlated with the reduction in stands with moderate canopy closure (Fig. 9E), the primary target of most thinning and harvest management prescriptions in this study. Wildfire and management each contributed to opening the canopy through either burning, harvesting or thinning in stands with moderate canopy closure. Under the contemporary fire-only scenario (F1), open canopy stands declined over the 50-year simulation from 20% of the landscape to under 15% (Fig. 9D; representing a decrease of over 100,000 ha), due to the absence of management. Closed forest (Fig. 9F) increased over time under the scenarios that included contemporary levels of wildfire and active management (M1F1).

Discussion

An important area of landscape ecology research is the investigation of how multiple natural and anthropogenic disturbances interact over time and alter landscape conditions (Darling and Cote 2008). To that end, we examined outcomes from different

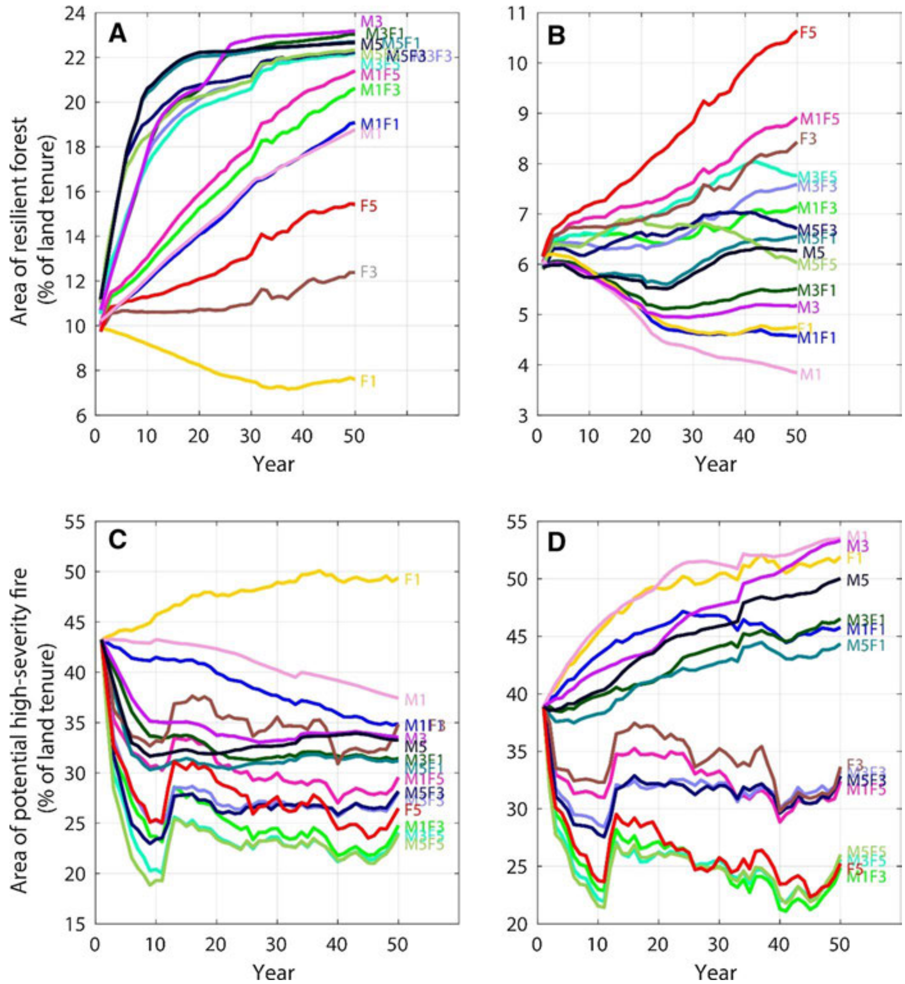


Fig. 5 Change over time in landscape response metrics for different combinations of forest management and wildfire for each of the 15 scenarios. Data are shown separately for federal and private industrial ownerships. Panels (**A**) and (**B**) present trends in the percent of resilient forest on federal and private lands, respectively. Panels (**C**) and (**D**) present trends on area with potential for high severity fire on federal lands and private lands, respectively. Note differences in y-axis scale for the two. *F* = wildfire scenario, *M* = active management scenario; 1, 3 and 5 indicate 1X, 3X and 5X the contemporary levels of each scenario

combinations of active forest management and wildfire disturbance on private versus federal forest lands, and whether specific combinations of management and fire accelerated landscape change. The relative efficiencies of the two disturbances (change per area disturbed) were also analyzed to identify potential non-additive synergistic effects on landscape response. On both ownerships contemporary fire (i.e., under current suppression practice) without active management led to increased carbon and potential high severity fire, and lower area of fire resilient forests, as forests continued to densify under fire exclusion (Merschel et al. 2018). As expected, higher levels of fire (F5) reversed these trends by triggering tree mortality, changes in forest structure and lower aboveground carbon. On federal lands, admixing active management with amplified fire accelerated these trends, especially in the first 20 years of the simulation (Figs. 5, 7). However, admixing active management and wildfire on private industrial lands reduced the rate of increase in fire resilient forests, and increased area of potential high fire severity.

The effect of amplifying both fire and active management was to accelerate landscape change compared to their individual effects, especially the relative proportions of forest structure classes. In terms of the relative effect of the two disturbances, we observed scenarios where fire and active management were equally efficient (carbon, discussed below), or responded to only fire (Fig. 6A) or management (Fig. 6B), with additive (Fig. 6C) and non-additive interactions apparent (Fig. 6B). Both antagonistic, and to a less extent, synergistic effects were observed (*sensu* Darling and Cote 2008).

Landowner management preferences changed the relative effects of the two disturbances. On federal

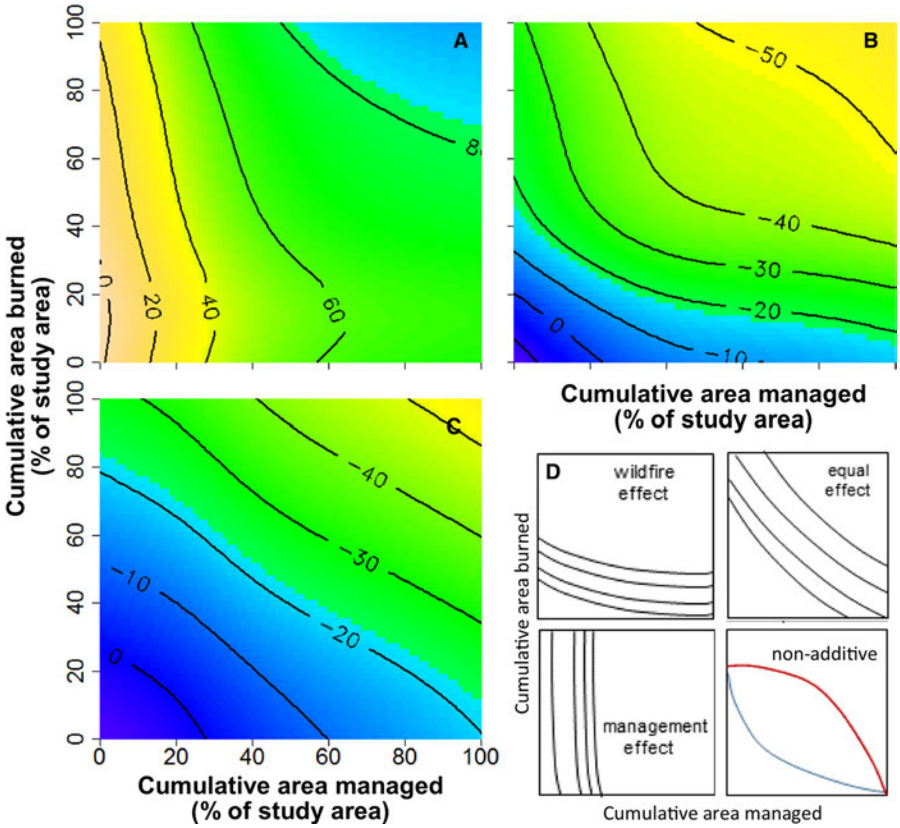
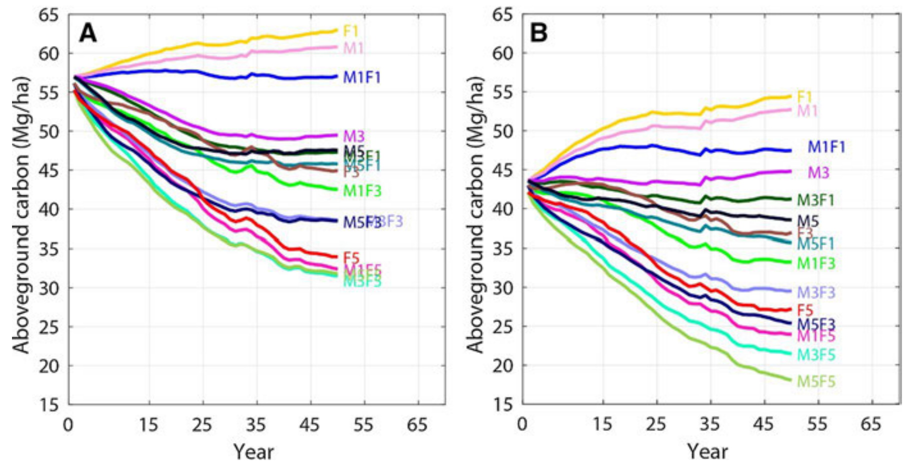


Fig. 6 Contour plots showing the relative effects of active management versus wildfire on rate of change in (**A**) percent area of resilient forest, (**B**) percent area with potential high-severity fire, and (**C**) aboveground carbon. Rates were calculated as the difference in any given simulation year relative to initial conditions (simulation year one). Note that cumulative values of the predictor can correspond to varying time intervals. All models used cumulative area burned and cumulative area treated as predictors for each year in the simulation. For example, 20% of the study area managed and 40% of the study area burned may occur in year 10, 22, and 30. The lower right panel (**D**) is an interpretation of the contour plots in terms of the relative effects of active management versus wildfire. Steep contour gradients on the y-axis indicate stronger effects of fire compared to management. In contrast, steep contour gradients on the x-axis indicate stronger effects of management relative to fire. Nonlinear contours indicate non-additive, interactive effects

Fig. 7 Change over time in aboveground carbon on federal (**A**) versus private lands (**B**) for each of the 15 scenarios. F = wildfire scenario, M = active management scenario; 1, 3 and 5 indicate 1X, 3X and 5X the contemporary levels of each scenario



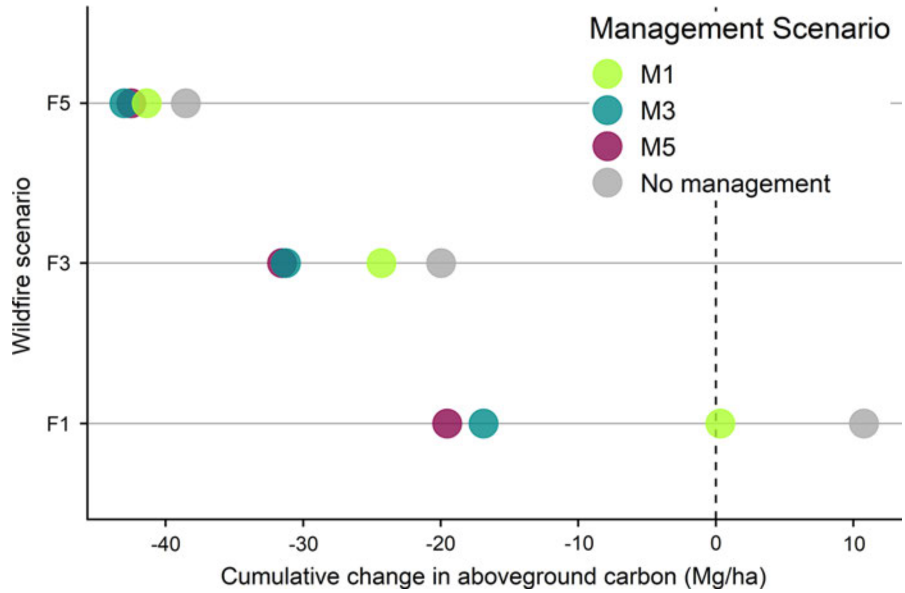


Fig. 8 Cumulative change in aboveground carbon from initial conditions (year 1) to year 50 for each of the 15 scenarios grouped by levels of active management and wildfire. F = wildfire scenario, M = active management scenario; 1, 3 and 5 indicate 1X, 3X and 5X the contemporary levels of each scenario

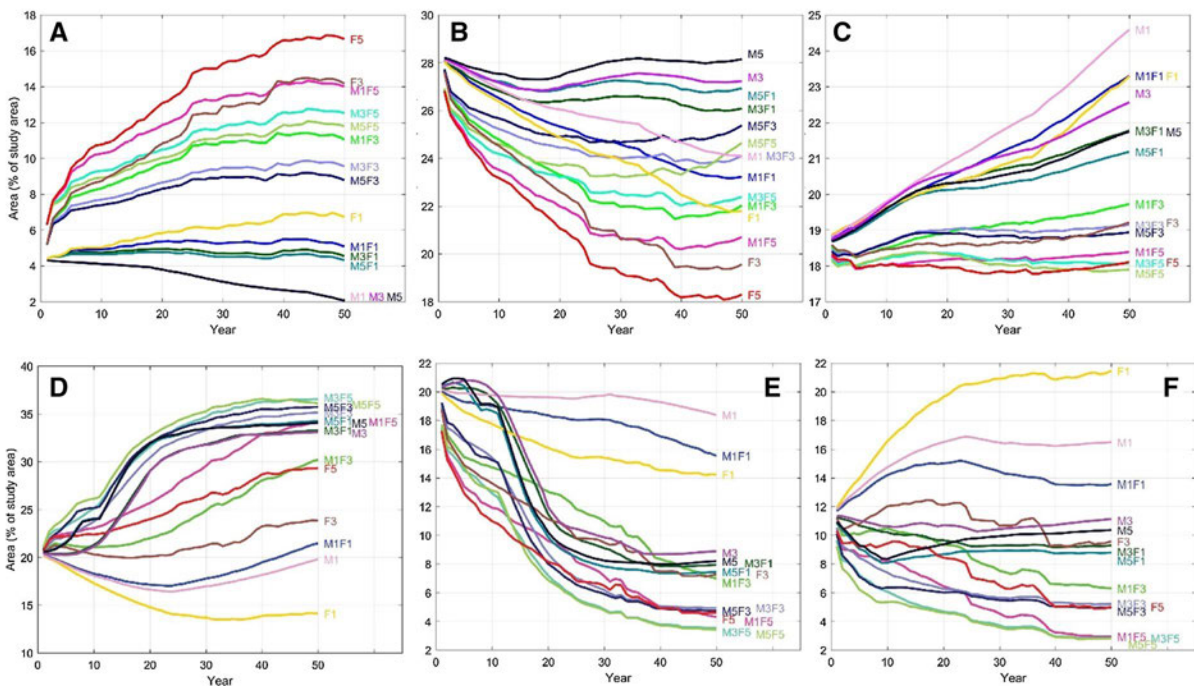


Fig. 9 Change over time in stand structure in terms of tree size for the 15 simulated scenarios: (**A**) early successional, (**B**) small-tree forest, and (**C**) large-tree forest; and canopy cover in (**D**) open, (**E**) moderate and (**F**) closed-canopy forests. Early successional is a post-disturbance state of grasses, shrubs and trees < 12 cm DBH, whereas small-tree and large-tree forests are defined as areas where tree size is > 12 cm and < 37 cm DBH, and > 50 cm DBH, respectively. Canopy closure classes are defined as follows: open ≤ 40%; moderate ≥ 40% and < 60%; and closed ≥ 60%. F = wildfire scenario, M = active management scenario; 1, 3 and 5 indicate 1X, 3X and 5X the contemporary levels of each scenario

lands treatments were the main driver of resiliency, whereas on private lands, increases were related to levels of wildfire, although in the latter the increase was relatively minor. For the study area as a whole, higher ratios of management to fire resulted in higher levels of resilient forest over time. Management decreased resilient forest on private lands because the model replicated the practice of harvesting large diameter trees that are required to meet our definition of resilient forest (Charnley et al. 2017). In contrast, prescriptions modeled on national forest lands had a diameter cap of 53 cm (21 inches) DBH, in accordance with regional policies at the time of the study. The minor increase in resilient forest on private lands likely resulted from large trees surviving low to moderate severity wildfire coupled with a harvesting rate that allowed them to persist on a small portion of the landscape.

Landowner preferences also changed the relative effects of management and fire on high severity fire potential. Treatments reduced potential high severity fire on federal lands, and the addition of fire accelerated this reduction. However, fire in the absence of management increased potential high severity fire over time. By contrast, treatments on private lands coupled with current levels of fire (F1) led to increased potential high severity fire, a direct result of silvicultural practices that create and perpetuate multistory forests prone to crown fire, as well as the lack of prescribed fire and other mechanical methods to reduce activity fuels generated during harvest. A recent study of fire behavior in a nearby location in Oregon found that the 2015 Douglas complex burned at higher severity on private lands where treatments did not include slash removal and underburning, and created younger stand structures that are prone to crowning (Zald and Dunn 2018). Similar findings were reported by Prichard et al. (2020) for the 2014 Carlton Complex fire. Activity fuels generated from harvesting vary significantly among harvesting methods (Stephens et al. 2009) and can be ameliorated from either mastication, prescribed fire, pile burning, or combinations thereof.

Simulating wildfire scenarios that bracketed pre-settlement, contemporary, and predicted future fire regimes in terms of area burned provided a wide spectrum of scenarios to observe fire-forest management interactions and fire feedbacks (Ager et al. 2017). Our maximum fire scenario (5x) had a rotation interval of 34 years (3.3% of the study area per year) or equal to the pre-settlement conditions as reported in Hagmann et al. (2019) for the surrounding ecoregion. In the same study, the contemporary fire rotation (1985 and 2015) was estimated at 400 years, burning at a mere 0.16% per year. This compares to about 0.7%, the annual rate of burning for our contemporary fire and management scenario (F1M1). Barros et al. (2021), modeling climate change in our study area, predicted an increase in mean annual area burned of 19% in the next 50 years. At broader scales, area burned in the western US is expected to increase 2–5 fold in response to a changing climate, although estimates are highly variable among studies and the various general circulation models (GCM) used in them (McKenzie et al. 2004; Lenihan et al. 2008; Littell 2010; Rogers et al. 2011).

Both disturbances had negative effects on aboveground carbon as they were implemented above contemporary levels, although at the highest levels of fire, the carbon loss from management was partially compensated by reduction in wildfire-caused losses (Fig. 8). Note that we were not able to directly estimate avoided wildfire emissions from wildfire from forest management and calculate net carbon under the different scenarios due to limitations in the Envision model. Comparing the effect of management versus fire on aboveground carbon, the effect of fire per unit area (% of the study area) on the reduction in carbon was slightly more than that of active management (Fig. 6C). For the landscape as a whole, increased carbon was correlated with increases in potential high severity fire (Online Resource 1 Fig. A3). Variable results among the 22 modeling studies reported in James et al. (2018) concerning net carbon from forest management are not surprising given the diversity of approaches and assumptions for the key carbon drivers, in particular: (1) fire regime (frequency and intensity); (2) treatment regime; (3) reduction in fire spread rate and intensity from treatments; (4) tree mortality and fuel combustion from forest and fuel treatments; and (5) wood products generated from treatments assumed to be fixed in building products. All of these parameters are highly variable across forested ecosystems and contemporary management, and thus different outcomes are expected. Relatively minor adjustments in modeling parameters can change net carbon from negative to positive as the compensatory action of these parameters plays out over time (Fig. 8). Our diverse scenarios exposed the sensitivity of landscape carbon to the respective disturbances, and suggest that future studies can benefit from simulating a wider range of scenarios, especially with respect to fire regimes, to assess sensitivity of parameters to carbon (Campbell and Ager 2013). Although our model does not consider belowground carbon, a recent review by Mayer et al. (2020) reported that mechanical restoration of the type practiced on federal lands (typically light to medium thinning) generally is neutral to belowground carbon, although there was considerable variation among the studies examined. The extent to which long-term restoration and fuel management can generate positive total carbon (Liang et al. 2018; McCauley et al. 2019) is highly dependent on future burn probabilities that are highly uncertain under current non-stationary fire regimes.

Besides carbon, one key comparison among forest landscape studies is the effect of simulated management on area burned and fire severity. Comparisons with prior studies, primarily conducted with LANDIS-II (e.g., Scheller et al. 2018), are limited in scope because wildfire response metrics are not consistently reported (Krofcheck et al. 2017b; Loehman et al. 2017; Scheller et al. 2018), nor are comparisons made between modeled outputs and observed fire statistics within the study area. O'Donnell et al. (2018) reported a doubling of the fire return interval when management was increased from 2 to 5%, and that management has a stronger effect than climate in terms of reducing area burned and severity. By contrast, Liang et al. (2018) and McCauley et al. (2019) reported no reduction in area burned from management. In the current study we observed a modest increase in the fire rotation interval from 30 to 37 years when treatments were increased from 0 to 2.5% per year. Finney et al. (2007) reported nonlinear reduction in spread rate with increasing treated area, with diminishing treatment effect when the treatment rate exceeded 2% per year. Many studies report a more significant reduction in fire severity compared to area burned (Krofcheck et al. 2017b; Liang et al. 2018; O'Donnell et al. 2018; McCauley et al. 2019), which can be explained by the long time required for modeled fuels to recover from treatment sufficiently to support high severity fire compared to the pretreatment fire spread rate. In general, recovery rates for high fire spread compared to high fire severity (30-100 years) are relatively short (5–20 years)(Prichard et al., 2017), and thus the effects of management on fire spread rates can dissipate prior to subsequent fires.

One common finding from many recent landscape modeling studies is that accelerated management and climate scenarios do not substantially change forest resiliency to wildfire and other attributes for decades in the simulations (Scheller et al. 2018; Liang et al. 2018; Hurteau et al. 2019; McCauley et al. 2019; Ager et al. 2020), partly because it takes a long time for fire resilient structure to develop, and it occupies a minority of the landscape in many of the areas studied. Contemporary wildfire and management disturbance levels replicated in models are relatively mild, perpetuating vegetation states for decades and dampening landscape feedbacks and other higher order interactions compared to many systems modeled with other agent-based urban and rural landscape models (e.g., Valbuena et al. 2010). Institutional factors that control federal forest management, including fire suppression, narrow the possibility for rapid changes in management and actors are relatively few, slow and noninteracting (Spies et al. 2017). Interestingly, the slow response to landscapes calibrated with contemporary disturbance levels provides a stark contrast to the most recent fire seasons in the western US where fires exceeding 100,000 ha are not uncommon (e.g., 2021 Dixie fire; 390,000 ha), creating dramatic changes on large areas of western landscapes, including delayed regeneration and loss of resiliency (Kane et al. 2013; Kemp et al. 2016; Owen et al. 2017; Stevens-Rumann and Morgan 2019). In the current study, our fire model did generate a 1.2 million ha fire in a 5X wildfire scenario that had a substantial influence on the rate of landscape change (Fig. 5, decade 1). The use of average weather and fuels input data to wildfire models generates average fire seasons that are apparently insufficient to generate "megafire" tipping points (Adams 2013) or surprise fires (Hulse et al. 2016) that can dramatically alter forest landscapes. It is interesting to speculate how a single fire event (e.g., Dixie fire) might alter the findings and conclusions about the effect of active management on forest resiliency and carbon reported in landscape modeling studies.

Our study is a precursor to further Envision modeling to examine how RO fire can counteract the adverse effects of climate change on fire regimes, while accounting for fire feedbacks (Ager et al. 2017), patch dynamics (Hessburg et al. 2015), and spatially explicit interactions between treated areas and wildfires (Young et al. 2019; Ager et al. 2021b). We recently incorporated downscaled and spatially explicit energy release component (ERC) (Bradshaw et al. 1983) data to predict and model future fire events (Barros et al. 2021), and built a fire suppression model that simulates RO fires based on weather and proximity to developed areas (Young et al. 2019). Climate-informed agent-based models like Envision will continue to play an important role in understanding how fire excluded multi-owner landscapes can be restored to resilient conditions given specific landowner preferences and feedbacks that shape landscape change and policies (Spies et al. 2018). Forest landscape models can be used to further explore conflicts and opportunities for RO fire in specific admixtures with management on landscapes fragmented with respect to ownership, development, and divergent land management objectives. Future model development can focus on incorporating finer scale vegetation data from Lidar and high-resolution fire spread models to understand how management-fire mixtures can promote multiscale patch scale heterogeneity important for fire resiliency (Moore et al. 1999; Allen et al. 2002; Hessburg et al. 2005; Perry et al. 2011; Cannon et al. 2020; Ritter et al. 2020) under climate change scenarios.

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Data availability Data available from authors upon request.

Code availability Not applicable.

Declarations

Conflict of interest Not applicable.

Ethical approval Not applicable.

Consent to participate Not applicable.

Consent for publication Not applicable.

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