Ecological Modelling 421 (2020) 108962

Contents lists available at ScienceDirect

Ecological Modelling

journal homepage: www.elsevier.com/locate/ecolmodel

Modelling the effect of accelerated forest management on long-term wildfire activity

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ARTICLE INFO

Keywords:
Forest restoration
Forest landscape model
Wildfire simulation
Forest Vegetation Simulator
FSim

ABSTRACT

We integrated a widely used forest growth and management model, the Forest Vegetation Simulator, with the FSim large wildfire simulator to study how management policies affected future wildfire over 50 years on a 1.3 million ha study area comprised of a US national forest and adjacent lands. The model leverages decades of research and development on the respective forest growth and wildfire simulation models, and their integration creates a strategic forest landscape model that has a high degree of transparency in the existing user communities. The study area has been targeted for forest restoration investments in response to wildland fires that are increasingly impacting ecological conditions, conservation areas, amenity values, and surrounding communities. We simulated three alternative spatial investment priorities and three levels of management intensity (area treated) over a 50-year timespan and measured the response in terms of area burned, fire severity, wildland-urban interface exposure and timber production. We found that the backlog of areas in need of restoration on the national forest could be eliminated in 20 years when the treatment rate was elevated to a maximum of 3× the current level. However, higher rates of treatments early in the simulation created a future need to address the rapid buildup of fuels associated with understory shrub and tree regeneration. Restoration treatments over time had a large effect on fire severity, on average reducing potential flame length by up to 26% for the study area within the first 20 years, whereas reductions in area burned were relatively small. Although wildfire contributed to reducing flame length over time, area burned was only 16% of the total disturbed area (managed and burned with prescribed fire) under the 3× management intensity. Interactions among spatial treatment scenarios and treatment intensities were minimal, although inter-annual variability was extreme, with the coefficient of variation in burned area exceeding 200%. We also observed simulated fires that exceeded four times the historically recorded fire size. Fire regime variability has manifold significance since very large fires can homogenize fuels and eliminate clumpy stand structure that historically reduced fire size and maintained landscape resiliency. We discuss specific research needs to better understand future wildfire activity and the relative influence of climate, fuels, fire feedbacks, and management to achieve fire resiliency goals on western US fire frequent forests.

1. Introduction

Forest landscape models (FLMs) are important tools used to address a wide range of forest management policy tradeoffs on public and private forests (Shifley et al., 2017). Here we narrow our definition of FLMs to systems that at a minimum can model forest growth, succession, management, and major disturbances (wildfire, insects, pathogens) at large scales (e.g. >50,000 ha). Several recent studies using FLMs have examined the effects of forest and fuels management on future wildfire activity, carbon, water yield, resiliency, and other forest attributes (Barros et al., 2017; Hurteau et al., 2019; O'Donnell et al., 2018; Spies et al., 2017). Studying longer-term (e.g. >20 years) dynamics between management and disturbances can reveal important ecosystem tipping points, feedbacks, and unintended consequences of management activities (Halofsky et al., 2014; Spies et al., 2014) that are difficult to otherwise detect. Most recently, applications of FLMs have provided insights on the potential effects of management on future fire and forest composition under a range of climate change scenarios (Lucash et al., 2018). Many of these studies in the US have used portions of the large (76 million ha) national forest network as study areas where wildfires are increasingly impacting ecosystem services and burning into adjacent developed areas.

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https://doi.org/10.1016/j.ecolmodel.2020.108962
Received 13 June 2019; Received in revised form 22 January 2020; Accepted 23 January 2020
Available online 25 February 2020
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Despite the growing number of studies, FLMs are substantially underutilized as scenario planning tools to craft forest policy in response to changing climate and disturbances. Existing studies sample a small portion of the physiographic conditions both in the US and globally. Moreover, the inherent complexity of modelling forest and fuel management systems under a highly stochastic background of wildfire activity contributes to contradictory findings among studies. Forest restoration programs on public and private lands in the US are funded in excess of 1 billion USD per year, yet there lacks a definitive set of case studies that identify long-term forest management options and priorities to achieve specific goals. Prioritizing forest management options is a complex tradeoff problem where multiple and conflicting objectives must be balanced under economic and social constraints (Barros et al., 2019; Pohjanmies et al., 2019; Spies et al., 2017). Investments to improve FLM frameworks and conduct further case studies can play an important role in advancing collaborative planning (Butler et al., 2015; Peterson et al., 2003) and pave the way for efficient ecological restoration investment strategies in many forest systems (Noss et al., 2009).

One essential part of advancing the application of FLMs for scenario planning is improving model transparency to stakeholders and other scientists. For instance, some models abstract landscapes into development states, gaps, cohorts, fire regimes and other strata that represent average conditions that may not exist in real landscapes. This simplification has been necessary due to a shortage of detailed inventory information at landscape scales, and computational constraints. However, with the development and broader application of imputation methods, detailed stand-level forest inventory and fuel data are now widely available (LANDFIRE, 2013a; Riley et al., 2016). These new data make it feasible to scale-up well developed, high-resolution forest stand simulation models that leverage decades of development and application by researchers and practitioners. One such model is the Forest Vegetation Simulator (FVS, Crookston and Dixon, 2005), an individual-tree growth model that inputs actual stand inventory data (populations of trees) and simulates a wide range of forest management actions for most major US tree species, forest types and stand conditions. The model can be used to simulate forest stand growth, succession, carbon, insect outbreaks, economics, and stand-scale wildfire. The model has seen wide application in the field for stand growth and management, and significant investments in development (Crookston and Havis, 2002; Havis and Crookston, 2008; Keyser and Keyser, 2017; Rathbun et al., 2012; Teck et al., 1997), but has not yet been leveraged to build a fully functional FLM. The potential to extend FVS to landscapes is made possible with the FVS Parallel Processing Extension (FVS-PPE, Crookston and Stage, 1991) which provides for coordinated landscape modelling by processing stands in parallel, i.e. reading a stand inventory for the landscape, and then pausing each time step to spatially re-prioritize management using internal or external algorithms and data. FVS-PPE was used to dynamically prioritize and constrain landscape management activities to examine management impacts on carbon, old growth, and northern spotted owl (*Strix occidentalis caurina*) habitat (Ager et al., 2007a,b; Ager et al., 2010). Finney et al. (2007) linked FVS-PPE to a spatial treatment optimization algorithm to demonstrate the importance of strategically locating fuel treatments to reduce fire spread. However, none of these studies incorporated wildfire as an endogenous disturbance agent that changed forest conditions over time.

Another modelling system that has not been leveraged for FLMs is the family of fire simulation models developed by USDA Forest Service researchers (e.g., FARSITE, FlamMap, FSim and FSPro) (Ager et al., 2011; Finney, 2006; Finney et al., 2011; Noonan-Wright et al., 2011; Riley et al., 2018). These models are widely used in the US and abroad to model large fire occurrence, spread, intensity, and suppression, and to assess the effects of restoration and fuels management on fire behavior. In particular, the FSim model (Finney et al., 2011) has been used for numerous assessments of wildfire exposure and risk (Haas et al., 2013). Despite the wide application and significant investments in these respective systems, neither has been integrated into an FLM, with one recent exception where the underlying fire behavior code library (Brittain, 2017) was integrated into the Envision agent-based FLM (Ager et al., 2018; Barros et al., 2017; Spies et al., 2017).

In this study, we describe the development and application of a new FLM, LSim, which integrates the aforementioned FVS (Crookston and Dixon, 2005), with FSim (Finney et al., 2011). LSim has the functionality to simulate spatially coordinated forest management over time under a background of large stochastic wildfires using submodels that have undergone decades of field application. This is in contrast to other FLMs that have yet to be used to guide site specific management activities as part of forest and fuels management on national forests. The LSim model provides a platform to simulate detailed prescriptions developed by silviculturalists in the field (e.g. Rathbun et al., 2012) as part of forest landscape management projects. We applied the model to the Deschutes National Forest in central Oregon, USA, to study how accelerated forest restoration might affect future wildfire area burned, fire severity, fire exposure to the wildland urban interface (WUI) and commercial timber production. The Forest has a history of large fires that are increasingly impacting conservation, amenity values, and developed areas, and extensive forest management and restoration programs are underway aimed at reducing fire impacts. We discuss future application of FLMs to address wildfire management and conservation policy issues on public wildlands in the western US.

2. Methods

2.1. Study area

The study area comprises 1,336,176 ha that include a mix of public and private lands composed mostly of the 696,538 ha Deschutes National Forest (DNF) and a 10 km buffer outside the DNF proclaimed boundary that includes lands managed by the Fremont-Winema National Forest, Confederated Tribes of Warm Springs, State of Oregon, and the Bureau of Land Management (Fig. 1). We added a 10 km buffer to eliminate edge effects on simulated wildfires. The DNF is currently partitioned into 18 planning areas (Fig. 1) and treatments within these areas are regulated by the DNF Land and Resource Management Plan (USDA Forest Service, 1990), the Northwest Forest Plan (USDA and USDI, 1994) and PACFISH (Henderson et al., 2005). Stands available for treatment (hereafter, manageable areas) correspond to 55% of the DNF and are within the General Forest Matrix and Deer Habitat designations, while the untreatable stands are primarily designated as Wilderness, Intensive Recreation, and a variety of other special management designations. Private inholdings and WUI in the study area account for 121,000 ha and 63,140 ha, respectively. We used the WUI boundary mapped by the interagency Central Oregon Fire Management Service and the State of Oregon. The WUI is primarily located along the northeast boundary of the DNF and in the central-southern portion of the study area (Fig. 1).

The study area sits along the eastern slope of the Cascade Range in central Oregon with slopes ranging from 0 to 30% and well-drained soils developed on lava plains. Climate is semiarid; mean annual precipitation is about 20 cm and mean annual temperature is about 9 °C. The east-west edaphic and topographic gradients exert a strong influence on vegetation types and distribution. The DNF is diverse and encompasses wet subalpine and mixed conifer mountain forests on the steeper slopes of the Cascades and dry-mixed conifer, dry pine and semi-arid juniper woodlands on more gently sloping topography at a lower elevation. The study area contains extensive forestlands with lodgepole pine (*Pinus contorta*), ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*) and mountain hemlock (*Tsuga mertensiana*) with cooler, wet subalpine forest (*Abies* spp.) to the west, and semi-arid juniper (*Juniperus occidentalis*) woodlands and arid shrublands to the east. About 24% of the study area

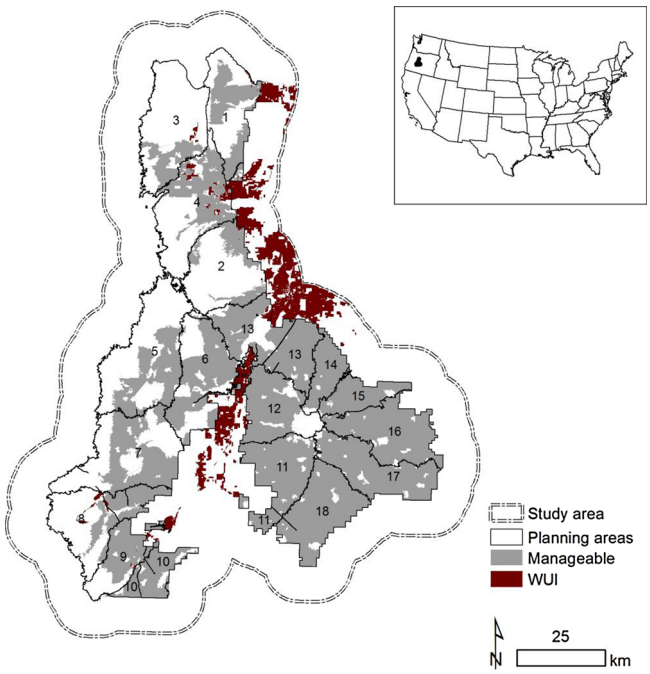

Fig. 1. Study area location on the Deschutes National Forest in central Oregon, USA and planning areas showing the areas available to treat with mechanical harvest under the Forest Land and Resource Management Plan. The majority of treatable stands (manageable) are under the General Forest Matrix and Deer Habitat designations, while the untreatable stands are primarily designated as Wilderness, Intensive Recreation, and other special management designations. Also shown are areas classified as wildland urban interface (WUI).

contains dry mixed conifer forests, with lesser amounts of high elevation forests (15%), ponderosa pine (13%), and wet-mixed conifer (13%) forests. Juniper and lodgepole pine forests combined cover about 18%, with the remaining lands consisting of arid shrub steppe and non-vegetated areas (17%).

Much of the lower elevation forested area has dense understories as a result of fire suppression and partial harvest during the 20th century that removed many of the large fire-resistant ponderosa pine and Douglas-fir, despite an increase over the last decade in thinning and underburning activities aimed to promote fire resiliency (Merschel et al., 2014). Fire history in the area is extensive with an average of 342 ignitions per year between 1992 and 2013, which burned an average of 6689 ha per year (Short, 2017). Although area burned per year is highly variable, ranging from 49 to 48,873 ha, wildfire activity has seen a significant jump in the past decade with several large fires in the study area including the B&B complex fire in 2003 (36,733 ha), Sunnyside Turnoff in 2013 (21,448 ha) and the Pole Creek fire in 2012 (10,843 ha).

2.2. Overview of LSim model components

2.2.1. Forest vegetation modelling system

LSim simulates vegetation succession and management using the Forest Vegetation Simulator (FVS, Crookston 2005). FVS is a distance-independent, individual-tree model that uses empirical growth relationships to model stand development and mortality. The unit of simulation is a homogeneous stand polygon that is described by inventory data on tree density by species and size, along with broader site information including elevation, aspect, and ecotype (Crookston and Dixon, 2005). There are 22 variants of the model that cover all the major forested regions in the US. The model is widely used in research and in operations to design silvicultural treatments (Crookston and Havis, 2002; Havis and Crookston, 2008; Keyser and Keyser, 2017).

Table 1
Types of treatment actions and associated forest types. See Supplementary Appendix A, Table A1, for thresholds used to select stands for each treatment action.

Forest type	Treatment action
All forest	Pre-commercial thinning followed by pile and burn to treat activity fuels
Fire tolerant	Commercial thinning followed by yard tops and prescribed fire
Fire intolerant	Commercial thinning followed by yard tops
Fire tolerant	Prescribed fire to reduce ingrowth and treat natural fuels
Fire intolerant	Yard tops followed by a prescribed burn
Lodgepole pine	Commercial thinning followed by pile and burn

Many FVS extensions have been developed, including stand-scale modelling of carbon, forest economics, fuels, and fire behavior (See Table 1 in Crookston and Dixon (2005)). A geospatial ArcMap interface to FVS is available and provides point-click functionality to execute FVS and view projected inventory data in the stand using the Stand Visualization System within ArcMap (Ager et al., 2011). The source code for FVS is freely available in the public domain and the model has been modified for use in other countries (Robinson and Monserud, 2003).

FVS is executed using an ASCII keyword file that identifies the source file with inventory data, a battery of simulation parameters on the site condition, simulation period and instructions for management actions if any. Although a large number of simulation parameters can be specified for a single stand simulation, the model can be run with less than 10 keyword statements. The keyword file is accompanied by an ASCII file or Microsoft Access database containing inventory data. The inventory file specifies trees per hectare, species, and size for each stratum in the stand. Further details are provided below for specific modelling functionality.

LSim uses a specific version of FVS created by Crookston and Stage (1991) to behave as a landscape model where groups of stands are processed in parallel through time, rather than in a serial fashion where the simulation over time is completed one stand at a time. This version, the Parallel Processing Extension (FVS-PPE), processes a set of stands for a given time period, and then pauses execution to allow for exogenous programs to execute and provide information for the next cycle. In our implementation these programs included spatial algorithms to prioritize stands for management, and fire intensity for stands that burned in a wildfire as described below.

In LSim, existing FVS calculations of tree growth, mortality, and other aspects of forest dynamics were left intact, but several modifications to the underlying FVS-PPE code were necessary to improve performance and scalability. These modifications made it feasible to simulate 50-year scenarios for large landscapes (600,000 ha) in less than an hour. Our modifications were built out of open FVS source code (revision 11/20/13) for FVS-PPE and the Southern Oregon and Northeast California variant of FVS (Keyser, 2008). Modifications to the FVS-PPE included: (1) removing the limit on the number of stands to simulate, (2) multi-threading to allow the use of multiple processors, (3) storing data in RAM to improve between-cycle processing, (4) replacement of the original FVS fuel model logic with a user-defined customized logic, and (5) multi-scale prioritization ranging from stand to planning area level.

2.2.2. Fire modelling

We incorporated a fire simulation submodel by linking in the FSim large fire simulator (Finney et al., 2011). FSim simulates large fire events (in contrast to stand-scale fire behavior modelled in the FVS-FFE) and uses the Minimum Time Travel (MTT) algorithm where rates of fire spread are predicted by semi-empirical fire behavior equations (Rothermel, 1972; Scott and Reinhardt, 2001) and crown fire is modelled after Wagner (1977) and Rothermel (1991). The MTT algorithm calculates fire spread by identifying the shortest travel time among

nodes of a gridded lattice (Finney, 2002). This method minimizes distortion to fire shapes compared to spread models that use cellular automata or contagion algorithms. A binary raster stack is used to store landscape attributes describing surface and canopy fuels and topography (Section 2.3.4; Supplementary Appendix C, Fig. C1). FSim simulates daily fire activity over a fire season using empirical relationships between Energy Release Component (ERC, Finney et al., 2011) and fire activity (probability, spread, containment). ERC has been shown in multiple studies to be a strong driver of fire ignition, spread, and containment (Ager et al., 2018; Finney et al., 2011). Prior to the simulation, relationships between ERC and fire probability are calculated in FireFamily+ (Bradshaw and McCormick, 2000) using empirical weather data obtained from remote automated weather stations (Western Regional Climate Center, 2014) and historical records of area burned (Short, 2015). These estimates are stored in a parameters file (FDist) for FSim. The same program is used to estimate time series parameters to describe seasonal trends, autocorrelation, and daily standard deviation in ERC using daily RAWS weather data (Supplementary Appendix C; Fig. C2). Daily data on wind speed and fuel moisture are generated as well (Finney et al., 2011) (Fig. 3). Wind is modelled in FSim using a joint probability distribution of speed and direction. The data are sampled from the afternoon hours for each month from monthly distributions. These estimates are also stored in the FDist parameters file. At execution, the program iterates though the fire season one day at a time, using the ERC-fire probability model to determine if there is an ignition, and daily weather inputs and fuels in the location of the ignition to determine fire spread. The fire spreads according to daily weather until contained according to a containment model (Supplementary Appendix C; Fig. C3). The containment model predicts effective suppression using historical relationships between containment success and daily ERC values (Finney et al., 2009). Outputs from the fire simulation include a perimeter shapefile and flame length raster grids for the study area.

2.2.3. Integrating FSim and FVS

Integrating FSim and FVS (i.e. LSim) required a wrapper to sequence their execution over time and transfer data between the programs at specified time intervals (Fig. 2). We used five-year intervals instead of annual fire simulations to improve model performance associated with building the fuels landscape files. At each 5-year cycle LSim pauses the vegetation modelling, translates the forest inventory information in FVS to a binary raster stack formatted as required by FSim, and then executes FSim to simulate five seasons of fires (Supplementary Appendix C; Fig. 3). When the wildfire simulation is complete, wildfire behavior for each pixel is passed back to FVS, overlaid with the stand polygon layer, and the inventory is adjusted for wildfire impacts (Figs. 2 and 3). The specific methods used for calculating fire effects on tree mortality are described in Section 2.3.6 below. The execution of FVS is then resumed for the next cycle. This same method was employed in prior studies to calculate tree mortality from landscape fire simulation models (Ager et al., 2010).

2.3. Model parameters

2.3.1. Forest and fuels management

We developed multipurpose stand prescriptions based on management practices on the DNF and implemented them using FVS keywords (Supplementary Appendix A, Table A2). On the bulk of the DNF, management is directed to restore low severity fire regimes by reducing surface and ladder fuels, and retaining large fire resilient ponderosa pine and Douglas-fir. Prescriptions were specific to each of the major cover types on the DNF (Table 1), as determined from forest vegetation maps and supported by empirical studies as effective for reducing potential fire behavior (Stephens and Moghaddas, 2005; Stephens et al., 2009). Simulated treatment actions included different combinations of thinning followed by surface fuel reduction treatment (biomass removal), pile burning or prescribed fire (Supplementary Appendix A, Table A2). Treatment actions were allocated based on cover type and variable species-specific thresholds of trees ha^{-1}, stand density index (SDI), or basal area, time since fire, and fuel loads. Prescribed fire parameters (fuel moistures and wind speed) were chosen to replicate typical fall prescribed burning on the DNF (Supplementary Appendix C, Table C2).

2.3.2. Forest growth and mortality

We modelled forest growth and mortality for each of 48,835 stands using the South Central Oregon and Northeast California variant (SO) of FVS, described in detail in Keyser (2008). Different variants are used to reflect differences in tree growth, mortality, and volume models for different geographical regions. The SO variant uses data from forest inventories, research plots, silviculture stand assessments and tree plantation studies on public and private lands to derive growth relationships for 33 species (Keyser, 2008). In FVS trees are grown in two different ways depending on the tree diameter. For small trees (DBH <7.62 cm) growth is height-driven. Height increments are predicted first for each species followed by diameter growth. Small tree diameter growth is predicted via several species using variant-specific curves that predict diameter as a function of height. By design, in the SO variant western juniper (*Juniperus occidentalis)* trees of all diameters are always grown as small trees.

For large trees (DBH ≥ 7.62 cm) growth is driven by diameter with slightly different approaches depending on the stand location within the SO variant region. For the DNF, diameter increments are predicted based on the periodic change in inside-bark diameter as a function of habitat type; plant association or site index; location; stand aspect, slope and elevation; stand crown competition factor; crown to total tree height ratio; and total basal area for trees that are larger than the subject tree (Dixon, 2002; Keyser, 2008). Height growth of large trees is estimated based on site index curves specific for each species (Keyser, 2008).

Tree mortality in the SO variant reflects background mortality and does not include increases in mortality from insects and pathogens. Modelling tree mortality from wildfire is described in Section 2.3.6. The SO variant estimates background and stand density-related mortality based on SDI. Users can define the maximum SDI based on species, species groups or use the variant default value. Stand density-related mortality occurs when tree density is the primary agent of competition, and mortality begins when stand SDI is $\geq 55\%$ of the maximum SDI threshold for the stand. When density-related mortality occurs, the number of trees that are estimated to die depends on the relationship between the stand SDI and maximum SDI threshold. Background mortality begins when the stand SDI is $<55\%$ of the maximum SDI threshold. The number of trees that die from background mortality is calculated based on an equation that is species-specific and accounts for the length of the cycle (Keyser, 2008). Once the total number of trees dying is estimated from either background or density-related mortality, mortality rates are dispersed to individual tree records. This is done by selecting trees based on the individual tree's percentile in the basal area distribution, adjusted by a species-specific factor that accounts for the species tolerance (Keyser, 2008).

2.3.3. Regeneration

Forest regeneration was modelled using a version of the Blue Mountains Regeneration model (Robinson, 2007) that was modified to include species unique to our study area (Supplementary Appendix B). The regeneration model was developed from extensive forest regeneration survey data obtained from the Wallowa-Whitman National Forest (WAW), La Grande Ranger District. Regeneration models in FVS are typically developed using the Event Monitor extension (Crookston, 1990) and keyword commands that add seedlings to stands after disturbances. The model used logistic regression to predict the probability of regeneration based on site variables (slope, aspect, elevation),

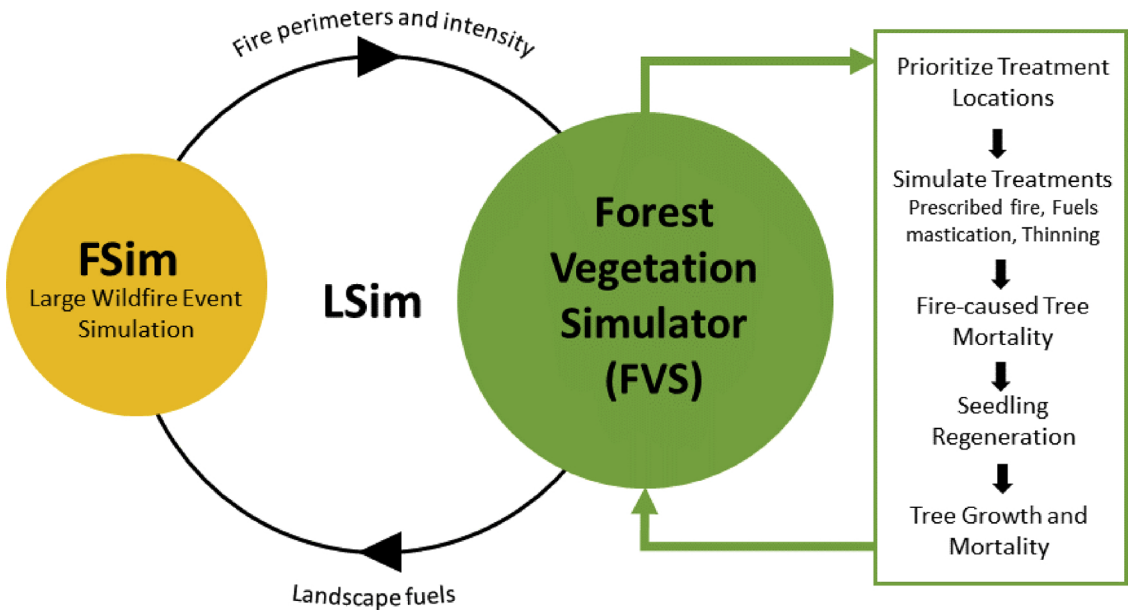

Fig. 2. Diagram showing the major components of the LSim model: large fire event simulator (FSim) coupled with the Forest Vegetation Simulator (FVS). Fire effects are calculated using the Fire and Fuels Extension to FVS (FFE) using fire behavior (flame length) calculated by FSim.

residual overstory canopy cover (%) and basal area (m^2) by species in the stand. The Blue Mountains regeneration model was incorporated into previous studies that examined 50-year management scenarios on the WAW (Ager et al., 2007b). Extensive testing was carried out as part of these latter studies using single stand simulations in FVS and the Stand Visualization System (SVS, McGaughey, 2002). In this process, forest and fuels management was simulated in FVS using a sample of ca. 100 stands with varying conditions in terms of species and basal area. The resulting stand images from SVS over time (30 years) were reviewed by silviculture staff in a subjective evaluation process and it was determined that the model performed well under a range of stand conditions, providing reasonable estimates of regeneration for different stand types and treatments. We adapted the Blue Mountains regeneration model (Supplementary Appendix B) for use on the DNF by cross walking species unique to the DNF with the closest analog in the Blue Mountains (Supplementary Appendix B, Table B1-2). For example, the ponderosa pine equations in the Blue Mountains were also used for sugar pine (*Pinus lambertiana*) regeneration on the DNF. We note that the dominant species (ponderosa pine, Douglas-fir, grand fir) are the same on the two national forests. Regeneration is added to the inventory at each cycle in a process where FVS writes an ASCII output file containing current stand information for species-specific trees ha^{-1},

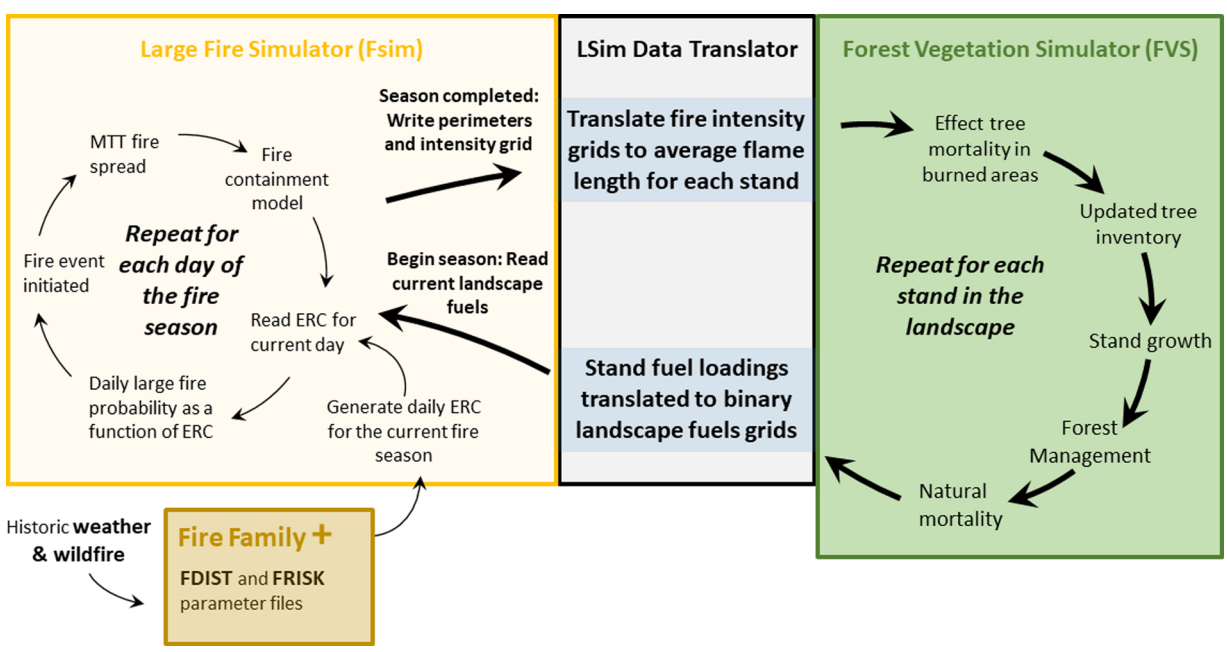

Fig. 3. Detailed diagram of the LSim model components showing the functionality in the two main submodels, the Forest Vegetation Simulator (FVS) and FSim, and the data translator within the parent LSim model. For each simulation cycle, FVS loops through each stand in the landscape to affect fire mortality, growth, management, and natural mortality. The resulting landscape fuels are translated to the binary format read by FSim. FSim then loops through each day in the fire season and simulates fires based on daily predicted weather. At the end of the fire season the wildfire intensity grids are read by FVS and tree mortality is predicted using the functionality in the FVS Fire and Fuels Extension.

basal area and canopy closure and then executes the regeneration model. The latter model computes regeneration counts by species, then writes the information to an ASCII file which is read by FVS and used to update the inventory record for each stand.

2.3.4. Landscape fuel dynamics

Surface fuel succession in FVS is handled by the FVS Fire and Fuels Fire and Fuels Extension (FVS-FFE). Our system deviated from the original FVS-FFE surface fuel model logic to account for temporal live fuel variability, particularly in shrubby lifeforms. On each LSim cycle, fuel models are selected based on live woody load, fuel bed depth, Plant Association Type (Hall, 1998), time since disturbance and canopy cover (Supplementary Appendix A, Fig. A1). Fuel models in the study area were a combination of the original set of thirteen fuel models (Anderson, 1982) and the full set of 40 fuel models developed by Scott and Burgan (2005).

Surface fuel models for the portion of the study area within the DNF were modelled over time using custom fuel model logic (Supplementary Appendix A, Fig. A1) and outputs from processing the national forest inventory data through FVS-FFE. Canopy fuels that described canopy cover, canopy base height, canopy bulk density and total stand height were also obtained from FVS-FFE outputs for initial conditions and through time. Both surface and canopy fuels in the buffer around the study area and inholdings (Fig. 1) were initially assigned with LANDFIRE 2008 rapid refresh FBFM40 data (LANDFIRE, 2013b). Other landscape inputs for fire modelling were also obtained from LANDFIRE and included elevation, slope, and aspect (Krasnow et al., 2009; Rollins, 2009). After discussions with local fuels planners, we chose a timber-litter (TL2) fuel model (Scott and Burgan, 2005) to represent treated stands.

2.3.5. Wildfire calibration

As described above, FSim requires local calibration to model daily fire occurrence and generate synthetic fire weather streams upon which ignition probabilities are estimated. For this process we used a 20 year (1992–2011) weather record from the Lava Butte remote automated weather station (Western Regional Climate Center, 2014) located 5 km south of Bend, OR, and fire history from Short (2015). These data were processed with FireFamily+ (Bradshaw and McCormick, 2000) to generate the FSim parameter files FDist and Frisk (Supplementary Appendix C, Fig. C2).

In this study we assumed random ignition locations within the study area. Although there are some ignition hotspots related to recreation use within the DNF (Ager et al., 2018), and the FSim program can be used with ignition probability grids, we had no basis to assume these hotspots would persist and affect large fire location over the 50-year simulation period.

2.3.6. Wildfire-caused mortality

Wildfire-caused tree mortality was predicted using fire behavior outputs (fireline intensity, FLI) from FSim and the stand fire modelling functionality in FVS. FVS has the capability to model stand-scale fire and predict mortality based on the first-order fire effects model (FOFEM, Reinhardt et al., 2008). The process requires flame length, scorch height and percentage crown fraction burned for each stand (Supplementary Appendix C, Table C1). First, fireline intensity grids from FSim were converted to flame length (m) using Byram's equation (with Wilson (1980) modification):

$$I_B = \frac{I_R}{60}\frac{12.6R}{\sigma}$$

where I_R (kW m^{-1}) is the fireline intensity, R is the surface fire spread rate (m min^{-1}), and σ is the surface area to volume ratio of the fuel bed (m^{-1}).

Scorch height (sHt) was calculated as:

sHt = 3.2808*6.026*pow(FL/3.2808, 1.4466)

We then estimated crown percentage burned based on the scorch height relative to the critical flame length for crowning. We then simulated a fire using FVS and the above parameters. Specifically, we used the FLAMEADJUST keyword to set the fire parameters and simulated a fire with the FVS SIMFIRE keyword. Additional parameters for the fire included windspeed of 30 kph, fuel moistures as outlined in Supplementary Appendix C, Table C2, and air temperature of 23.9 °C. These latter parameters were held constant for each stand. If polygons were partially burned we assumed they burned in entirety using a >50% cutoff as determined by the number of burned pixels.

2.4. Forest management scenarios

We implemented three treatment priorities to prioritize planning areas and select stands for treatment that reflect existing management practices on the DNF. Treatment priorities addressed key management issues related to wildfire threats to the adjacent WUI (DISTWUI), fire resiliency and forest restoration (FIRESEV), and economic potential (ECON) from commercial thinning. The objectives of the treatment priorities (Table 2) were to treat stands: (1) closer to the WUI, (2) with higher potential fire severity, and (3) higher volume, respectively. Distance to the WUI was calculated for each stand based on the shortest distance between the stand's centroid and the nearest WUI polygon. Potential fire severity (versus that from a simulated wildfire in FSim) was estimated for each stand using FVS-FFE's POTFMORT function which returns potential severe fire mortality as a percentage of total basal area assuming a wind speed of 32 km hr^{-1} and fuel moistures of 4%, 4%, 5%, 10%, 15%, 70% for 1-h, 10-h, 100-h, 1000-h, Duff and Live fuels, respectively. Live fuels are separated into two classes for fuel moisture purposes: live woody and live herbaceous. Basal area of trees <53 cm DBH (those that can be harvested based on DNF harvest prescriptions) was the priority metric for ECON and was calculated for each stand based on merchantable volume reported by FVS.

Each treatment priority was designed to test the expected effect on a

Table 2
Treatment priorities, objectives, prioritization metrics, response variables and treatment levels associated with each scenario. Each treatment priority was simulated at three treatment levels, which include current, doubling and tripling the current treatment target on the Deschutes National Forest. The combination of treatment priority and treatment level corresponds to a simulation scenario.

Treatment priority	Objective	Prioritization metric	Response variable	Treatment level	Scenario
DISTWUI	Treat stands closer to WUI	Distance to WUI (km)	Area burned in WUI (ha)	1×	DISTWUI1×
				2×	DISTWUI2×
				3×	DISTWUI3×
FIRESEV	Treat stands with higher potential fire severity	Potential basal area mortality (% of total basal area)	Total area burned (ha)	1×	FIRESEV1×
			Potential flame length (m)	2×	FIRESEV2×
				3×	FIRESEV3×
ECON	Treat stands with higher volume	Basal area (m^2)	Volume harvested (m^3 ha^{-1})	1×	ECON1×
			Volume killed by wildfire (m^3 ha^{-1})	2×	ECON2×
				3×	ECON3×

given response variable (Table 2). Prioritizing distance to WUI was expected to reduce area burned in the WUI. Targeting areas with high potential mortality was expected to both reduce wildfire-caused mortality and potential flame length. Finally, treating stands with the highest basal area was expected to maximize timber volume and reduce volume killed by wildfire. These three prioritization criteria were implemented at three different levels - 36,000 ha per five-year cycle, 72,000 ha per five-year cycle and 108,000 ha per five-year cycle - corresponding to current (1×), doubling (2×) and tripling (3×) the current treatment target on the DNF, respectively. Note that the actual treatment intensity may or may not have met the target dependent on the availability of stands that met the treatment threshold based on management practices on the DNF. The combination of a treatment priority (DISTWUI, FIRESEV, ECON) with a treatment level (1×, 2×, 3×) is referred to as a scenario, resulting in a total of nine simulation scenarios.

2.5. Simulation sequence and response metrics

We simulated scenarios in five-year time steps over 50 years, hereafter referred to as a cycle. The sequence of activities, including management, growth and mortality, regeneration, wildfire, and wildfire effects, is shown in Fig. 2. In each cycle, five years of management followed by five years of wildfire were simulated, the latter as predicted by the FSim model. Each scenario was executed over 30 replicates to capture variability in wildfire impacts.

Response variables associated with each prioritization metric are shown in Table 2. For each scenario, we examined the spatial arrangement of area treated and summarized area treated, area burned, volume harvested and volume killed by wildfire. We quantified cumulative area burned over time in the WUI for each scenario. To provide a measure of potential fire behavior at the landscape scale we calculated potential flame length. The latter describes the expected flame length if the entire study area burned under a predefined set of fire weather. Potential flame length provides a response to the effect of treatments for the whole study area, whereas area burned over time can only capture the effect of treatments when they are intersected by a simulated fire. We calculated potential flame length by post-processing fuels landscapes at the end of each LSim cycle with a command line version of FlamMap (Brittain, 2017) using a southwest wind azimuth and wind speed of 29 km h^{-1}.

3. Results

3.1. Treatment location

Treatments clustered in different locations for each of the three treatment priorities, with FIRESEV treatments allocated along the east and west side of the DNF and ECON mostly focusing on the east side (Fig. 4). The DISTWUI priority resulted in a pattern that initially placed treatments near the WUI but over time spread away from the WUI as closer areas that were eligible to treat decreased (Fig. 5). Thus, the DISTWUI priority generated the desired treatment pattern. Increasing treatment levels led to greater area being treated but maintained the spatial restrictions to treating within planning areas, thus reflecting typical management on real landscapes.

3.2. Area treated

The average area treated for each treatment priority was 0.8% (1×), 1.4% (2×) and 1.4% (3×) of the total land base, i.e., both managed and unmanaged lands (Fig. 6). At the lowest treatment level, this corresponded to seven times more area treated than burned. The latter corresponds to approximately 0.2% of the total land base, with minor variations among priorities and treatment levels. Under the 1× treatment intensity, all scenarios met the annual treatment target

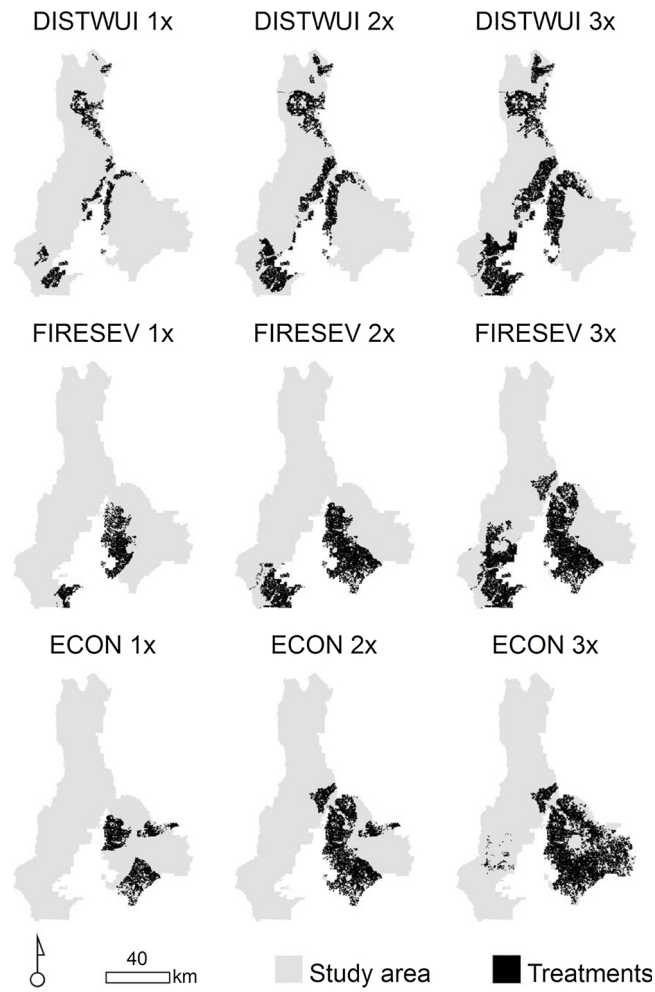

Fig. 4. Example of treatment locations at simulation year five for one replicate and nine scenarios. See Table 2 for scenario descriptions.

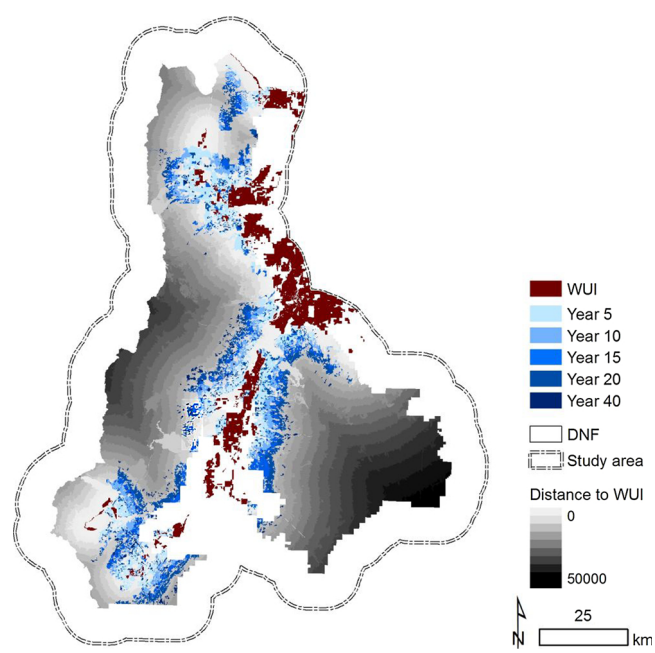

Fig. 5. A sample of realized treatment locations symbolized by simulation year in blue based on the distance to wildland urban interface (DISTWUI) treatment priority for the first 40 years (for interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

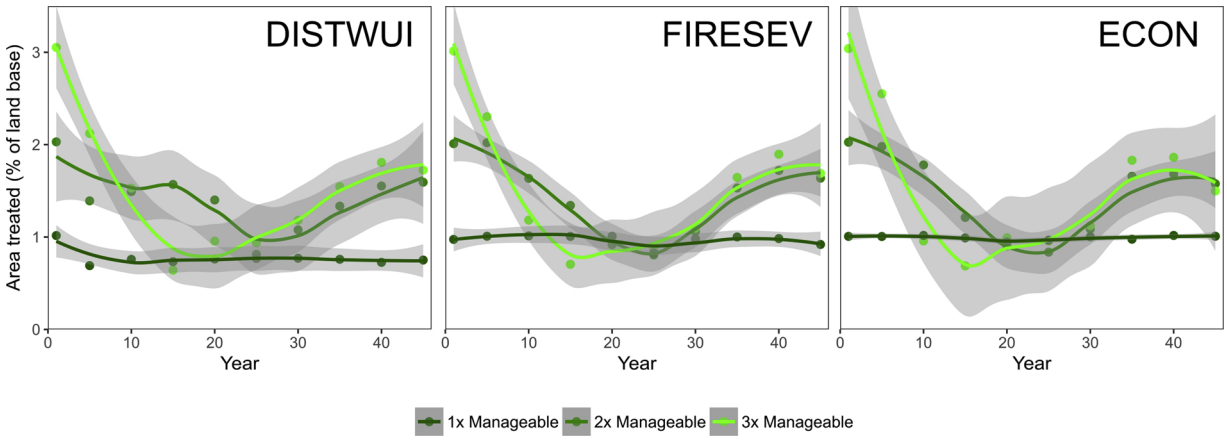

Fig. 6. Mean area treated per year across 30 replicates for scenarios where 1× = 36,000 ha/cycle, 2× = 72,000 ha/cycle and 3× = 108,000 ha/cycle. One cycle corresponds to five years. Grey bands correspond to 95% confidence intervals. See Table 2 for scenario descriptions.

throughout the duration of the simulated 50-year period. However, when treatment levels were doubled and tripled the treatment rates dropped as the area eligible for treatment was reduced below the treatment target. Eligible stands could no longer meet treatment targets around year 25 for all 2× scenarios and approximately 5 years earlier for the 3× scenarios. Similar results showing a shortage of areas to treat under accelerated restoration scenarios were found by Barros et al. (2017) for the same study area, but using a different landscape simulation model.

Simulated area treated varied among replicates within scenario, particularly under higher treatment targets for the ECON priority (Fig. 6). Variability in the area treated among replicates had two causes: (1) the effect of wildfires on stand treatment priorities, and (2) the alteration of landscape conditions over time in relation to the metrics for each of the three specific priorities. Because the spatial distribution of treatments was different among the three metrics the potential for fire-treatment interactions depended on whether a particular scenario concentrated or scattered treatments (Fig. 5).

3.3. Area burned

Variability in annual area burned differed among both replicates and years, and between scenarios. The coefficient of variation (CV) among replicates in total area burned across the simulation period averaged 31% compared to an interannual CV of 223%. Comparison with the area burned from historical fires between 1990–2012 shows similar interannual variability (Fig. 7). The replicates varied in terms of the timing and location of fire events.

The map of fire perimeters by decade during one simulation for the FIRESEV 1x scenario (Fig. 8) showed that fires are generally distributed throughout the landscape and do not show marked change in size over the simulation period despite fuel treatments.

The DISTWUI priority was more effective at reducing area burned in the WUI than alternative scenarios, but differences between DISTWUI and alternative priorities under 1× and 2× were small (Fig. 9). Reduction in area burned in the WUI was more effective under 3×, corresponding to a 10% reduction in area burned.

Percentage area burned for the manageable and non-manageable land base showed a larger effect for the former compared to the latter (Fig. 10). Non-manageable lands were those that were placed into reserves according to the DNF Forest Plan. The reduction in area burned for the non-manageable lands demonstrates a shadow effect of the treatments. It occurred primarily under the FIRESEV and DISTWUI priorities because treatment placement resulted in more treated stands on the east side of the DNF, adjacent to non-manageable lands (Fig. 4).

Temporal trends in area burned were slight or nonexistent over the

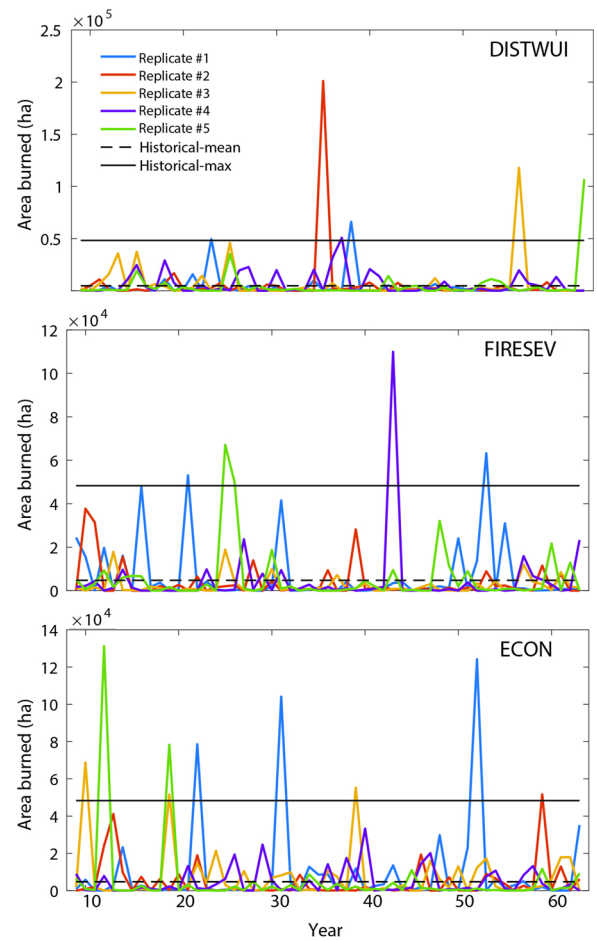

Fig. 7. Variability in area burned (ha) among five simulated replicates for each treatment priority and under the mid-range treatment intensity (2×; 72,000 ha per 5-year cycle). See Table 2 for scenario descriptions. Also shown is the maximum and mean historical area burned per year between 1990 and 2012.

50-year simulation suggesting that at current fire levels the combined changes in vegetation and fuels from succession and management were not sufficient to change overall fire activity within the study area (Fig. 10). However, landscape fire resilience (measured as a decrease in potential flame length) increased on manageable lands (Fig. 11) during the first decade and remained stable for the remainder of the simulation time, whereas in non-manageable lands flame length remained the same throughout the 50-year period. This result is surprising because it

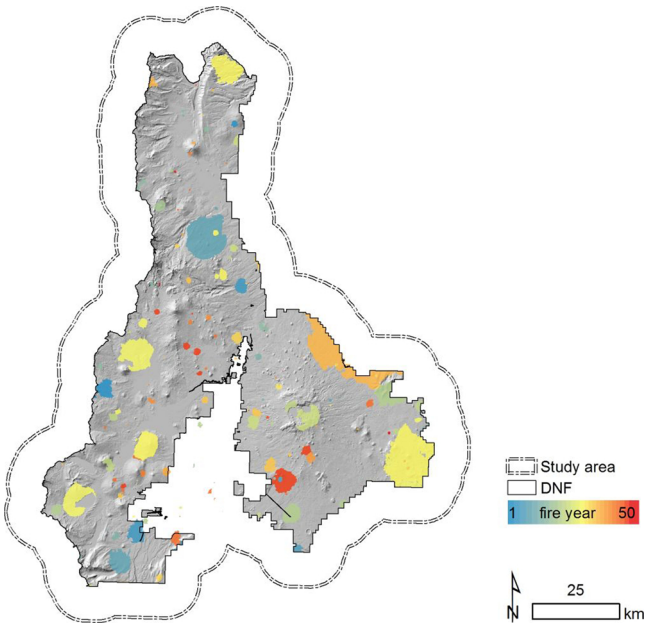

Fig. 8. Map of fire perimeters color-coded by simulated fire year during a single run of the scenario to prioritize stands with higher potential fire severity treating 36,000 ha per five-year cycle (FIRESEV1 ×) on the Deschutes National Forest (DNF).

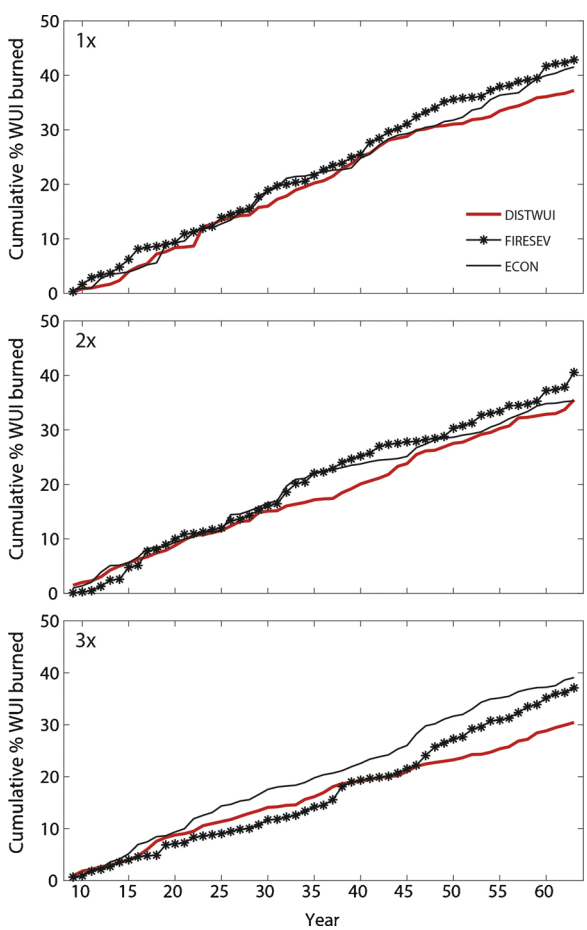

Fig. 9. Cumulative area burned in the wildland urban interface (WUI; as a percentage of the WUI land base) over time, for different treatment priorities and treatment levels. See Table 2 for scenario descriptions.

suggests that vegetation succession and fuel accumulation on non-manageable areas are not contributing to increasing flame lengths. In addition, all prioritization scenarios resulted in the same reduction in flame length on treated lands, contrary to what we hypothesized. This suggests that at comparable treatment levels, the alternative prioritization scenarios achieved landscape conditions that resulted in similar potential flame lengths, allowing for tradeoffs in other landscape services and values.

3.4. Volume harvested and volume killed by fire

Average volume harvested per hectare increased over time and was higher for 1× than for 2× and 3× across all priorities, because thinning out the smaller trees allowed larger trees to continue to grow at an increased rate (Fig. 12). Under 1× more trees were left to grow across the landscape than in 2× and 3×, comparatively. Once a stand was thinned it became unavailable for harvest until the basal area increased, which allowed the remaining trees to grow in the interim. This resulted in increased volume over time as stands that were thinned became available for harvest. The DISTWUI priority showed a more stable harvest volume per hectare throughout the trajectory because it did not target high-volume stands specifically, so stands that were added to meet the area target had an increasingly lower stand density in the FIRESEV and ECON priorities.

Standing merchantable volume killed by wildfire increased over time in non-manageable areas (Fig. 13). In treatable areas volume killed by fire on a per hectare basis was more or less constant with a slight increase in year 30.

4. Discussion

We found that accelerating forest and fuels management led to non-linear changes in landscape conditions that stabilized after 20–30 years when the backlog of areas to treat under current thresholds was eliminated. However, accelerating treatments created a pulse of early seral forests that required treatments later in the simulation. Management treatments had a strong effect on potential fire intentsity, reducing potential flame length for the entire study area on averge by up to 26% within the first 20 years. By contrast, reductions in fire activity as measured by area burned were relatively small, which was surprising given that we assumed simulated treatments would substantially reduce fuels and fire spread. The lack of effect is partly because 60% of the DNF is in reserves where mechanical treatments are prohibited, thus ensuring ample fuels for new ignitions to burn without encountering either treatments or past fire footprints where fuels are reduced. We did note, however, that fire activity on non-manageable areas was affected by treatment intensity in the manageable areas, thus creating a spillover effect between adjacent treated and non-treated landscapes. Large reserves occur on all western US national forests covering between 30-60% of their total area and include wilderness, roadless, and other conservation and amenity reserves where mechanical management is prohibited. Another explanation is that the re-treatment cycle relative to the rate of fuels succession was inadequate to maintain landscapes with low to moderate spread rates. Rapid succession of herbaceous and brush fuels within the study area has been well documented (Dyrness and Youngberg, 1966) as well as increased flammability from needle drape on bitterbrush (*Purshia tridentata*).

A key observation from the study is the substantial inter-annual variability in area burned, illustrating the possibility of alternative future landscape conditions under the same management policy. The interannual coefficient of variation (CV) exceeded 200% and was almost 10-fold larger than the inter-replicate CV. Our estimate from simulation was very close to inter-annual variability in the historical record (1992–2009; 236%). We expected lower inter-replicate

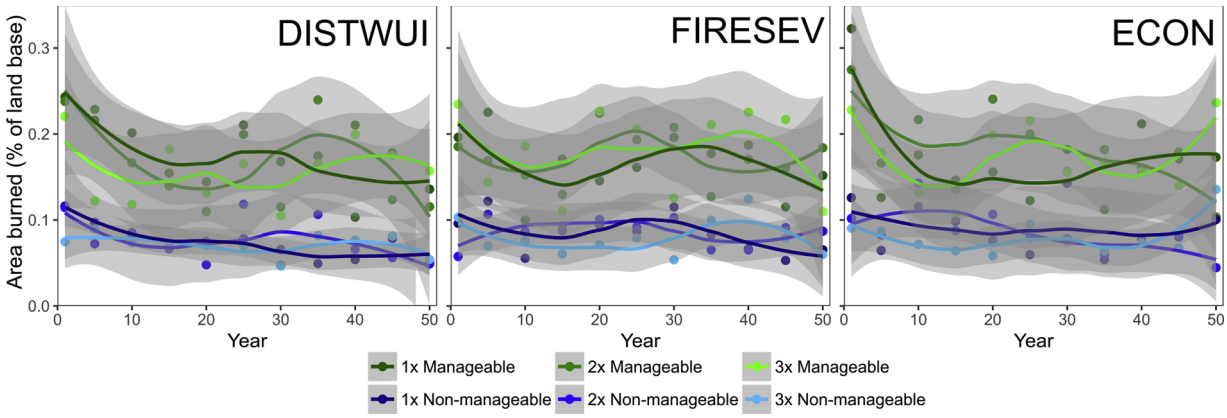

Fig. 10. Mean annual area burned (as a percentage of the land base) across 30 replicates for different treatment priorities and treatment levels on manageable and non-manageable lands. Grey bands correspond to 95% confidence intervals. See Table 2 for scenario descriptions.

variability since variation in area burned is averaged over the simulation period for each replicate. The importance of variability versus the mean area burned is often overlooked and is important since landscape response to fire events is nonlinear with fire size. High variation in area burned can accelerate landscape change and set the stage for larger future fires via fuel homogenization (Hessburg et al., 2016). In other words, resilient forests can absorb the effects of many small fires better than fewer larger ones (Churchill et al., 2013), and presumably this is why clumpy spatial structure in fire-prone forests is a key factor in resiliency. Thus, inter-annual variance in fire size is perhaps as important as the mean area burned given similar intensities in terms of landscape change. The substantial inter-annual and inter-replicate variability we observed in the model is not often reported nor observed in other landscape simulation studies.

Differences among the spatial treatment strategies were minor compared to treatment intensity, meaning that location was less important than area treated. This finding was most surprising for the WUI scenario where treatments targeted areas near WUI. Over the simulation period we estimated that fires will burn through 30–40% of the total WUI area including re-burning. WUI area burned for the WUI scenario was reduced from 35 to 28% between the 1× and 3× management strategies, a small difference compared to simulation studies with static landscapes and similar focus on treatments (Ager et al., 2016; Scott et al., 2016). The lack of differences was in part caused by overlapping treatment locations among the spatial strategies (Fig. 4) where over time project areas rotate through much of the same area.

This study resulted in a new landscape modelling system that leverages decades of Forest Service research and development. By virtue of its coupling with FVS, the model has immediate potential applications in a range of studies concerning the combined and independent effects of fire and management on fire regimes, feedbacks, stand structure, forest patch size distributions, carbon, wildfire, habitat, amenity protection, and economics. The model can be adapted to any forested region in the US using the 20 existing FVS extensions. The LSim model is not limited to small landscapes (e.g. thousands of hectares) as reported for the underlying FVS-PPE in a recent review of FLMs (Shifley et al., 2017). LSim provides a landscape framework to replicate detailed silvicultural modelling as provided for in the FVS prescription keywords. The wildfire simulation submodel (FSim) is widely used for planning and risk assessment in the US (Haas et al., 2013). Integrating existing models like FVS and FSim that have respective user communities and prior management application helps convey a high degree of transparency to agency managers and scientists, which is a key step to further advance the application of forest landscape models for ecological restoration (Spies et al., 2014). Enhancements to LSim includes fire weather that incorporates projected climate change using Energy Release Component data from future climate scenarios projected by the global circulation models in the Coupled Model Intercomparison Project 5 (Abatzoglou and Brown, 2012). Climate-induced changes in vegetation can also be modelled with LSim via the FVS Climate Change Extension (Crookston, 2014; Crookston et al., 2010).

One key advantage of LSim over perhaps all other FLMs is the assimilation of the FVS economics extension (Martin, 2009) which provides for detailed financial analyses of wood products and revenues from forest and fuels management. Economic factors are the primary limiting factor in ecological restoration programs in the western US (Barbour et al., 2008; Stephens et al., 2016) due to low value of wood products in many areas with overstocked stands and high fire risk (Ager et al., 2017). Estimating revenue from logging activities requires simulating the harvesting process where trees are bucked into logs, and

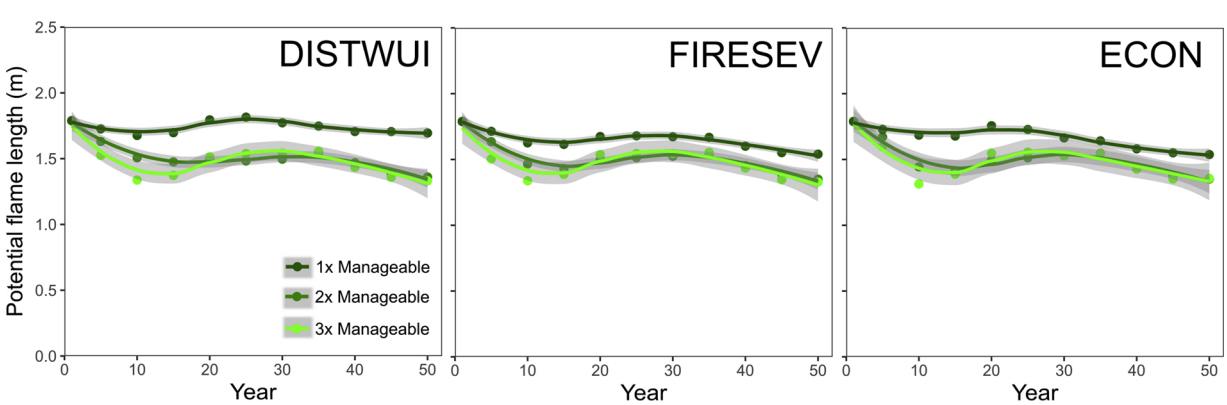

Fig. 11. Mean potential flame length per year across 30 replicates for different treatment priorities and treatment levels. Grey bands correspond to 95% confidence intervals. See Table 2 for scenario descriptions.

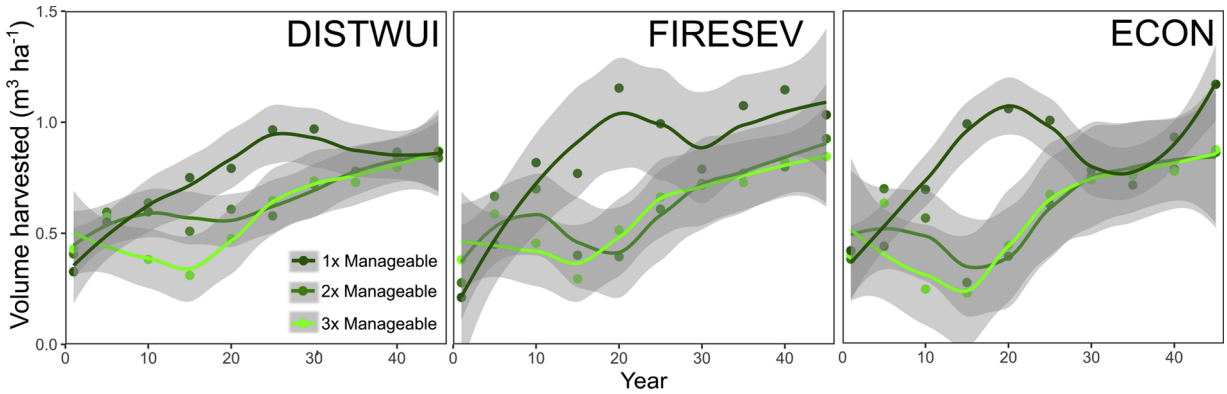

Fig. 12. Mean volume harvested per year across 30 replicates for different treatment priorities and treatment levels. Grey bands correspond to 95% confidence intervals. See Table 2 for scenario descriptions.

each log is valued based on species and small end diameter (Martin, 2009). Log values by species and size are available from local and national reporting agencies (BBER, 2019b). Likewise, logging, fuel treatment, and hauling costs are widely available from empirical data or harvest cost models (BBER, 2019a; Fried et al., 2016). Net revenue from management activities can be calculated by subtracting harvesting and hauling costs from log values. Wider application of financial analyses to prioritize restoration treatments can help improve the efficiency of restoration efforts by pinpointing where timber revenues are positive and treatments are needed to reduce risks and hazards (Ager et al., 2017). For instance, our results showed that higher levels of harvest activities (i.e., 3x) resulted in as much as a 150% drop in the volume per hectare of harvested material compared with lower levels of harvest (Fig. 12), meaning that areas that could potentially pay for high cost restoration treatments (e.g. fuel mastication) rapidly diminish as area treated increases. These results suggest a non-linear cost of restoration as management activities are accelerated in terms of area treated.

Landscape modelling of forest management activities on US public forestlands is a complicated multiscale prioritization problem that is only approximated in other landscape models. For instance, it is possible that some studies allocate treatment in a given time step throughout the study area (e.g. Syphard et al., 2011) rather than our process of replicating the current practice of implementing planning areas (5000–30,000 ha) one at a time and sequencing the planning areas according to management priorities. Stand-level prescriptions are also complex in that they are typically multi-year sequences of mechanical thinning, surface fuels mastication, piling, and prescribed fire, all tuned to local stand structure, species composition, and fire regime to meet multiple forest health, resiliency, and fire management objectives (Churchill et al., 2013; Cochran et al., 1994; Haugo et al., 2015; O'Hara et al., 2010). All of these activities are governed by site-specific directions in forest plans that specify a wide range of forest management constraints (Ager et al., 2016; Ringo et al., 2015). Studies that simulate decades of prescribed fire as the only landscape treatment (Hurteau et al., 2016; Krofcheck et al., 2017) are unrealistic on national forests where federal laws mandate commercial harvest to fund restoration and contribute to rural economies. Moreover, prescribed fire has both operational constraints and highly variable effects on tree mortality, including impacts on large trees, which makes it difficult to schedule on large landscapes over time.

Further advances in landscape modelling and additional case studies will provide information that can help landowners and managers understand how divergent management intensities and spatial strategies can potentially change current wildfire trajectories over time. These models can also be used to reinforce the uncertainty associated with wildfire management policies by illustrating how highly stochastic wildfire events can mask progress towards a management goal. Enhancing these models to include agent-based decision frameworks (Spies et al., 2014) and climate change (Hulse et al., 2016), can contribute to improved risk governance systems by disentangling the relative effects of climate, succession, management, and wildfires in fire-prone forest systems.

Declaration of Competing Interest

The authors declare that they have no conflict of interest.

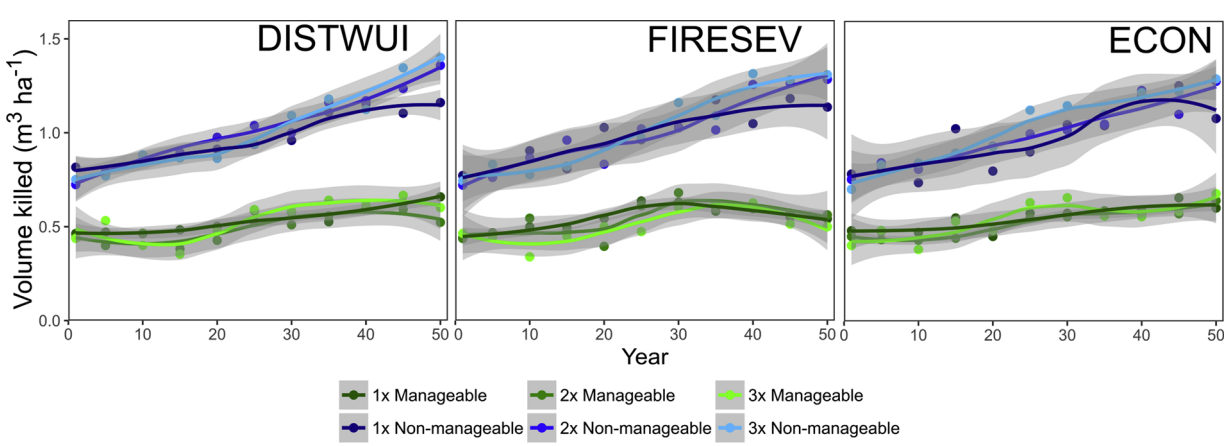

Fig. 13. Mean standing merchantable volume killed by wildfire per year across 30 replicates for different treatment priorities and treatment levels on manageable and non-manageable lands. Grey bands correspond to 95% confidence intervals. See Table 2 for scenario descriptions.

Acknowledgments

This research was supported by the US Forest Service, Rocky Mountain Research Station, National Fire Decision Support Center. We thank Luke Sheivlin, GreenWave Professional, LLC, Missoula, MT for programming support and two anonymous reviewers for their time and insightful comments.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at 10.1016/j.ecolmodel.2020.108962.

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