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Allowing a wildfire to burn: estimating the effect on future fire suppression costs

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Abstract. Where a legacy of aggressive wildland fire suppression has left forests in need of fuel reduction, allowing wildland fire to burn may provide fuel treatment benefits, thereby reducing suppression costs from subsequent fires. The least-cost-plus-net-value-change model of wildland fire economics includes benefits of wildfire in a framework for evaluating suppression options. In this study, we estimated one component of that benefit – the expected present value of the reduction in suppression costs for subsequent fires arising from the fuel treatment effect of a current fire. To that end, we employed Monte Carlo methods to generate a set of scenarios for subsequent fire ignition and weather events, which are referred to as sample paths, for a study area in central Oregon. We simulated fire on the landscape over a 100-year time horizon using existing models of fire behaviour, vegetation and fuels development, and suppression effectiveness, and we estimated suppression costs using an existing suppression cost model. Our estimates suggest that the potential cost savings may be substantial. Further research is needed to estimate the full least-cost-plus-net-value-change model. This line of research will extend the set of tools available for developing wildfire management plans for forested landscapes.

Additional keywords: bio-economic modelling, forest economics, forest fire policy, wildland fire management.

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Introduction

For most of the last century, federal forest fire policy in the United States has been one of aggressive suppression of all wildfire as rapidly as possible. Forest fire suppression expenditures by the USDA Forest Service were reimbursed under the Forest Fires Emergency Act of 1908 and, hence, there was no effective budget constraint. The Great Fire of 1910, which burned over 3×10^6 acres (1.2×10^6 ha) in Washington, Idaho and Montana and took more than 80 lives, lent urgency to the fight against wildfire; in fact, the public attitude became one of 'righteous war' in which 'fire was *the* enemy' (Carle 2002, p. 19).

But opposition to this policy and support for a policy of 'light burning' simmered in the background. Fire ecologists argued that wildfire can play an important role in maintaining healthy forests in fire-adapted forest ecosystems (H. Biswell keynote speech, cited in Carle 2002). This is especially true in dry ponderosa pine (*Pinus ponderosa*) forests, where frequent, low-intensity, low-severity wildfires were common in the pre-suppression era (Everett *et al.* 2000). In addition to favouring fire-adapted species, such as ponderosa pine, these frequent wildfires removed surface fuels and the ladder fuels that can carry fire into the forest canopy where it is more likely to kill trees (Weaver 1943; Pollet and Omi 2002).

In the 1970s, fire policymakers began to acknowledge the fact that decades of successful wildfire suppression had driven forest conditions in the western United States well outside their natural range. In 1978, the 'suppress at all costs' policy was officially abandoned and the use of managed wildfire for fuel reduction was allowed; this policy change has been repeatedly refined, with the most recent version (the 2009 reinterpretation of the 2003 'Interagency Strategy for the Implementation of Federal Wildland Fire Management Policy') providing clarification and flexibility for fire managers to use wildland fire to achieve forest management objectives (Lasko 2010).

Nonetheless, massive accumulation of forest fire fuels (downed woody debris and dead standing trees) and changes in the species composition and forest structure create conditions in which wildfire, when it does occur, is far more likely than in the past to display extreme behaviour over a greater extent. Larger, high-severity fires are more costly both in terms of suppression costs and in terms of risk to ecological and resource-use values (Calkin *et al.* 2005). For example, average annual USDA Forest Service expenditure on fire suppression since 2000 is three times what it was in the previous three decades (Abt *et al.* 2009). Climate change projections indicate that the weather conditions under which the largest, most expensive fires occur are likely to become more prevalent, which lends urgency to efforts to restore forests to a more fire-resilient state (Brown *et al.* 2004).

The Fire Regime Condition Class system currently in use defines three categories to classify landscapes that (1) vary only slightly from the natural range of variation, (2) depart moderately from the natural range of variation or (3) have fire regimes and vegetation attributes that have been substantially altered from their historical range and high risk of losing key ecosystem components (Barrett *et al.* 2010). Today, nearly 4×10^7 ha of federal land, administered by the USDI Bureau of Land Management and USDA Forest Service, fall in the third category and are high priority for restoration (Schmidt *et al.* 2002).

Restoration objectives can be achieved with restoration thinning, mechanical removal of accumulated fuels, prescribed burning and other means. There is a substantial amount of literature that explores the effectiveness of these methods, individually and in combination, in meeting the goal of altering fire behaviour at the stand level (Pollet and Omi 2002; Agee and Skinner 2005; Hudak *et al.* 2011). Landscape-level planning requires that researchers also begin to account for spatial relationships between treated and untreated stands, which may be contingent on treatment methods (Stratton 2004; Finney 2007; Wei 2012). Finally, because fuel treatment is costly (Donovan and Brown 2007), there is a growing literature that explores cost-effective placement of fuel treatments on the landscape (Calkin and Gebert 2006; Hartsough *et al.* 2008; Huggett *et al.* 2008; Rummer 2008).

Fuel treatment is one set of activities that might replicate the restorative function that frequent light burning served in the past, but costs limit the speed at which these activities can be carried out. Conditional use of wildland fire, either instead of or in combination with fuel treatment, might provide a means of achieving restoration objectives more cost-effectively than with fuel treatment alone (Miller 2003; Kauffman 2004). However, although allowing a wildfire to burn may yield positive benefits (including beneficial changes to wildlife habitat, removal of diseased material, and reductions in fire hazard and suppression costs for subsequent fires), it also poses risk of damage (such as destruction of wildlife habitat, timber, structures and human life). It is important to weigh the potential costs and benefits when considering when to allow a wildfire to burn.

The least-cost-plus-loss model first proposed by Sparhawk (1925) for analysing optimal fire suppression expenditure neglected the possibility of beneficial wildfire effects (Baumgartner and Simard 1982). Althaus and Mills (1982) included these benefits in the model by replacing 'loss' with 'net-value-change' and Donovan and Brown (2005) applied it to demonstrate an analysis of wildfire benefits.

In this study, we developed the least-cost-plus-net-value-change model as a conceptual framework for evaluating fire suppression options. We then developed a modelling platform that allowed us to simulate sequences of fires with evolving vegetation on a landscape over time. We applied the simulation platform to estimate one component of net-value-change from allowing a wildfire to burn, the expected reduction in the present value of future suppression costs, for a study area in the south-eastern portion of the Deschutes National Forest in central Oregon. We used Monte Carlo methods to generate a sample of possible scenarios for subsequent fire ignition and weather events. Monte Carlo methods are useful for estimating expected outcomes when there is uncertainty about the inputs to a complex process with many interactions (Kalos and Whitlock 2008). In our analysis, we generated a sample of fire ignitions and concurrent weather from historical frequencies. We combined models of fire suppression effectiveness (Finney *et al.* 2009), wildfire behaviour (Finney 1998) and vegetation development (Dixon 2002) to simulate each future scenario with and without suppression of a fire of interest in the current period under the assumption that subsequent fires would be treated with full suppression effort. We applied a suppression cost model (Gebert *et al.* 2007) to estimate the change in the expected present value of suppression costs for subsequent fires.

In two related applications of Monte Carlo methods to fire behaviour using FARSITE, Ager *et al.* (2010) used Monte Carlo realisations of ignition locations for a given weather stream to estimate burn probabilities across the landscape under typical severe fire weather; Finney *et al.* (2011) used Monte Carlo realisations of short-term future weather conditions to generate burn probabilities across a landscape for a known ignition or fire perimeter, and compared the results with known historical fire perimeters. In our application, the attributes both of ignitions and weather in any fire season are uncertain.

A least-cost-plus-net-value-change model is developed in the next section as a theoretical framework for the analysis. In the third section, we describe the modelling platform that we developed and the methods by which we estimated the expected present value of future fire suppression cost savings arising from the fuel treatment effect of a current fire for our study area in the Deschutes National Forest. Results are presented and discussed in the fourth section. The paper concludes with a discussion of the implications of our results and prospects for carrying this research further.

Theoretical framework

Although we estimate only one component of net-value-change (suppression cost reductions for subsequent fires), we frame the problem in this section as an optimisation in which a fire manager chooses to allow a fire to burn in the current period if the net-value-change is positive. We refer to this fire as the *fire of interest.* Although the simulation model we develop does not allow us to solve the optimisation problem, it lays the groundwork for extending the analysis in that direction in the future and it allows us to interpret our results in the context of a planning environment.

The fire of interest occurs at time $t = 0$. It is an ignition, either a lightning strike or a human-caused fire, that would spread in the absence of suppression effort. It is possible for more than one ignition to occur at time $t = 0$, in which case they are treated as a single event. Let x_0 be a dichotomous variable: $x_0 = 0$ if the fire of interest is allowed to burn unsuppressed and $x_0 = 1$ if not. For this study, we assume that subsequent fires will be treated with full suppression effort and we evaluate potential suppression cost savings resulting from the current fire of interest. That is, $x_t = 1$ for $t = 1,\ldots,T$. We plan to relax this assumption in future research once we develop a full model of net-value-change and can adjust the policy for subsequent fires in a meaningful way. We also hope to extend the choice set to include a wider range of fire suppression options, including partial containment and strategic placement of fuel treatments on a landscape.

We define variables as follows:

- s_t is a vector of state variables describing the landscape at time t. Variables include aspect, elevation, slope and vegetation; s_0 describes the initial landscape, in which the fire of interest occurs. The landscape evolves over time so that $s_{t+1} = S(s_t, w_t, x_t)$ in each time period $t = 0,\ldots, T-1$. $S(s_t, w_t, x_t)$ is a model of state transitions and represents the effect of fire and the subsequent development of fuel and vegetation on the landscape.
- $\boldsymbol{w}$ is a set of random variables $(w_0, w_1, ..., w_{T-1})$ that drive fire behaviour during each time period $t = 0,\ldots, T-1$. This includes the location and timing of ignitions and the weather that occurs over the course of the fire season. The information describing a particular ignition in time period t, w_t, is known at time t.
- $r(s_t, w_t, x_t)$ is the value generated on the landscape in time period $t = 1,\ldots,T-1$.
- $c(s_t, w_t, x_t)$ is the cost of suppression in time period $t = 1,\ldots, T-1$. If $x_0 = 0$, $c(s_0, w_0, x_0) = 0$.
- $V_T(s_t)$ is the value of the landscape at the end of the time horizon.
- i is the real discount rate at which future costs and revenues are discounted to the present using the discount factor e^{-it}.

In the complete optimisation problem, the fire manager chooses x_0 to maximise the net present value of the forested landscape (v) on which the fire occurs over the time horizon ($t = 0,\ldots,T$) defined as:

$$v(s_0, \boldsymbol{w}, \boldsymbol{x}) = \sum_{t=0}^{T-1} e^{-it}[r(s_t, w_t, x_t) - c(s_t, w_t, x_t)] + e^{-iT} V_T(s_T) \tag{1}$$

A rational land manager, facing the dichotomous choice that we pose, would choose to allow a fire of interest to burn rather than suppress it if the net-value-change was positive, so that:

$$\Delta v = v(s_0, w, x|x_0 = 0) - v(s_0, w, x|x_0 = 1) > 2 \tag{2}$$

Splitting Δv into its component parts yields:

$$\begin{aligned}\Delta v = {} & [c(s_0, w_0, x_0 = 1) - (r(s_0, w_0, x_0 = 0) - r(s_0, w_0, x_0 = 1))] \\ & + \left[\sum_{t=1}^{T-1} e^{-it}\{r(s_t, w_t, x_t|x_0 = 0) - r(s_t, w_t, x_t|x_0 = 1)\}\right] \\ & - \left[\sum_{t=1}^{T-1} e^{-it}\{c(s_t, w_t, x_t|x_0 = 0) - c(s_t, w_t, x_t|x_0 = 1)\}\right] \\ & + [e^{-iT}(V_T(s_T|x_0 = 0) - V_T(s_T|x_0 = 1))]\end{aligned} \tag{3}$$

The first term in brackets is the difference in value occurring in the current period ($t = 0$) as a consequence of allowing the fire of interest to burn rather than be suppressed. This will be positive if the avoided suppression cost exceeds the additional loss to fire in the current period. The second term in brackets is the change in the present value of benefits from the landscape in future periods as a consequence of allowing the fire of interest to burn. It will be positive if the fuel treatment provided by the fire of interest reduces loss in subsequent fires. The third term is the change in the present value of suppression costs from fire in future periods from allowing the fire of interest to burn. It contributes positively to Δv if the fuel treatment provided by allowing the fire of interest to burn causes subsequent fires to be less costly to contain. The last term is the change in the value of the ending landscape as a consequence of allowing the fire of interest to burn.

The third term (in brackets), the reduction in the present value of suppression costs for subsequent fires from allowing the fire of interest to burn (assuming subsequent fires will be suppressed), is the focus of this analysis. We denote it as:

$$B(s_0) = -\sum_{t=1}^{T-1} e^{-it}\{c(s_t, w_t, x_t|x_0 = 0) - c(s_t, w_t, x_t|x_0 = 1)\} \tag{4}$$

We denote the present value of future suppression cost savings for a particular fire of interest (m) as $B^m(s_0, w_0^m)$ where w_0^m represents the realised attributes of that fire (location and timing of ignition and the weather leading up to it) that are known at time $t = 0$. We estimated its expected value by simulating N sample paths, which we denote as w_t^{mn} for $t = 1,\ldots, T-1$ for the nth sample path, and computing the average over the sample:

$$\begin{aligned}E[B^m(s_0, w_0^m)] = {} & -N^{-1} \sum_{n=1}^{N} \\ & \sum_{t=1}^{T-1} e^{-it}\left\{c(s_t, w_t^{mn}, x_t|x_0 = 0) - c(s_t, w_t^{mn}, x_t|x_0 = 1)\right\}\end{aligned} \tag{5}$$

A sample path is a particular realisation of w_t^{mn} for $t = 1,\ldots, T-1$; it represents one scenario for future fire ignitions and weather.

Likewise, we generated an estimate of the expected present value of $B(s_0)$, the future suppression cost savings for a landscape (s_0) before w_0^m is realised, by computing the average across the expected value of all $m = 1,\ldots,M$ fires of interest:

$$E[B(s_0)] = M^{(-1)} \sum_{m=1}^{M} E[B^m(s_0, w_0^m)] \tag{6}$$

Data and methods

We developed a simulation platform for our analysis with the following components: a procedure to draw a set of sample paths from historical frequency distributions of ignitions and weather, an existing simulation model of fire spread and crown fire, a state-and-transition model developed from simulations of vegetation development and fire effects using an existing vegetation simulation model, an existing model of fire duration, and an existing econometric model of large fire suppression costs.

These components are described below. We used this platform to estimate potential future fire suppression cost savings as follows. We started with an initial landscape (s_0) that includes the state variables that drive fire behaviour – topography, surface fuel and attributes of the canopy fuels. We then developed a set of M fires of interest, which occur at $t = 0$. These fires of interest are represented by w_0^m, which includes the stochastic variables that drive fire behaviour – ignitions, weather and fire duration. For each fire of interest, we developed a set of N sample paths, represented by w_t^{mn}, $t = 1, \ldots, T-1$, that includes the same stochastic variables as the fire of interest, realised for all subsequent fires. With that in hand, the procedure to compute $E[B^m(s_0, w_0^m)]$ for the mth fire of interest is:

- For each sample path, $n = 1, \ldots, N$.
- For each value of $x_0 = 0,1$.
- For each time period, $t = 0, \ldots, T-1$.

(1) Simulate fire for given s_t and w_t^{mn}.
(2) For each 30-m^2 plot of land, or pixel, record if there was crown fire, surface fire or no fire.
(3) Update the surface and canopy fuel state variables for each pixel according to $s_{t+1} = S(s_t, w_t, x_t)$.
(4) Compute area burned by fire type and compute discounted suppression cost for suppressed fires ($e^{-it}c(s_t, w_t, x_t)$).

Finally, compute $E[B^m(s_0, w_0^m)]$ as in Eqn 5. We repeated the procedure for M fires of interest and computed $E[B(s_0)]$ as in Eqn 6.

The study area

The initial landscape is a study area of ~72 164 ha in the south portion of the Fort Rock Ranger District in the Deschutes National Forest of central Oregon (Fig. 1). The site is predominantly populated with ponderosa pine (*Pinus ponderosa*) and lodgepole pine (*Pinus contorta*), but also contains some mixed conifer, including mountain hemlock (*Tsuga mertensiana*). There is variability in topography, including some ridges and buttes across the site, but the overarching trend is a gentle decline in elevation from north to south. Elevation ranges from

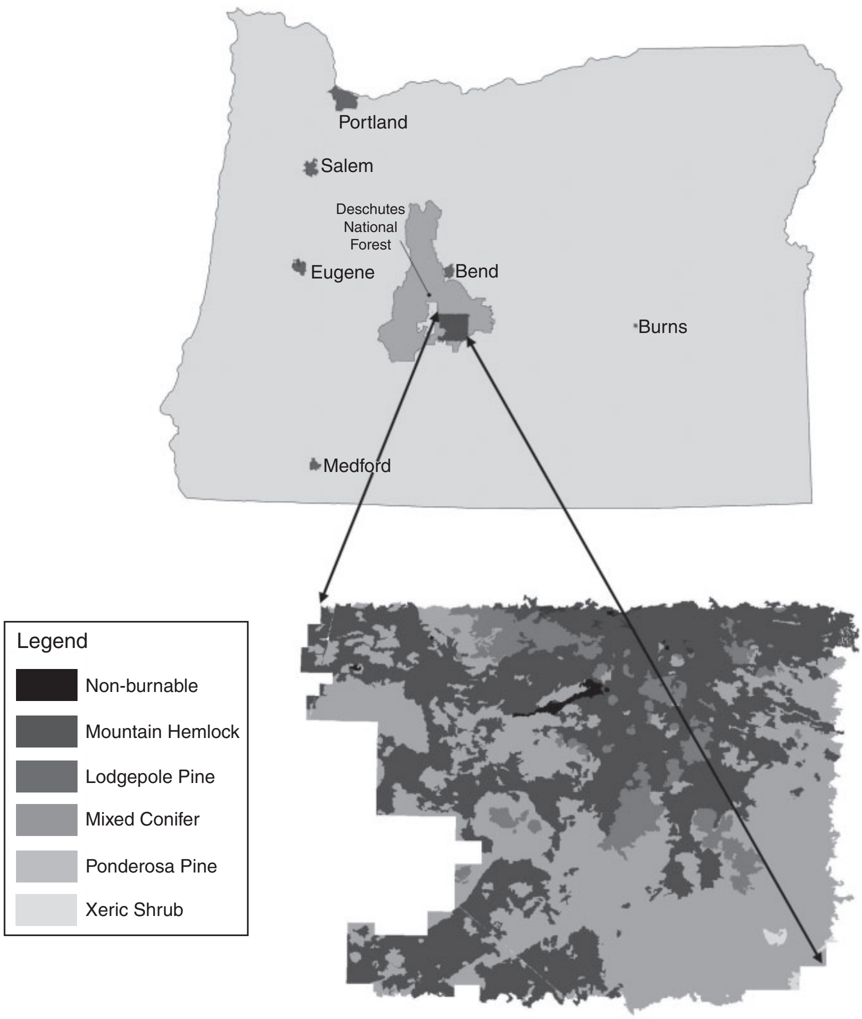

Fig. 1. The 72 164-ha study area in the southern portion of the Fort Rock Ranger District of the Deschutes National Forest in Oregon.

1300 to 2300 m. Because restoration is one of the management objectives in the Deschutes National Forest (USDA Forest Service Deschutes National Forest 1990, p. 4), clarified in the Central Oregon Fire Management Plan (COFMS 2009), and this particular site is relatively distant from concentrated residential development, it represents an area where a fire may actually be allowed to burn with no or minimal suppression actions.

The state of the initial landscape (s_0) is described by vegetation and fuel characteristics determined using the Forest Vegetation Simulator (FVS; Dixon 2002) and remotely sensed images of topography at a resolution of 30-m^2 pixels (LANDFIRE, http://www.landfire.gov/index.php, accessed 13 February 2011). The vegetation and fuels data were derived from stands that were delineated based on the homogeneity of vegetation and topographical characteristics. Tree lists from Forest Inventory Analysis (FIA) plots (USDA Forest Service 2000) were assigned to each stand using the gradient nearest neighbour method (Ohmann and Gregory 2002). All processing of the data into stands and assignment of tree lists was performed at the Western Wildland Environmental Threat Assessment Center in Prineville, Oregon (A. Ager and N. Vaillant, pers. comm., 7 November 2009). Surface and canopy fuel characteristics were assigned to each stand using the fire and fuels extension of the southern Oregon and northern California variant of the single-tree growth model FVS (Dixon 2002; Keyser 2008). All spreading fires were simulated using the Linux version of the fire simulation model FARSITE (Finney 1998). The FARSITE model was created to simulate wildfire behaviour on a landscape based on landscape characteristics, weather and ignition locations. It is spatial and temporal, allowing weather and wind to vary during a wildfire simulation. FFE-FVS was used to generate a table of state-transitions for the surface and canopy fuel attributes that then was employed in the simulations to update the post-fire landscape (described below).

The sample paths ($\mathbf{w}^{mn}$)

We generated a set of $N = 50$ sample paths for each of $M = 500$ fires of interest at time $t = 0$ with a time horizon of $T = 100$ and 1-year time periods[A]. Each sample path ($\boldsymbol{w}^{mn}$) must contain realisations of the random variables that drive FARSITE for each fire, including the fire of interest. For each fire of interest, the information described in w_0^m is held constant across the 50 futures (w_t^{mn}, $t = 1,...,T-1$) for each value $n = 0\ldots49$. These variables include the location of ignitions on the landscape, daily weather observations of maximum and minimum temperature, relative humidity and precipitation, and hourly wind speed, wind direction and cloud cover. The weather before the fire is employed to condition fuel moisture content at the start of the fire. The weather during the fire affects fire spread and crown fire activity. Weather also determines the duration for both suppressed and unsuppressed fires.

Historical hourly wind and weather data for the years 1985–2009 were obtained for the closest remote automated weather station (RAWS), Cabin Lake, from the Western Regional Climate Center (http://www.wrcc.dri.edu/, accessed 15 July 2011). We drew a weather stream for the entire fire season from this set of 25 observations. The weather that influences a particular fire depends on when the ignition occurs during the fire season.

Historical ignition data were obtained from the Deschutes National Forest Supervisor's office in Bend, Oregon (L. Miller, pers. comm., 23 July 2010).These included locations and dates of ignitions for the years 1985–2009. There was an average of 13 ignitions per year in the study area. Ignition variables were derived from the following historical ignition frequencies over the 25-year dataset: number of days each year on which at least one ignition occurred (average of 9 per year with a range from 4 to 19), dates of ignition days and number of ignitions per ignition day (average 1.49, with a range from 1 to 8). This resulted in an average of 15 ignitions per year in the sample paths (slightly more than the historical average to account for those that are located in areas with no burnable fuel). In order to check the validity of the simulated values, two measures of fire weather severity, energy release component (ERC) and spread component, were compared between the historical and simulated ignitions. Spread component is an indicator of potential fire spread rate based on wind and weather and ERC is a measure of expected energy release based on fuel moisture content (Bradshaw *et al.* 1984). The average values for ERC and spread component in the simulation fell within one percentage point of the historical values.

Approximately 98% of all ignitions in the forests of the northern Rockies and the east Cascade Range for which suppression is attempted are contained by initial attack (M. Finney, pers. comm., 4 February 2011). As a result, only the 2% of suppressed fires that escape initial attack spread on the landscape, requiring the simulator to determine fire size. Because most ignitions escape initial attack during weather events in which fire spread rates are high and fuel moisture is low, we drew spreading ignitions from the subset of ignitions that occurred on days for which spread component and ERC both exceeded the 90th percentile. To achieve a total probability of escape equal to ~2%, the probability of escape conditional on fire weather severity for our sample was set to 64%. The spreading ignitions were positioned on the landscape by drawing from a map of ignition probabilities (Fig. 2) created from

[A]The rationale for selecting each of these parameters is as follows. We wanted the time horizon to be at least long enough to allow lodgepole pine stands to burn in a fire of interest and to return to their current conditions. We found, after examining the results of our simulations, that 100 years was more than enough. The simulation process is computationally expensive. Each 100-year simulation could take as long as 20 min. We ran $N \times M = 25\,000$ paired simulations. Even though we had access to the Oregon State University College of Engineering High-Performance Computing Cluster (http://engineering.oregonstate.edu/computing/cluster/about.html, accessed 15 March 2013), we had to economise on simulations. Because we are ultimately interested in how the variables that are known at the time of ignition (w_0) affect the magnitude of suppression cost savings, we chose to simulate a relatively large number of fires of interest ($M = 500$) at the cost of simulating relatively few sample paths, $N = 50$, for each fire of interest. We could have reduced the confidence intervals around our estimates of cost savings for each fire of interest by increasing N. But we did find that the marginal gain in precision of the estimate was decreasing rapidly as we increased N.

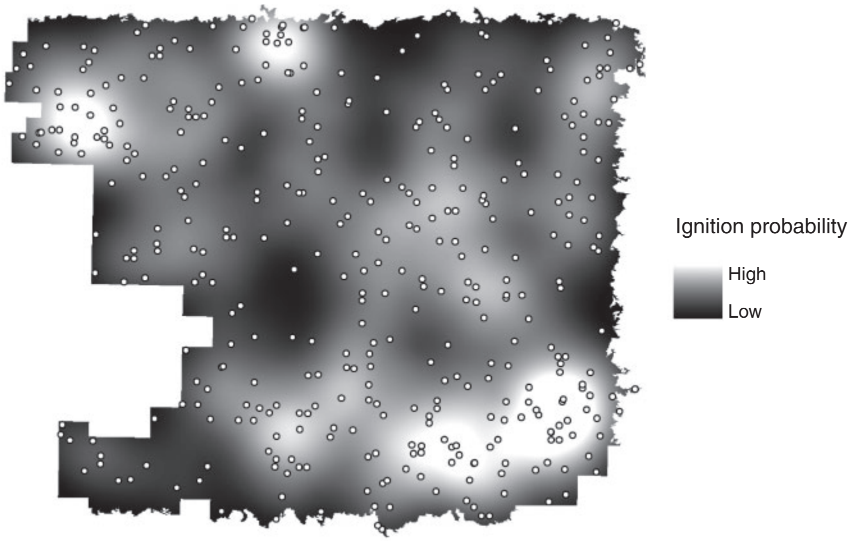

Fig. 2. Historical ignition points from 1980 to 2009 laid over map of ignition probabilities for each 30-m^2 pixel created using kernel smoothing.

historical ignition locations using the kernel smoothing function in *ArcGIS* (ESRI 2011) with a bandwidth of 4000 m. The fire of interest, which is allowed to burn in the let-burn scenario, was also assigned a location so that it could be simulated in FARSITE.

Fire duration for spreading ignitions under suppression was determined using a regression model of the probability of containment on a given day as a function of whether or not this was a spreading day (i.e. the spread rate was predicted to be higher than average for that fire on that day), the number of spreading intervals that had occurred to date and the fuel type (Finney *et al.* 2009). By experimenting with the fire spread model BehavePlus (Andrews *et al.* 2005), we identified a threshold above which a day was a spreading day in our study area defined by fuel moisture lower than 12% and wind speed greater than 15 miles h^{-1} (24 km h^{-1}). We then classified each day following an ignition accordingly. Suppression success was drawn according to the regression model for each day following a spreading ignition until the fire was contained. Fires that were not suppressed spread until either a fire-ending weather event (which we defined as a day when both spread component and ERC fell below the 20th percentile) or the end of the fire season (which we set at 31 October based on historical records) occurred.

The state-transition model, $S(s_t, w_t, x_t)$

The vector of state variables for each time period (s_t) must contain the attributes of the vegetation and topography that drive FARSITE for each pixel (or cell) on the landscape. The vegetation attributes include vegetation cover type by dominant species, surface fuel model (Anderson 1982) and forest canopy percentage cover, base height, total height and bulk density, output from FFE-FVS (Dixon 2002; Keyser 2008). A surface fuel model is a representation of surface fuels that allows broad classification of a wide number of ecosystems for the purpose of modelling wildfire spread. Using FFE-FVS, we selected a subset of the 13 fuel models developed by Anderson (1982) that apply to our study area. The forest canopy fuel attributes are employed to simulate crown fire behaviour in FARSITE. The vegetation attributes must be updated at the end of each time period. The state-transition model ($S(s_t,w_t, x_t)$) guides the transition of these state variables for each pixel in each time period depending on whether and how it burned. The topographical attributes include elevation, slope and aspect; these do not change and, hence, are not included in the state-transition model.

Table 1. Number and ranges of categories for vegetation state variables in the state vector (*s*)

Variable	Number	Class or range midpoint
Cover type	4	Lodgepole pine, ponderosa pine, mountain hemlock, mixed conifer
Surface fuel model[A]	6	5, 8, 9, 10, 12, 99
Canopy		
Cover (%)	4	0, 25, 55, 90
Total height (m)	4	0, 8, 24, 40
Base height (m)	5	0, 1, 2, 7, 15, 30
Bulk density (kg m^{-2})	5	0, 0.03, 0.08, 0.15, 0.28

[A]These fuel models are described in Anderson (1982).

$S(s_t,w_t, x_t)$ is implemented as a table linking initial states with ending states for each of three transition types (growth without fire, surface fire and crown fire) for each possible initial state. We kept the size of the state space manageable by binning the continuous variables as shown in Table 1[B]. The thresholds for each attribute were selected to reflect major changes in crown

[B]Without binning, the state space would be infinite. The alternative would be to model the transitions in FFE-FVS interactively in the simulations. However, the computational time required to do that is prohibitive.

fire behaviour. Each pixel on the initial landscape was assigned an initial state and a representative tree list according to its attributes. The initial stands for each cover type were simulated in FFE-FVS for the 100-year time horizon without fire to generate a base set of potential ending states. The stands comprising the base set of states were then simulated in FFE-FVS by burning with surface and with crown fire to generate post-fire states. The rest of the table was populated by iteratively growing, burning in surface fire and burning in crown fire each stand when it entered a new state until no new states were being generated. We also tracked time-in-state for unburned pixels; they transition only when they have been in a particular state long enough for at least one state variable to move from one bin to the next. The initial 'time-in-state' variable was assigned randomly to pixels in each state at a stand level on the initial landscape, in order to prevent large contiguous blocks from transitioning at once. Once a pixel reaches its climax state, it stays in the same state unless it is burned.

Fuel models describing surface fuel conditions are the most important fuel variable for determining fire spread rates. After an area burns, its fuel model is set to non-burnable for a given period, depending on the cover type of the stand and the expected post-fire build-up of fuels. Dry ponderosa pine stands required 20 years to replace fuels to reach a burnable state; mixed conifer, 30 years; mountain hemlock, 40 years; and lodgepole pine, 50 years. The length of time after a fire that it takes for fuels to reaccumulate enough for a new fire to spread varies in response to fire severity, precipitation, site class and climate. The values used here were based on published mean fire-return intervals (Kilgore 1981; Bork 1984; Shuffield 2011) and expert opinion, and may be altered in future work in order to capture the effect of these assumptions on the results.

Suppression cost estimation

Suppression cost was estimated and discounted to the present for each of two scenarios: allow the fire of interest to burn and suppress the fire of interest. We estimated suppression cost for three wildfire size categories: very small fires (less than 0.4 ha or 1 acre), which we assumed to be contained by initial attack, small fires that escaped initial attack (0.4–121.4 ha, 1–300 acres) and large fires (over 121.4 ha). All costs were adjusted to 2010 dollars using the all-commodity producer price index (USDL 2011). Very small fires were assigned a fixed initial attack cost of US$710 based on average reported suppression costs for fires smaller than 0.4 ha in the Deschutes National Forest between 1985 and 2009. Gebert *et al.* (2007) estimated a regression equation for predicting suppression cost for large fires. This was subsequently updated using new data (M. Thompson, pers. comm., 23 August 2010). The equation estimates suppression cost in dollars per hectare as a function of ERC, fuel type (brush, timber, slash), fire size, slope, elevation, aspect, distance to town and housing values within 32 km, and is based on fires reported in the National Interagency Fire Management Integrated Database (Bunton 2000) for large fires in the western USDA Forest Service Regions 1–6. We applied that equation to estimate suppression cost for fires over 121.4 ha by assuming the last two variables to be constant across fires and calibrating the equation for distance and property values in La Pine, the only town within 32 km. The Forest Service has not traditionally tracked unique characteristics for small fires that escaped initial attack (0.4–121.4 ha, 1–300 acres), so for these fires, we used a weighted average between the initial attack cost and the value computed by the suppression cost equation to estimate cost per hectare. A real discount rate of 4% was employed to compute present value as per USDA Forest Service policy (Row *et al.* 1981).

One potential cost of let-burn that we excluded is the cost of monitoring. A wildfire would not be allowed to burn without some amount of monitoring and possibly protection of specific resources on the landscape. Other than timber, there are few resources that could require protection within the study area. In addition, there is an extensive road system that allows rapid access throughout the study area, which decreases monitoring costs. As a result, we assume that these costs would be small. In the absence of a reasonable method for estimating monitoring costs, we elected to exclude them from our analysis.

Discussion of results

A histogram of estimated suppression cost savings ($E[B^m(s_0, w_0^m)]$) for $M = 500$ fires of interest is shown in Fig. 3 in US \$100 000 intervals based on $N = 50$ sample paths for each. The distribution has two peaks. The first peak near zero is the result of fires of interest that are small and as a result, do not, on average, have much effect on future suppression costs. The second peak is the result of the average future suppression savings from larger fires. Because the distribution of values for each of $N = 50$ sample paths was not normal, we calculated bootstrap confidence intervals using the accelerated bias-corrected percentile method (Givens and Hoeting 2005, p. 261) to estimate the 95% confidence interval around each mean. We found that 91.2% of the 500 fires of interest had a positive mean with a 95% confidence interval that excludes 0.

Our estimate of expected present value of suppression cost savings ($E[B(s_0)]$) for the study area landscape was US$34 per hectare or ~US$2.47 million. This is the average over all $M = 500$ fires of interest and $N = 50$ sample paths (a total of 25 000 paired simulations). Again, owing to the non-normal distribution of point estimates, we used the accelerated bias-corrected percentile method to estimate confidence intervals. The 95% bootstrap confidence interval around the mean has a lower bound at US$2.36 million and an upper bound at US$2.59 million, which indicates that, on average, future suppression cost savings are positive on this landscape.

The simulations that generated very large suppression cost savings typically had two characteristics: (1) a large initial fire of interest and (2) a subsequent ignition early in the time horizon during severe fire weather. That subsequent ignition occurred in a location that had been burned in the let-burn scenario and had not reaccumulated enough fuel for a fire to spread, but that had not been burned in the suppress scenario and, because of severe weather, developed into a large fire that was costly to suppress. The sample paths that had *positive* but small suppression cost savings also had future ignitions in areas that were burned in the let-burn scenario but not in the suppress scenario; however, they either occurred later in the time horizon (so benefits were more heavily discounted and fuels had subsequently grown to replace

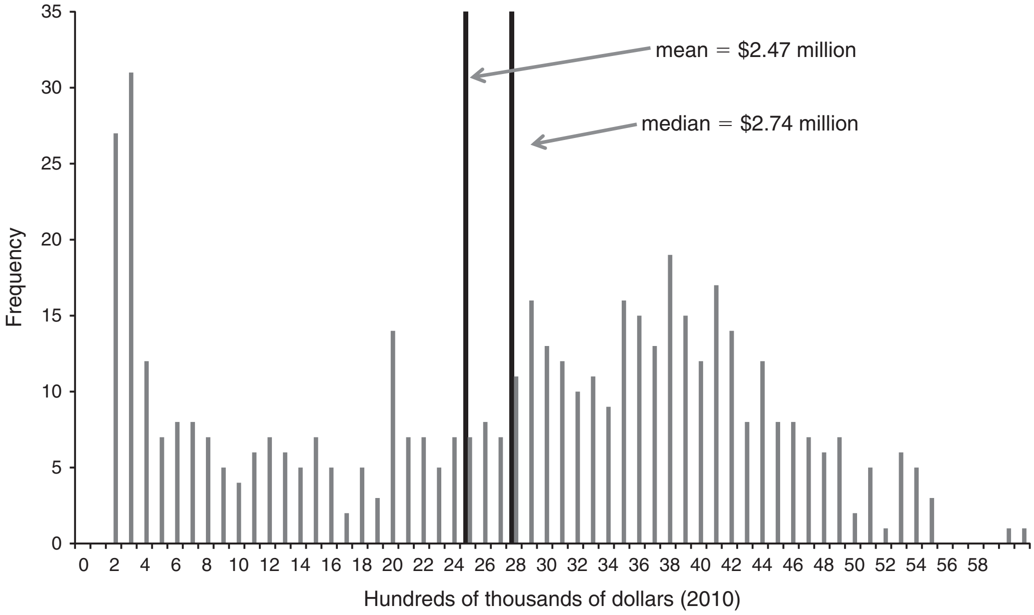

Fig. 3. Frequency with which estimated expected present value of future suppression cost savings for a fire of interest ($E[B^m(s_0, w_0^m)]$), $m = 1,\ldots,500$, falls in US$100 000 value intervals.

those that had burned), close to the end of the fire season, or in milder weather and so were contained quickly.

There were several simulations that exhibited no future suppression cost savings (2294 out of 25 000 paired simulations). This lack of cost savings in simulations is the result of fires of interest that ignited either during marginal weather and did not spread, or burned areas that did not burn again in the future. And there were a few paths that had negative suppression cost savings, meaning that future suppression costs were higher in the let-burn scenario than in the suppress scenario. This happened when a future ignition occurred in an area that had been burned in the fire of interest of the let-burn scenario and not in the suppress scenario. Subsequent fires took place after a period that was long enough that the fuels had evolved into a burnable state, but they evolved differently between the two scenarios. In many cases, early seral vegetation includes a higher load of small fuels, which results in a higher spread rate than is found in older stands. As a result, the area burned in the let-burn scenario evolved into a high-spread-rate fuel model, whereas the area that did not burn in the suppress scenario stayed in a relatively slower-spread-rate fuel model. For further details, see Houtman (2011).

In order to validate our visual inspection of the data with regard to the relationship between expected benefit and fire size (ha), we ran a logit regression of a binary expected benefit variable on the fire size of the fire of interest. To create the binary expected benefit variable, we split the sample set of 500 fires of interest into two categories, where fires producing an expected benefit greater than the median value were assigned a value of 1 and fires producing less than the median value were assigned a value of 0.

The results show that average suppression cost savings increased with the size of the fire of interest (z values are -8.84 for the constant -7.677 and 9.60 for the coefficient for fire size 0.0002 ; Rho^2 adjusted $= 0.714$; the variable p_m is the probability that the expected benefit of fire of interest m is greater than the median expected benefit):

$$logit(p_m) = -7.677 + 0.0002 \times fire\ size_m$$

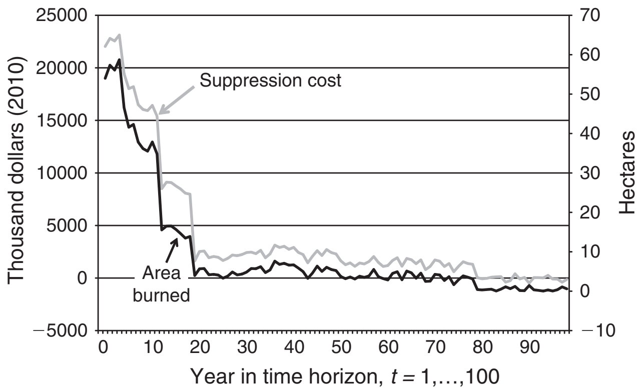

Fig. 4. Average annual change in suppression cost in thousands of dollars (2010) and area burned in ha for each year in the time horizon, $t = 1,\ldots,100$.

A large fire produces more fuel treatment than a small fire, which can increase the difference in the size and, hence, the estimated difference in fire suppression costs for subsequent fires. The average annual change in suppression cost and the average annual reduction in area burned for the 500 fires of interest in each year in the time horizon are shown in Fig. 4. These variables are highly correlated because, for a given sample path, fire size is the most important factor determining fire suppression cost in the equation that we used. This shows that the effect of the fire of interest on subsequent fires largely disappears after ~25 years under our assumption that all subsequent fires will be suppressed. This result also depends on our assumptions about the length of time it takes for the areas that are burned in the fire of interest to generate sufficient fuel loads to carry a fire.

Surface fire and crown fire have very different effects on forest ecosystems. Crown fire is often stand-replacing, resulting in a greater loss of timber value, recreational opportunities and wildlife habitat, whereas surface fire typically results in reduced fuel load and less densely stocked stands and, hence, is largely beneficial. We found that the proportion of the total area burned in crown fire in subsequent fires was approximately the same whether the fire of interest was allowed to burn or not (averaging 7–8%). However, because the total area burned was less in the let-burn scenario, the extent of crown fire was also reduced.

Our analysis indicates that the potential exists for unsuppressed wildfire to generate positive benefits in the form of reduced future suppression costs, but that is only one component of the total cost-plus-net-value-change represented by Eqn 3. The benefit of allowing a fire of interest to burn also includes avoided current suppression cost and reduced damage from subsequent fires due to lower fuel loads. However, the potential benefit of wildfire may well be offset by the potential damage that it may cause, possibly by a large amount.

In this study, our objective was to estimate potential future suppression cost savings from allowing a fire of interest to burn on a particular landscape. However, to put our estimates of E $[B^m(s_0, w_0^m)]$ in perspective, we also developed a preliminary estimate of one component of fire damage – loss of timber value resulting from unsuppressed fire. We emphasise that this is a rough estimate constructed for exploratory purposes only. Although timber harvest is scheduled for our study area under the current Deschutes National Forest Plan (USDA Forest Service Deschutes National Forest 1990), in the future, we also will need to consider other relevant management objectives when evaluating the optimality of a let-burn decision, including, but not limited to, wildlife habitat, restoration, recreation use and risk to adjacent properties.

For our estimate, we assumed standard timber management regimes for ponderosa pine and lodgepole pine based on communication with Deschutes National Forest silviculturists (M. Deppmeier and B. Schroeder, pres comm., 5 August 2010). We also assumed that the entire study area is managed for timber on these regimes, that there are no restrictions on removals and that the forest is currently regulated so that harvest equals growth. These assumptions mean that our rough estimate represents an upper bound on potential timber value loss to fire. Yield estimates were based on average 50-year site indexes for lodgepole pine and ponderosa pine for the study area (Bennett 2002; Emmingham *et al.* 2005)[C]. For ponderosa pine, we assumed that surface fire would cause no damage but that crown fire would be stand-replacing. For lodgepole pine, surface fire was assumed to reduce harvest volume by 50% in the next harvest and crown fire was assumed to be stand-replacing. Although salvage logging is common after a fire, we assumed no post-fire salvage harvest. Harvest and haul cost and log prices were obtained from the Oregon Department of Forestry (http://www.oregon.gov/ODF/STATE_FORESTS/timber_sales/logP304.shtml, accessed 12 April 2011)[D].

For each sample path, we computed the area of lodgepole pine and ponderosa pine burned in surface fire and in crown fire in each time period for the suppress scenario and for the let-burn scenario. We then computed value loss to fire under each scenario as the present value of the change in land-and-timber value[E] on the landscape resulting from fire in each time period, including the current time period, $t = 0$, and took the difference between the estimated loss for the let-burn and for the suppress scenarios. This yielded an average change in net present loss of timber value to fire of approximately US$18.08 million for the study area or US$250 per hectare for the study area landscape.

Combining suppression costs savings with loss of land-and-timber value yields an average cost-plus-net-value-change of $\Delta v = -$US$15.06 million. This means that under our timber management log price assumptions, it is generally not optimal to allow wildfire to burn on this landscape, given the value at risk of loss to fire as we defined it here. Nonetheless, with these estimates, 23 of the 500 fires of interest, or 4.6%, yielded positive net benefits ($\Delta v > 0$) from allowing the fire of interest to burn. For these paths, the fires that were allowed to burn tended to be surface fires in ponderosa pine that were smaller than the average unsuppressed fire. We anticipate that a more realistic value-at-risk estimate that is consistent with the management objectives described in the Deschutes National Forest Plan (USDA Forest Service Deschutes National Forest 1990) will yield a higher proportion of the sample loss-plus-net-value-change estimates that exhibit positive net benefits.

[C]Ponderosa pine stands were assumed to be thinned every 20 years to a base growing stock of 43.5 million board-feet (MBF, 1 board-foot = 1 ft × 1 ft × 1 in) per hectare, which corresponds to age 60 on 50-year site index 80, removing 27.5 MBF per hectare (Bennett 2002). We used current standing volume to determine when existing stands would first be thinned in the absence of fire. Lodgepole pine stands were assumed to be clearcut-harvested at age 80, yielding 38.5 MBF per hectare, which corresponds to a 50-year site index of 60 (Emmingham *et al.* 2005). The existing lodgepole pine forest area was assumed to be fully regulated so that 1/8th of the area would be harvested each decade.

[D]We used average quarterly log prices from 1995 to 2009 (the same period over which the suppression cost equations were estimated) for the Klamath region in Oregon of US$544 per for ponderosa pine sawlogs and US$375 per MBF for lodgepole pine less 'rule-of-thumb' harvest and haul cost of $225 per MBF (Oregon Department of Forestry, http://www.oregon.gov/ODF/STATE_FORESTS/timber_sales/logpdef.shtml, accessed 12 April 2011). The real discount rate was 4% (Row *et al.* 1981).

[E]Land and timber value (LTV) for unburned lodgepole pine is the present value of a perpetual series of clearcut-harvest revenue every 80 years with 1/8th of the area scheduled for first harvest at the end of each of the first 8 decades. For area burned in surface fire, harvest volume is reduced by 50% for the next scheduled harvest. Area burned in crown fire reverts to bare land with the next scheduled harvest occurring in 80 years. LTV for unburned ponderosa pine is the present value of a perpetual series of thinning harvest revenue every 20 years with the next scheduled thinning dependent on standing volume in the initial stands. For area burned in surface fire, there is no change. Area burned in crown fire reverts to bare land and the next scheduled thinning occurs in 80 years. Loss to fire is estimated in each scenario as the change in LTV in each time period discounted to the present.

Conclusion

One of the potential benefits of allowing a wildfire to burn is that it provides 'free' fuel treatment, resulting in reduced fuel loads that make subsequent fires easier and less costly to contain. In this analysis, we estimated the expected value of that benefit on a landscape in the Deschutes National Forest of central Oregon using Monte Carlo methods. We combined models of fire behaviour, forest vegetation, fire suppression effectiveness, and fire suppression cost to simulate fire on the landscape, update the vegetation and forest fire fuels, and estimate the effect of allowing a current wildfire to burn on the suppression cost for subsequent fires.

Our estimate indicates that potential cost savings may be substantial. For the sample path that exhibited the highest expected benefit, the present value of the reduction in future suppression costs was nearly US$5.8 million. For most of the sample paths, the estimated benefit was modest, but positive, averaging US$2.47 million for the study area landscape over a sample of 25 000 paired simulations. For a few, future suppression costs were actually higher in the let-burn scenario. The category into which each fire of interest falls is dependent on how fuels, and specifically surface fuels, transition over time with and without a burn in the current period. We found that estimated expected future suppression cost savings were positively correlated with the size of the fire of interest. This is not surprising because large fires provide more fuel treatment.

However, fire damage may also be positively correlated with fire size as more forest is burned. The risk of damage from unsuppressed fire must be weighed against the potential benefit within the context of the owners' management objectives when making a decision about whether a particular fire should be allowed to burn. It is the *net* benefit of allowing a fire to burn that is the relevant criterion. We constructed a preliminary estimate of the potential loss of timber value in order to get an idea of the likelihood that suppression cost savings might outweigh fire damage in our study area. We included both loss to the fire of interest and reduced loss to subsequent fires resulting from the fuel treatment effect of the fire of interest. On average, the estimated loss outweighed the estimated benefit by an order of magnitude. Nonetheless, even with an estimate of timber value at risk that is highly likely to be biased upwards, the benefit exceeded the cost for 4.6% of the sample. This suggests a compelling avenue for future research – to investigate the conditions (i.e. weather, ignition location, ignition timing, value-at-risk, etc.) under which the benefit of allowing a fire to burn is likely to exceed the cost and then to use that information to develop a tool to inform the forest planning process by identifying areas that meet those conditions – areas that could be considered for cautious use of wildfire as a management tool.

In order to understand how timing and location of fires affect the management of fire for the purpose of achieving land-management objectives, it will be necessary to expand certain areas of this research and consider how to incorporate that knowledge into the existing fire-management planning process.

First, the effect of wildfire on the full range of ecosystems services that are generated on this landscape, including timber, recreation, wildlife habitat and aesthetic values, must be modelled and valued in a way that allows comparison with potential suppression cost savings. Fire effects may involve damages in some periods and benefits in others as vegetation develops over time. Ideally, the range and extent of ecosystem services considered in the model should reflect current management objectives for the study area and be consistent with the Deschutes National Forest Plan (USDA Forest Service Deschutes National Forest 1990).

Second, the new interpretation of federal wildfire policy permits managers considerable flexibility in allowing wildfire to spread in order to achieve ecologically beneficial outcomes. The past contrast between suppressing wildland fires and wildland fire use no longer exists. Instead, a given fire may be managed for ecological benefits on one flank, while being aggressively suppressed on another flank to protect highly valued resources from loss. In this new paradigm, all fires have a suppression objective; however, suppression activities may not occur until the fire reaches designated areas. Thus, a more realistic simulation effort could be engaged by identifying areas within the forest where transition to suppression objectives are likely to occur and simulating fire spread and management response to wildfire movement.

The potential for wildfire to either expand into areas designated to trigger suppression, or burn under conditions where the ecological fire effects switch from beneficial to detrimental owing to intensity, is closely tied to the weather in the days and weeks after the initial ignition. These variables are difficult to predict, particularly early in the fire season. Given this uncertainty, managers are cautious of allowing wildfires to burn early in the fire season, when potential fire spread and effects may become more extreme as the fire season progresses, and fire management plans may not sufficiently consider the role of individual fires in achieving broader-scale land-management goals (Doane *et al.* 2006). Simulation efforts such as this could test rules of fuel conditions, time of year, weather variables and values at risk in order to explore more flexible fire-management plans that may promote the expansion of ecological objectives of the fire-management program.

The results shown in Fig. 4 indicate that fuel treatment benefits of allowing one fire to burn are largely dissipated after the first 25 years of the simulation time horizon owing to reaccumulating fuel loads. This is partially the result of excluding the long-term effect of fires on the ecology of burned areas. In reality, the ability to achieve ecological objectives through burning may be enhanced in areas that have already experienced a burn within the historical fire return interval (Finney *et al.* 2005; Fontaine *et al.* 2009). This level of simulation is currently challenged by our lack of knowledge regarding how suppression activities affect final fire size, resource value change and even management costs. However, emerging risk-based decision-support tools (see Calkin *et al.* 2011 for a review) may allow simulation exercises that can test alternative future scenarios and help managers explain proposed changes in fire management to the public.

In the simulations reported in this paper, a policy of 'suppress all wildfire' was imposed in future time periods. But as a society, we have created a situation in which the status quo for wildfire management is no longer sustainable; increasing fuel loads combined with likely effects of climate change will make it even more difficult and costly to contain the wildfires of the future unless there is some success in restoring historical fire regimes to the fire-prone forests of the western United States. Current federal wildfire policy now prescribes allowing wildfire to burn on some landscapes as a natural ecosystem process when it can be done while maintaining a high level of firefighter and public safety (NWCG 2001). Every National Forest is required to have a fire management plan that describes how ignitions will be treated. For example, one goal for an area that is targeted for forest restoration could be to restore forest conditions that would allow a let-burn policy for many, if not most, wildfires.

Accordingly, we intend to extend this research by applying the simulation platform we constructed here to develop a policy rule that could be dynamically applied to the let-burn decision for each subsequent fire depending on the state of the fuels on the landscape, the ignition location, both spatially and temporally, the weather occurring at the time of the ignition and the absence or presence of simultaneous fires. This will require development of a more comprehensive and credible model of values at risk on the landscape that reflect management objectives for the study area. It will also require implementation of an algorithm that allows us to learn a 'best' policy for subsequent fires from repeated simulations, perhaps using methods of reinforcement learning or approximate dynamic programming (Powell 2009).

There are barriers to the implementation of a policy of allowing wildfire to burn, including concern on the part of fire managers regarding personal liability should wildfire destroy property or take human life. The analysis reported here takes one step towards a better understanding of when a let-burn choice might be worth that risk.

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