Simulation of the Consequences of Different Fire Regimes to Support Wildland Fire Use Decisions

Carol Miller

Aldo Leopold Wilderness Research Institute,
Rocky Mountain Research Station, USDA Forest Service,
790 E. Beckwith Ave., Missoula, MT 59801 USA
Tel.: (406) 542-4198; e-mail: cmiller@fs.fed.us

ABSTRACT

The strategy known as wildland fire use, in which lightning-ignited fires are allowed to burn, is rapidly gaining momentum in the fire management community. Managers need to know the consequences of an increase in area burned that might result from an increase in wildland fire use. One concern of land managers as they consider implementing wildland fire use is whether they can meet the goals in the land management plan for the desired distribution of forest structural stages across the landscape with further increases in fire. These questions were explored for a 49,532 ha landscape on the Boise National Forest in Idaho that typically experiences mixed-severity and stand-replacing fires. The landscape simulation model TELSA was used to evaluate how increases in fire frequency and area burned might affect landscape composition and structure. Information about frequency, annual area burned, and size-class distributions of fires derived from a fire atlas for the northern Rocky Mountains were used to define the fire regime parameters for five different simulation scenarios. Scenarios with higher fire frequency and area burned resulted in landscapes dominated by earlier successional forest stages and only small patches occupied by large trees. Simulated variability in area occupied by different tree-size classes on this landscape was much greater than the desired ranges defined in the land management plan for the forest at large. A measure of dissimilarity (Euclidean Distance) from desired composition was used to evaluate scenarios for their relative ability to achieve long term land management goals. The lowest values of Euclidean Distance were for a scenario that represents a substantial increase in fire over 20th century fire regimes. Euclidean Distance increased for scenarios with very high rates of burning, implying an upper limit to the desired amount of fire for this landscape. These findings could be used to develop guidance for achieving desired conditions with wildland fire use.

Keywords: contagion, desired future conditions, Euclidean Distance, forest age-class distribution, landscape composition, landscape fire succession models, northern Rocky Mountains

Citation: Miller, C. 2007. Simulation of the consequences of different fire regimes to support wildland fire use decisions. Fire Ecology 3(2): 83-102.

INTRODUCTION

Over the past 35 years, the fire management strategy of allowing lightning-ignited fires to burn (wildland fire use, WFU) has been applied primarily in wilderness and national parks, where the goal is to restore and maintain natural ecological processes (Parsons and Landres 1998, USDA and USDI 2001). Increasingly this fire management strategy is being applied to lands with multiple, non-wilderness objectives (Miller 2003; Tim Sexton, Forest Service, personal communication). To effectively implement WFU on lands that may be managed for a myriad of wildlife, timber, watershed, and recreation objectives, land managers need to understand how WFU fires will affect their ability to achieve the specific desired future conditions (DFC) articulated in land management plans. Managers also need to understand the consequences of this management strategy in the context of a changing climate in which increased temperatures and longer fire seasons will likely result in an increase in area burned (Wotton and Flannigan 1993, McKenzie *et al.* 2004, Westerling *et al.* 2006). The effect of climate change together with increased implementation of WFU strategies will most certainly result in increased fire frequencies. As such, land managers need to know what to expect with increased fire, and whether WFU will help or hinder achievement of desired future conditions.

Land or resource management plans define desired future conditions for a landscape, and federal policy recognizes that these plans have to account for the role of fire (USDA and USDI 2001). WFU activities should support the achievement of those desired goals (USDA and USDI 2005). With the WFU decision, land managers have some control over when and under what conditions a landscape will be burned, and consequently, how forest structure is affected (Fulé and Laughlin 2007). The cumulative result of these decisions affects the frequency, severity, and size of fires (i.e., the fire regime), which in turn can influence the structure and composition of landscapes (Romme 1982, Turner and Romme 1994, Brown *et al.* 1999, Weir *et al.* 2000).

Fire can create heterogeneity that may be essential for the evolution of diversity and stability in ecosystems (Turner *et al.* 1994), and this heterogeneity may influence behavior of subsequent fires (Turner *et al.* 1989). An understanding of how fire regimes affect the landscape patterns to which species respond is an important need for effective land management planning (e.g., Cissel *et al.* 1999).

Many simulation studies have investigated consequences of historical (e.g., pre-Euro-American settlement) and altered fire regimes on various aspects of landscape structure and composition (e.g., Baker 1994, Gardner *et al.* 1996, Gauthier *et al.* 1996, Boychuk and Perera 1997, Keane *et al.* 1999, Miller and Urban 1999, Wimberly 2002, Scheller *et al.* 2005, Schoennagel *et al.* 2006, Didion *et al.* 2007). Studies like these are very effective for exploring landscape dynamics—especially in response to fire. Approaches using landscape fire succession models (*sensu* Keane *et al.* 2004) can be particularly valuable for describing reference conditions in terms of a historical range of variability (HRV) of landscape composition or structure (Morgan *et al.* 1994, Landres *et al.* 1999). Departure from a reference HRV has been used to describe forest health and design forest restoration treatments (e.g., Hann and Strohm 2003). In certain situations, HRV may not be attainable—or even desirable—and therefore is not the same as the desired future conditions defined in planning documents. Even so, the simulation approach for determining HRV is becoming a standard for informing the setting of fire and land management goals (Pratt *et al.* 2006, Holsinger *et al.* 2006).

Landscape fire succession models can be valuable for informing and supporting long term fire and land management planning (e.g., Chew *et al.* 2004, Fall *et al.* 2004, Scheller *et al.* 2005). However, their value in supporting fire management decisions in the short term, such as at the time of the WFU decision, has not yet been explored. After desired future conditions are determined and land management goals are set, managers need to understand how fire management can help achieve those specific goals. For example, should they implement strategies like WFU that result in more fire over the next 10 or 20 years, or should they minimize fire during that time frame?

I used a state-and-transition type landscape fire succession model to illustrate how simulation modeling might be used to support wildland fire use decisions and planning. I explored the consequences of five different levels of fire on the composition and structure of a forested landscape in the northern Rocky Mountain region of the United States. I parameterized the model with fire history information derived from a digital fire atlas of the region and initialized simulations with current vegetation data. I used simulation results and a measure of dissimilarity (Euclidean Distance) to evaluate how different levels of fire would affect the ability of managers to achieve and maintain the desired distribution of tree-size classes on the landscape.

METHODS

Study Area

The study area is a 49,532 ha landscape in southwest Idaho, USA, in the Bear Valley and Elk Creek watersheds of the upper Middle Fork Salmon River sub-basin (Figure 1). The northern third of the landscape is within the Frank Church River of No Return Wilderness. Much of the area that lies outside the wilderness portion is inventoried as roadless (Forest Service 2003). Approximately 5,800 ha have been recommended by the Forest Service for wilderness designation, and portions of two rivers eligible for Wild and Scenic designation also fall within the study area. Elevations range from 1,800 m to 2,900 m. Surface geology is dominated by granitics of the Idaho batholith, and the landscape is characterized by glaciated mountains, rolling uplands, and broad valley bottomlands. Slope gradients average between 15 % and 40 &. Vegetation consists largely of lodgepole pine (*Pinus contorta*) and subalpine fir (*Abies lasiocarpa*) forests, interspersed with meadows.

WFU is authorized for the entire study area, but has not yet been implemented outside the wilderness portion. Approximately 60 % of the forested portion typically experiences a mixed severity fire regime (Fire Regime type III, Schmidt *et al*. 2002) with average fire return intervals of 80 yr and severities that include a mixture of stand-replacing and non-lethal fire. The remaining 40 % experiences stand-replacing fires (Fire Regime type V) that result in high rates of tree mortality. A very small percentage of the area experiences frequent, low severity fires (Fire Regime type I). Approximately six lightning-caused ignitions occur per year on average in the study area (Forest Service 1993). According to the digital fire atlas, a cumulative total of 17,352 ha burned between 1908 and 2003 (Gibson 2006). Like much of the western U.S., area burned has increased dramatically since the 1980s, partly in response to changes in climate (Morgan *et al*. in press). Five of the ten largest fire-years recorded in the fire atlas for this landscape have occurred since 1985.

Potential Vegetation Groups (PVGs) have been described for the area (Steele *et al.* 1981). Although PVGs are labeled according to the vegetation types that could potentially occupy a site, they do not necessarily describe the existing vegetation and are really classifications

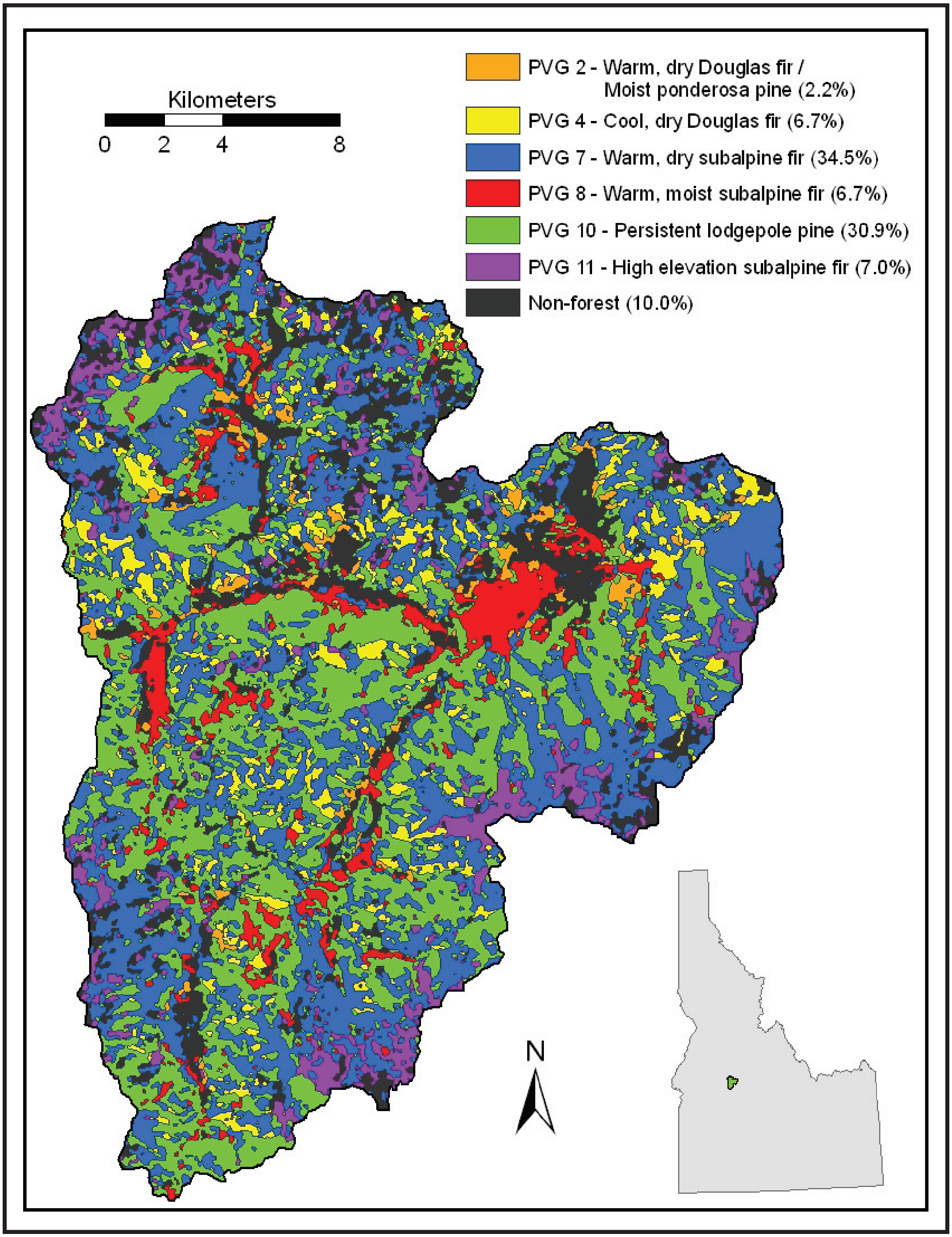

Figure 1. The location and distribution of Potential Vegetation Groups (PVGs) of the 49,532 ha Bear Valley-Elk Creek study area.

of biophysical setting (e.g., areas with similar temperature, moisture, and soil conditions). The range of desired conditions for tree size classes for each of the PVGs is described in Table 1. Tree size class refers to the average diameter of trees in the overstory or uppermost tree layer. Tree size corresponds loosely with tree age, with older trees attaining larger sizes. Desired ranges for PVG 10 (persistent lodgepole pine) are not specified for the largest tree size class (>50.8 cm dbh) because even very old trees in this PVG typically do not attain these sizes. Desired future conditions for forested vegetation on the Boise National Forest were determined using HRV as the reference condition for each PVG (Morgan and Parsons 2001). The HRV estimates were developed previously through simulation with the Vegetation Dynamics Development Tool (VDDT, ESSA 2005). Adjustments were made depending on the management status of the land. For example, the lower end of the HRV for large trees was used as DFC for lands outside inventoried roadless areas, whereas the higher end of HRV for large tree was used for roadless areas.

Simulation Model

TELSA (Tool for Exploratory Landscape Scenario Analyses) was used to investigate how different fire regimes affect forest structure and composition of this landscape. TELSA is a spatially explicit modeling framework that can project the consequences of alternative management and fire scenarios (Kurz *et al.* 2000). TELSA simulates forest succession and the effects of fire and forest management activities as changes in age and structural stages of stands using successional pathway diagrams developed with VDDT. These processes occur within and among simulation polygons that are classified by forest type and structural stage.

In this study, I used six successional pathway diagrams representing the six forested PVGs that occur in this study area. Succession for non-forest PVGs was not simulated due to lack of confidence in the mapping of non-forest PVGs and associate succession pathway diagrams. The diagrams for the six forested PVGs, or state-and-transition models, were previously developed using VDDT as part of the Southwest Idaho forest plan revision process (USDA Forest Service 2003). Pathway diagrams for five of the six PVGs comprised 12

Table 1. Desired conditions developed for the Boise National Forest Plan.

		Range of desired tree size classes[1] (% of area)				
Potential Vegetation Group	**Area (ha)**	**Seedling >1.4 m tall**	**Sapling 2.5-12.6 cm dbh**	**Small 12.7-30.5 cm dbh**	**Medium 30.6-50.8 cm dbh**	**Large >50.8 cm dbh**
PVG 2: Warm, dry Douglas-fir/moist ponderosa pine	1,064	5-7	3-7	5-21	7-35	30-80
PVG 4: Cool, dry Douglas-fir	3,339	14-15	7-9	19-22	24-36	20-34
PVG 7: Warm, dry subalpine fir	17,082	7-9	11-15	21-22	32-36	20-21
PVG 8: Warm, moist subalpine fir	3,298	15-17	11-15	22-23	28-29	20-21
PVG 10: Persistent lodgepole pine	15,317	16-23	11-16	46-48	20	-
PVG 11: High elevation subalpine fir	3,464	9-15	14-15	19-22	22-38	20-27

[1] English equivalents for tree size class descriptions: Seedling (<4.5 ft tall); Sapling (1” to 4.9” dbh); Small tree (5” to 12” dbh); Medium tree (12.1” to 20” dbh); Large tree (>20” dbh).

structural stages, or state classes, that describe a combination of canopy closure and tree size class (Figure 2). The pathway diagram for PVG 10 (persistent lodgepole pine) comprised only 9 structural stages because the largest size class (>50.8 cm dbh) is not relevant for this PVG. Forest development is simulated as a series of transitions between state classes that are either deterministic and age-related, or stochastic and related to fire. For example, in the absence of fire, a forest in the warm-dry sub-alpine fir type (PVG 7) remains in the sapling/low-canopy-closure state class for a specified period of time (in this case, 9 years) before transitioning to the small-tree/moderate-canopy-closure state. However, when a stand-replacing fire occurs, the forest immediately transitions to the seedling state class. Although the VDDT models developed previously included probabilities and pathways to represent forest management activities and multiple change processes (i.e., including insect outbreaks), I did not simulate any management activities and only considered stand-replacing and non-lethal fires.

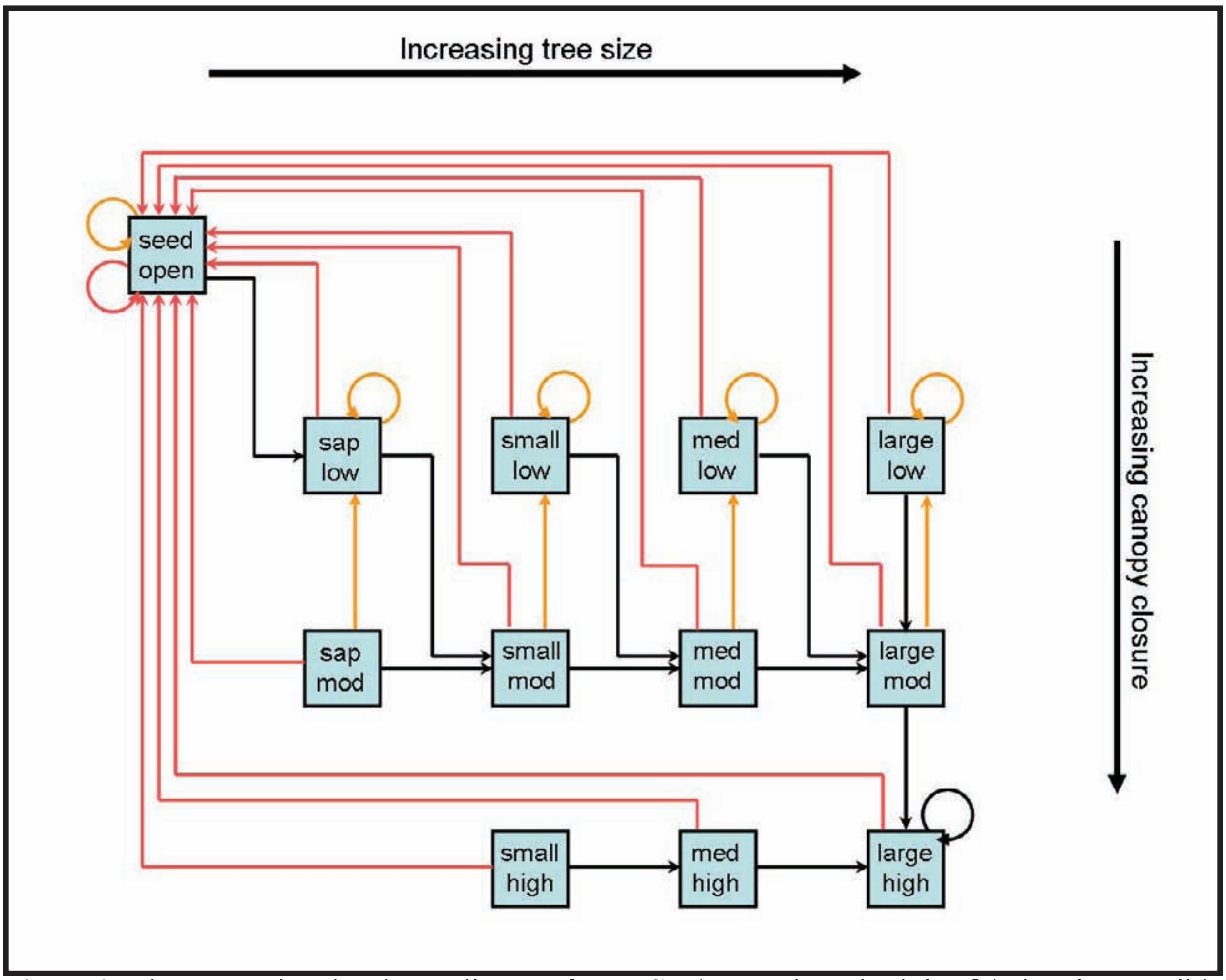

Figure 2. The successional pathway diagram for PVG 7 (warm-dry sub-alpine fir) showing possible transitions due to succession and fire among 12 successional states. Deterministic succession transitions are shown as black lines; transitions resulting from stochastic non-lethal and stand-replacing fires are shown in orange and red, respectively. Each state is defined by one of five tree size classes (see Tables 1 and 3) and one of four canopy closure classes: open (<10 %), low (10 % to 39 %), moderate (40 % to 69 %), and high (>69 %).

Fires are simulated stochastically in TELSA. The location, number and size of fires in a simulation year are determined from the collective probabilities for the state classes that exist on the landscape in that year, as well as user-provided information about the size distribution of fires and the inter-annual variability in fire probabilities. The total area to be burned in a year is determined from the current distribution of state classes, the fire probabilities for each state class, and a multiplier that describes between-year variability in area burned. Multiple fire events may occur in a single year to attain the total area burned. Before a fire event is simulated, its target size is drawn from the fire size distribution. The fire is then initiated in an eligible (i.e., burnable) polygon picked at random and then spread to eligible neighbors until the target event size (determined from the fire size distribution) is reached. Fires are spread in a contagious fashion from an initial polygon to adjacent polygons, with fire spread being forced along polygon boundaries. When the fire size has reached its target size, it is allowed to finish burning the current polygon before it is stopped. The effect of fire is to change the state class or age of the simulation polygon according to the rules in the successional pathway diagrams. Individual fire events may have a mixture of stand-replacing and non-lethal fire types; the type of fire that affects a simulation polygon is determined by the probabilities for its current state class. Table 2 lists the probabilities of both types of fire used in the VDDT state-and-transition models for the two dominant PVGs (7 and 10). Although TELSA has the option to simulate directional influences of a prevailing wind and topography on fire spread, I did not include these wind and slope effects in this study in order to avoid introducing an additional source of variability. In addition, spread was not simulated through non-forested PVGs due to the lack of reliable information on fire probabilities.

Input Data

Vegetation classifications developed for the forest plan revision (USDA Forest Service 2003) were used to define six forest PVGs plus one non-forest group (grassland, shrubland, barren, and water) in the study area. Approximately two-thirds of the non-forest group (8 % of the study area) comprises grasslands and shrublands, the remaining non-forest area (4 % of the study area) is water, rock or barren land. Each forest PVG represents a group of forested habitat types that share similar environmental characteristics, site productivity, and fire regimes. The main forested vegetation groups in the study area are warm, dry sub-alpine fir and persistent lodgepole pine, accounting for 65 % of the landscape (Figure 1). Spatial data on forest cover type, tree size class and percent of canopy cover were previously developed for the Forest Service from Landsat data (Redmond *et al.* 1997). These data represent current forest structural conditions and were used to initialize simulations (Table 3). These cover-type data and the PVG data were checked for consistency. Where there was disagreement (i.e., a cover type that is not expected to occur within a PVG), the PVG assignment for a 30 m pixel was changed to a compatible PVG.

Data about the cover type and PVG classifications were used to generate a spatial polygon coverage in ArcGIS 9.1 (ESRI Redlands, CA). Each polygon might be thought of as representing a forest stand. The resulting forest polygons ranged in size up to 535 ha. Because fires might be expected to affect only partial forest polygons, these polygons were further subdivided so that fires could be simulated at a finer resolution. I used TELSA's tessellation procedure (Okabe *et al.* 1992) to divide forest polygons into sufficiently small simulation polygons. For this 49,532 ha landscape, 11,367 simulation polygons were created, averaging 4 ha in size.

Table 2. Transition probabilities for PVGs 7 and 10.

State class (Tree size[1]/canopy closure[2])	Annual probability		
	Stand-replacing fire	**Non-lethal fire**	**All fire**
PVG7			
Seedling/open	0.00162	0.00062	0.00224
Sapling/low	0.00205	0.00020	0.00225
Sapling/moderate	0.00205	0.00020	0.00225
Small/low	0.00205	0.00020	0.00225
Small/moderate	0.00205	0.00020	0.00225
Small/high	0.00225	0.00000	0.00225
Medium/low	0.00184	0.00041	0.00225
Medium/moderate	0.00305	0.00020	0.00325
Medium/high	0.00325	0.00000	0.00325
Large/low	0.00184	0.00041	0.00225
Large/moderate	0.00225	0.00020	0.00245
Large/high	0.00425	0.00000	0.00425
PVG10			
Seedling/open	0.00134	0.00066	0.00200
Sapling/low	0.00134	0.00066	0.00200
Sapling/moderate	0.00134	0.00066	0.00200
Small/low	0.00167	0.00033	0.00200
Small/moderate	0.00184	0.00016	0.00200
Small/high	0.00184	0.00016	0.00200
Medium/low	0.00200	0.00000	0.00200
Medium/moderate	0.00200	0.00000	0.00200
Medium/high	0.00200	0.00000	0.00200

[1] Tree size class descriptions as in Tables 1 and 3.
[2] Canopy closure classes as percent of non-overlapping canopy: open (<10 %), low (10 % to 39 %), moderate (40 % to 69 %), high (>69 %).

Digital fire-polygon data compiled from 11 national forests and two national parks were used to derive the fire regime parameters used by TELSA (Gibson 2006). The temporal extent of these data varies for each reporting unit. For example, reliable data for several national forests date back to 1900, whereas the Boise National Forest portion extends only from 1908 to 2003 because records were not kept before 1908. To parameterize the fire size distributions used by TELSA, I used all the digital polygons in the regional atlas from 1900 to 2003 (Table 4). To inform the parameterization of inter-annual variability in area burned, I used the fire atlas to derive annual area burned on just this 49,532 ha landscape from 1908 to 2003. The inter-annual variability in area burned on the landscape has been high. Only a few years are responsible for most of the cumulative area burned, and in many years, no fires were recorded at all. These yearly data were used to derive randomized time series of

Table 3. Current landscape composition and initial conditions for simulations.

Potential Vegetation Group	Total Area (ha)	Area in tree size class[1] (% of area)				
		Seedling >1.4 m tall	Sapling 2.5-12.6 cm dbh	Small 12.7-30.5 cm dbh	Medium 30.6-50.8 cm dbh	Large >50.8 cm dbh
PVG 2 Warm, dry Douglas-fir/moist ponderosa pine	1,064	0	22	35	26	17
PVG 4 Cool, dry Douglas-fir	3,339	1	19	56	19	5
PVG 7 Warm, dry subalpine fir	17,082	24	8	48	16	4
PVG 8 Warm, moist subalpine fir	3,298	3	13	58	22	3
PVG 10 Persistent lodgepole pine	15,316	6	11	56	28	-
PVG 11 High elevation sub-alpine fir	3,464	0	35	55	3	6
Forest total	43,563	12	13	52	20	3
Non-forest (includes rock, water and bare ground)	5,969	-	-	-	-	-
Total	49,532	-	-	-	-	-

[1] English equivalents for tree size class descriptions as in Table 1.

Table 4. Fire size class distribution derived from the regional fire atlas used in TELSA simulations

Maximum size (ha)	Number of fire perimeters in the regional atlas	Percent of all fire perimeters
1	245	2.42
10	2126	20.99
100	4209	41.55
1,000	2730	26.95
10,000	733	7.24
100,000	86	0.85

fire probability multipliers that are then used by TELSA to adjust the fire probabilities in the succession models for each of the PVGs. Five different time series of multipliers defined the simulation scenarios described below.

Simulation Scenarios

Five different fire regimes were represented by five simulation scenarios. Each scenario was created from a randomly generated 1,000 year time series of probabilities. The first was based on the 96 year (1908 to 2003) record of area burned for this landscape. In this scenario, fires were allowed to occur in 121 of the 1,000 years. The remaining four scenarios represent increasing frequency of fires, as well as increasing frequency of large fire-years (years in which more than 5 % of the landscape burns). Ten replicates of each of the five scenarios were simulated. These replicates were not meant as a Monte Carlo simulation or to test for significant differences among scenarios, but rather as a means to illustrate variability resulting from the stochastic processes in TELSA that select the location and size of individual fire events. Each simulation was run with an annual time step for 1,000 years. Fire information was output by the model every year and forest state class information was output every ten years.

Analysis

Model output describing forest state class information was grouped into the five tree size classes for each PVG (four tree size classes for PVG 10): seedling (1.4 m height), sapling (2.5 cm to 12.6 cm dbh), small tree (12.7 cm to 30.5 cm dbh), medium tree (30.6 cm to 50.8 cm dbh) and large tree (>50.8 cm dbh). To compare the simulated distribution of tree size classes to the desired landscape condition outlined in the land management plan, Euclidean Distance (ED) was computed as:

$$ED = \sqrt{\sum_{j=1}^{6}\sum_{i=1}^{5}(x_{ij} - y_{ij})^2} \qquad (1)$$

where x_{ij} is the simulated land area (in hectares) in tree size class i and PVG j and y_{ij} is the desired land area in tree size class i and PVG j. Because the land management plan provided a range of values for the desired condition for each tree size class, I used the midpoint of each range to compute ED. As computed here, ED has units of area (hectares). High values of ED indicate large departures from the desired condition, and an ED of 0 indicates that the distribution of size classes is identical to the desired distribution.

Spatially explicit information on tree size class was extracted every 100 years for each of the simulations (5 scenarios, 10 replicates each). For each scenario, this resulted in 100 maps that were then rasterized in ArcGIS 9.1 for input to the spatial analysis program FRAGSTATS (McGarigal and Marks 1995). Two metrics of landscape pattern were computed: landscape contagion (CONTAG) and mean patch size of the large tree size class. Contagion refers to the tendency of patch types to be spatially aggregated. Measured in percent, the index approaches its maximum value of 100 when there is only one patch type. A second set of maps was created to compute mean patch size of large trees. Because there is no large tree size class for PVG 10, the medium tree size class for PVG 10 was reclassified and added to the large tree size class before analysis with FRAGSTATS.

RESULTS

Simulated fire regimes for each scenario were summarized and compared to the fire atlas record for this landscape (Table 5). The rate of burning simulated by Scenario 1 approximates the rate of burning this landscape experienced from 1908 to 2003. The other scenarios represent varying rates of burning, with Scenario 3 representing more than twice as much fire as Scenario 1, and Scenario 5 representing seven times as much fire. Large fire-years (years in which more than 5 % of the landscape burns) occurred seven times more often in Scenario 5 than in Scenario 1.

Area occupied by each of the five tree size classes was averaged over the 1,000 year simulations and compared among scenarios (Table 6). The result of increasing fire was more area occupied by the smaller size classes and less area occupied by the larger size classes. In other words, the scenarios with more fire resulted in a younger forest comprising earlier successional states. To illustrate the successional and fire dynamics simulated by the model, area occupied by each of the five tree size classes was plotted at 10 year intervals for Scenarios 1, 3, and 5 (Figure 3). Scenarios 2 and 4 were not plotted for purposes of readability. Smaller size classes (e.g., seedling and sapling) exhibited higher variability than larger tree size classes (e.g., medium and large tree). This variability was much wider than the desired ranges defined in the land management

Table 5. Summary of simulated and recorded fire regimes.

Scenario	Number of fire-years [2]	Number of large-fire-years [1,2]	Mean fire interval (yrs)	Standard deviation fire interval (yrs)	Median fire interval (yrs)	Maximum fire interval (yrs)	Mean annual area burned (ha) [1]	Standard deviation annual area burned (ha) [1]	Total area burned (ha) [1,2]	NFR (yrs) [1]
1	121	16	8.03	7.17	6	39	185	1,358	184,787	236
2	252	25	3.96	3.24	3	17	326	1,722	325,666	134
3	269	60	3.67	2.97	3	17	453	1,527	453,293	96
4	331	86	3.03	2.41	2	19	947	3,139	947,174	46
5	394	120	2.54	1.97	2	13	1,720	3,987	1,309,972	33
Fire atlas	125	21	6.45	6.11	5	22	181	1,288	180,750	241

[1] Values for simulation scenarios are average of 10 simulations.
[2] Values for fire atlas are based on data from 96 year record (1908 to 2003) standardized to 1,000 years.

Table 6. Summary of simulated tree size classes and Euclidean Distance from desired conditions. Values are mean of 10 replicate simulations, averaged over 1,000 years of simulation.

		Area occupied by tree size classes (ha)				
Scenario	Euclidean distance (ha)	Seedling >1.4 m tall	Sapling 2.5-12.6 cm dbh	Small 12.7-30.5 cm dbh	Medium 30.6-50.8 cm dbh	Large >50.8 cm dbh
1	13,582	5,469	2,534	5,992	14,735	14,833
2	10,233	8,703	3,770	8,513	12,684	9,893
3	7,882	11,219	4,969	10,059	10,844	6,473
4	10,349	17,634	6,838	10,744	6,196	2,152
5	11,764	21,783	7,698	9,560	3,674	848

[1] English equivalents for tree size class descriptions as in Table 1.

plan for the forest at large. As a consequence, the simulated landscape composition often fell outside these desired ranges in all scenarios.

The overall departure from the desired forest composition was computed using Euclidean Distance and plotted at 10 year intervals for Scenarios 1, 3 and 5 (Figure 4). Euclidean Distance varied throughout the 1,000 year simulation, with more variability occurring in Scenario 1 than in Scenarios 3 or 5. In general, Scenario 3 resulted in lower ED, indicating that forest composition was closer to the desired conditions than in the other scenarios. ED for Scenario 3 averaged 7,882 ha over the 1,000 years, while for the other scenarios it averaged from 10,233 ha to 13,582 ha (Table 6). Visual inspection of Figure 4 reveals that in Scenario 1, fluctuations in ED are inversely related to fluctuations in area burned while in Scenario 5, ED more directly tracks area burned.

To further elucidate the relationship of area burned to fluctuations in ED, I examined the change in ED in 10, 20, or 50 years relative to the initial ED and the cumulative area burned in the same time period (Figure 5). For low initial values of ED (e.g., <7,500 ha), the

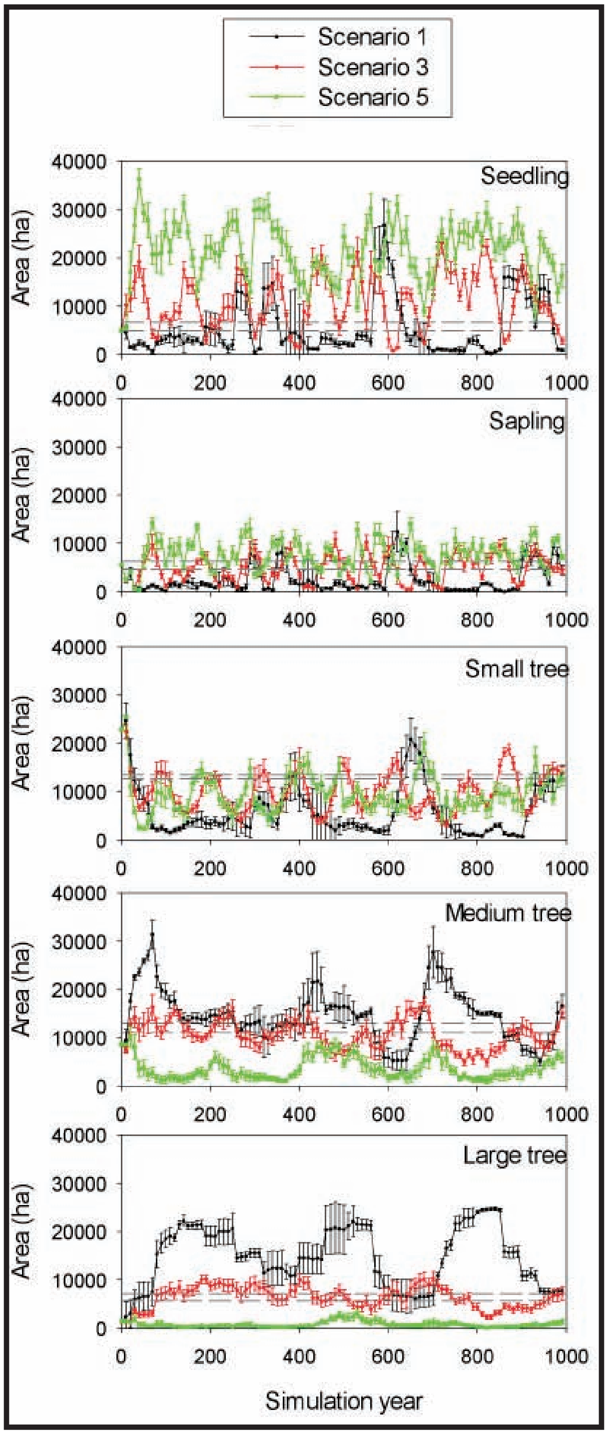

Figure 3. Area occupied by forest in each tree size class versus simulation year for Scenarios 1, 3 and 5. Year 0 represents current conditions. Average of 10 replicate simulations shown, error bars reflect standard deviation among 10 replicates. Horizontal dashed lines represent the low and high values of the range specified as desired conditions.

change in ED is almost always away from desired conditions, depicted by red symbols on the bubble plot. For higher initial values of ED, the direction of ED change depends on the cumulative area burned.

Differences among scenarios were apparent for landscape contagion and patch size of large trees (Figure 6). Landscape contagion was lowest for Scenario 3. The patch size of large trees decreased with increasing amounts of fire, with the smallest patches resulting from Scenario 5.

DISCUSSION

Desired conditions described in the Boise National Forest plan were informed by HRV estimates from modeling results that were based originally on expert opinion of historical fire intervals for PVGs in the region (Morgan and Parsons 2001). Historical fire intervals may not apply in the future, however. A variety of factors including climate variability and change, exotic plants, increased fuel continuity, as well as adoption of fire management strategies such as WFU, will most likely conspire to increase the area burned on this landscape in the future. Simulation modeling can help determine whether this increase in fire will assist or hinder attainment of desired conditions.

Simulation results for Scenario 1 suggest that the rates of burning that occurred throughout the 20th century on this landscape are insufficient for meeting land management goals. In Scenario 1, fires were infrequent and much of the landscape became occupied by the oldest forest age class (i.e., large tree size class) (Figure 3). The long intervals between fires did not allow for the creation of younger age classes and smaller tree size classes on the landscape. Scenario 3, with twice the amount of fire than Scenario 1, resulted in a younger forest with smaller trees, and thus one that was

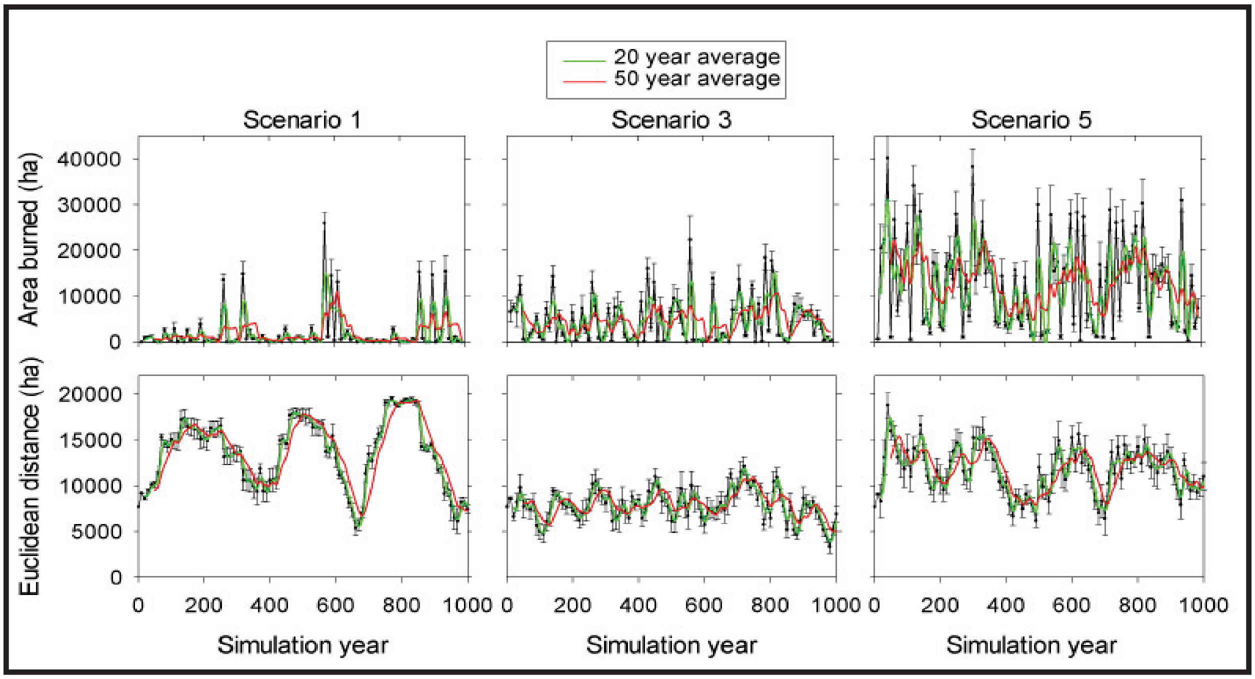

Figure 4. Area burned and Euclidean Distance for Scenarios 1, 3 and 5 in 10-year intervals. Year 0 represents current conditions. Average of 10 replicate simulations shown, error bars reflect standard deviation among 10 replicates. Twenty-year and 50-year running averages displayed in green and red, respectively.

closer to desired conditions. Scenario 5, with the most fire, resulted in almost complete loss of the large tree size class, a desired component of the landscape (Figure 3). Such patterns are expected and have been demonstrated in other simulation studies (e.g., Fall *et al.* 2004, Scheller *et al.* 2005, Schoennagel *et al.* 2006, Didion *et al.* 2007). It should be noted that none of the five scenarios resulted in a landscape composition that consistently fell within the desired range of conditions. The forest plan for the Boise National Forest defined the desired forest composition in terms of ranges that were intended to be used on a much larger, forest-wide basis. The variability seen in the simulation results suggest that these ranges are too narrow to be useful for a landscape of this size with these fire dynamics (Keane *et al.* 2002, Pratt *et al.* 2006). Simulation modeling should be employed to adjust the desired conditions developed for the forest as a whole to those that are meaningful at the scale of an individual landscape.

Euclidean Distance was used to measure departure from desired forest composition at the spatial extent of this landscape. As a composite metric, it captured departures for each of the five tree size classes and provided a way to compare the implications of different fire regimes. Although ED fluctuated widely throughout the 1,000 year simulation period, ED for Scenario 3 reflected landscape composition that was more similar to desired conditions than the other scenarios. ED also provided a way to track landscape composition through time and highlighted the effect that years with numerous fires can have in shifting forest composition toward desired conditions (Figure 4). Other dissimilarity or similarity metrics (e.g., Sorensen's Index) should be equally useful as ED (Mueller-Dombois and Ellenberg 1974, Holsinger *et al.* 2006).

Simulation results also illustrated the feedback between forest composition and fire that results in an increase or decrease in the departure from desired conditions.

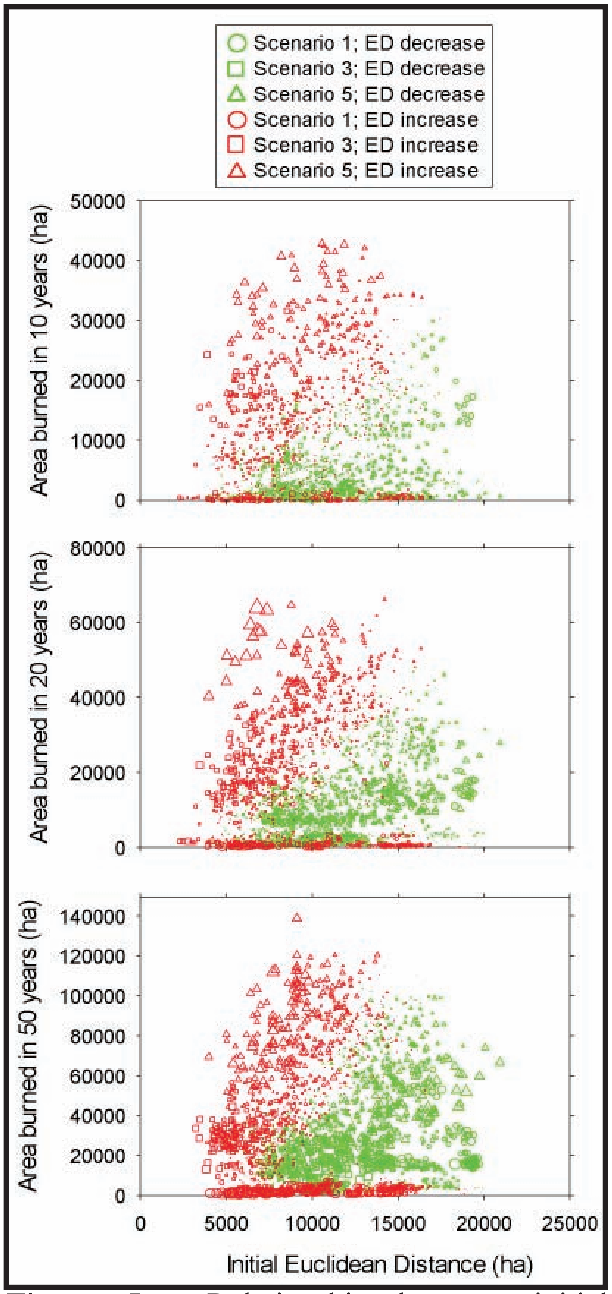

Figure 5. Relationship between initial Euclidean Distance, cumulative area burned in the subsequent 10, 20 and 50 years, and change in ED during the same time period. Red symbols indicate an increase in ED; green symbols indicate a decrease in ED. The size of the symbol is proportional to the magnitude of ED change.

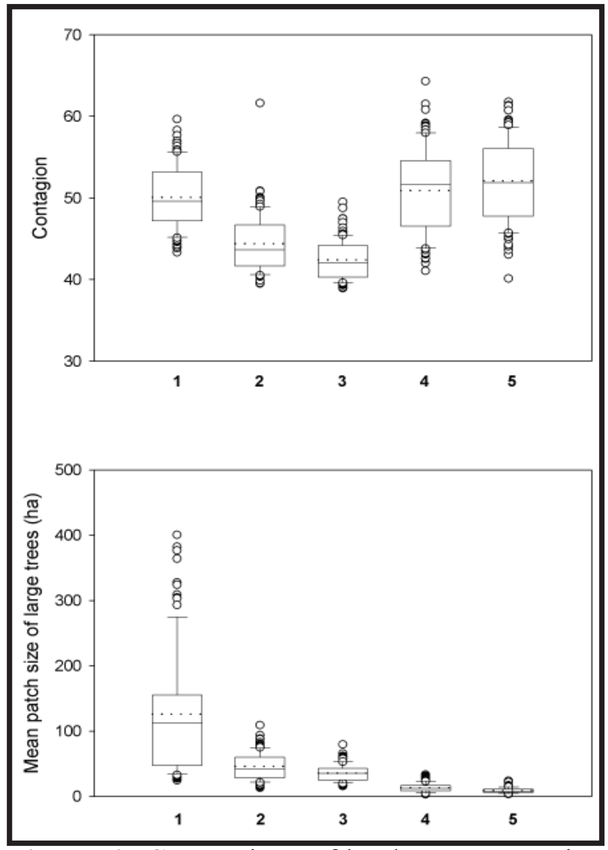

Figure 6. Comparison of landscape contagion and mean patch size of large tree size class occurring at 100-year intervals for 10 replicates each of the five simulation scenarios. Dotted lines in the box plots represent mean values.

This relationship varied among scenarios. Landscape composition in Scenario 1 became closer to desired conditions only after significant area burned. In Scenario 5, on the other hand, landscape composition appeared to move away from desired conditions in response to fire, suggesting that this scenario's natural fire rotation of 33 years is too much fire for achieving desired conditions. In wilderness areas and parks, there has been little discussion about how much fire is appropriate, probably because any and all lightning-ignited fire is viewed as being commensurate with land management goals. Departure indices such as ED could be calculated for current landscape composition and the thresholds in area burned that appear in Figure 5 might be used to help identify 10, 20, or 50 yr fire management targets. Each WFU-candidate fire might be viewed as an opportunity to reach this target. Measurable targets for area burned would be attractive because with WFU, land managers do have some control over when and how much fire can occur. However, this control is limited—climate is a powerful driver of wildland fire regimes and every year there are examples of ineffective suppression due to fire behavior in dry, windy conditions. This could increasingly become the case if predictions for future climates are borne out (McKenzie *et al.* 2004, Westerling *et al.* 2006).

I examined the effect of fire on the distribution of tree size classes because that was the most clearly articulated and quantifiable set of goals in the forest plan, and because forest age structure can be considered a valuable integrator of other ecosystem attributes (Didion *et al.* 2007). In addition to forest age structure or tree size class distributions, land managers need to manage a variety of other resources including wildlife populations, recreation opportunities, and watershed conditions (Hyde *et al.* 2006). Tree size class distributions probably are not adequate for describing landscape composition for all these purposes. However, methods do exist for cross-walking successional state classes to other resource values so that other resource management goals might be considered (Black and Opperman 2005). Furthermore, to effectively manage resources such as wildlife populations, an understanding of landscape pattern is necessary (e.g., Bender *et al.* 1998). Although the Boise National Forest plan does not specify desired conditions for landscape structure, metrics such as those presented in Figure 6 can be computed from simulation output and used to help develop prescriptive guidance. I examined only two aspects of landscape pattern (landscape contagion and patch size of the large tree size class); the use of additional metrics should be explored that may better address particular resource concerns.

Model results need to be interpreted in light of the limitations and simplifying assumptions made in the formulation and parameterization of TELSA. The state-and-transition models for each PVG assumed deterministic succession that proceeds at a fixed rate. The probabilities for the state transitions due to fire depended only on the current successional state and ignored previous history. Vegetation was classified at the PVG level, and so species-specific dynamics and responses were not simulated. The understanding of forest dynamics with which these models were developed was based largely on observations throughout the 20th century. This is a short slice of time, especially for the long-interval stand-replacing fire regimes these subalpine forests experience. TELSA is unable to address species-level response to climatic variation. As warming trends continue, novel environmental conditions could preclude certain species from re-establishing after a fire, invalidating the successional pathways used here. This is an inherent limitation of the state-and-transition modeling approach.

Future simulations could be designed to examine the effect of fire regimes on landscape pattern in more than the cursory manner done here. Effects of wind and topography on fire spread were not included in the simulations, but these effects could alter the distribution of fires and tree size class on the landscape (Wimberly 2002). If older, large trees are more persistent in areas that are topographically protected from fire, this is important information for land managers who are challenged with maintaining this forest component on the landscape. Landscape size and shape have been shown to affect estimates of fire rotation and landscape pattern in other simulation studies due to boundary effects of not including the immigration of fires from outside the landscape (Wimberly *et al.* 2000, Keane *et al.* 2002). Finally, although non-forested PVGs occupy a small portion of this landscape, their succession and fire dynamics should be included in future simulations. Including non-forested PVGs would affect where fires are allowed to start and spread, thus affecting the rate at which adjacent forested PVGs experience fire.

Scenario 3 resulted in a distribution of tree size classes that was closer to desired conditions than the other scenarios, which might suggest to land managers an optimal fire regime for achieving land management goals. It should be noted that this is more than twice the rate of burning experienced by this landscape between 1908 to 2003. However, because I simulated only fires, these results ignore the existence of other change agents on this landscape that could contribute to cumulative effects on forest age structure (Fall *et al.* 2004). The effects of fire on the age class or tree size class of the landscape must be considered in conjunction with other ecological processes, particularly stand-replacing change agents. For example, mountain pine beetle can cause extensive tree mortality and increased frequency and magnitude of outbreaks are expected in the northern Rocky Mountains (Logan and Powell 2001). Potential synergies between fire, insect outbreaks, and climate change could lead to further compounding of change rates (*sensu* Paine *et al.* 1998).

CONCLUSION

Every decision about fire is a land management decision that affects the landscape in its own way. Furthermore, each landscape is unique. As WFU is increasingly applied to lands with multiple non-wilderness objectives, managers will need to know how it will affect their ability to meet land management goals. Simulation models such as TELSA are useful as planning tools and can be used to help managers anticipate landscape-scale consequences of different fire regimes. In the simulations I conducted, a doubling of the area burned over 20th century levels resulted in tree size class distributions that were much closer to desired conditions, suggesting that for this landscape, land management goals may be more easily achieved with such an increase in fire. Simulation results also indicated an upper limit to the desired amount of fire. Euclidean Distance proved to be useful for measuring departure from desired conditions and could also prove useful for developing targets for area burned.

ACKNOWLEDGEMENTS

Carly Gibson developed the fire atlas data for the northern Rocky Mountains, Kathy Geier-Hayes helped me select the Bear Valley-Elk Creek study area, Carey Crist provided the spatial data I needed, Jim Merzenich helped me correctly interpret and use the VDDT models, and Leonardo Frid provided training and advice on TELSA. TELSA is developed by ESSA Technologies Ltd. and can be downloaded from www.essa.com. This project was co-funded by the Joint Fire Sciences Program (project #03-1-1-07) and National Fire Plan. Brett Davis and two anonymous reviewers provided valuable comments on an earlier draft that improved my thinking and the manuscript.

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