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Integrated analysis of landscape management scenarios using state and transition models in the upper Grande Ronde River Subbasin, Oregon, USA

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Abstract

We modeled the integrated effects of natural disturbances and management activities for three disturbance scenarios on a 178,000 ha landscape in the upper Grande Ronde Subbasin of northeast Oregon. The landscape included three forest environments (warm-dry, cool-moist, and cold) as well as a mixture of publicly and privately owned lands. Our models were state and transition formulations that treat vegetation change as probabilistic transitions among structure and cover types. We simulated background natural disturbance (i.e., historical), active fuel treatment, and fire suppression only disturbance scenarios for 200 or 500 years, depending on scenario. Several interesting landscape hypotheses emerge from our scenario simulations: (1) changes in management approach in landscapes the size of our study area may take decades to play out owing to the time required to grow large trees and the feedback loops among disturbances, (2) the current landscape is considerably different from that which might exist under a natural disturbance regime, (3) fire suppression alone does not mimic background natural disturbances and does not produce abundant large tree structure, and (4) dense, multi-layered large tree forests may be particularly difficult to maintain in abundance in this and similar landscapes owing to wildfire and insect disturbances.

Keywords: Landscape models; Landscape ecology; Historical range of variability; Forest structure; Forest disturbance; Pacific northwest; Interior northwest landscape analysis system

1. Introduction

Many questions regarding the management of diverse landscapes in the interior Pacific Northwest involve the combined effects of natural disturbances and management activities on natural resource conditions. For example, how will fuel treatment activities change wildfire occurrence and severity across large landscapes, and what effect will these treatments have on other resources? Are current vegetative conditions and associated wildlife habitat characteristics sustainable? If existing vegetation were allowed to develop with either no management, or with fire suppression only, how would this compare with historical conditions? When considering management alternatives for a particular landscape, what are the long-term effects of each alternative on the vegetation?

Landscape simulation models address questions regarding the reaction of large landscapes to various management and policy scenarios (Bettinger et al., 1997, 1998; Hann et al., 1997; Mladenoff and He, 1999; Graetz, 2000; USDA and USDI, 2000). Advances in modeling techniques, computer technology, and geographic information systems (GIS) have made it possible to model large landscapes at increasingly finer scales of spatial and temporal resolution (Barrett, 2001). In the past, resource planning models have focused primarily on conifer succession and management while representing other ecosystem elements as byproducts (e.g., Johnson et al., 1986; Alig et al., 2000). Although progress has been made in the formulation of multi-objective goals in landscape simulations (e.g., Sessions et al.,

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1999; Wedin, 1999), there remain many challenges to building landscape planning models that include all of the important disturbance processes that influence change. For example, previous efforts have often not included widespread, chronic disturbances such as ungulate herbivory. Of particular interest are the net, synergistic effects of various disturbances (e.g., fire, insects, management activities, and large herbivores) across a large ecologically diverse landscape. Our approach treats vegetation as discrete types and management activities and natural disturbance as transitions among those types to project the long-term net effects of alternative management scenarios across a large landscape.

2. Study area

The upper Grande Ronde Subbasin occupies approximately 178,000 ha of mixed forest and rangelands on the eastern flank of the Blue Mountains southwest of La Grande, Oregon, USA (Fig. 1). The majority of the area (122,114 ha) is managed by the USDA Forest Service with the remaining land in mixed ownerships. Most of the remaining land is in private ownership (53,551 ha), with smaller amounts of tribal (1373 ha), and state (885 ha) lands. The topography is varied and complex, with deeply dissected drainages feeding into the Grande Ronde River as it runs north through the center of the area. Vegetation ranges from dry bunchgrass-dominated communities at the lower, north end of the drainage, to high-elevation conifer forests at the southern end (Johnson and Clausnitzer, 1992). Elevations range from 360 to over 2100 m.

The current disturbance regime is driven by occasional large wildfires, insect outbreaks, and recent land management. A number of wildfires burned about 16,000 ha (9% of the watershed) in the last 10 years. Outbreaks of western spruce budworm (*Choristoneura occidentalis*), bark beetles (*Dendroctonus* spp.), and Douglas-fir tussock moth (*Orgyia pseudotsugata*) over the last several decades have caused extensive mortality to Douglas-fir (*Pseudotsuga menziesii*) and grand fir (*Abies grandis*) (USDA Forest Service, 1980–2000; Hayes and Daterman, 2001; Torgersen, 2001). Extensive timber harvest has occurred in much of the area, including clearcut, shelterwood, selection, commercial thinning, precommercial thinning, and fuel treatments.

The upper Grande Ronde Subbasin potentially contains habitat for three wildlife listed as threatened or endangered under the Endangered Species Act: the Canada lynx (*Lynx canadensis*), the gray wolf (*Canis lupis*), and the American bald eagle (*Haliaeetus leucocephalus*). In addition, Wisdom et al. (2000) identified 40 additional terrestrial vertebrates of concern likely to occur in the upper Grande Ronde Subbasin. There are also several threatened or endangered

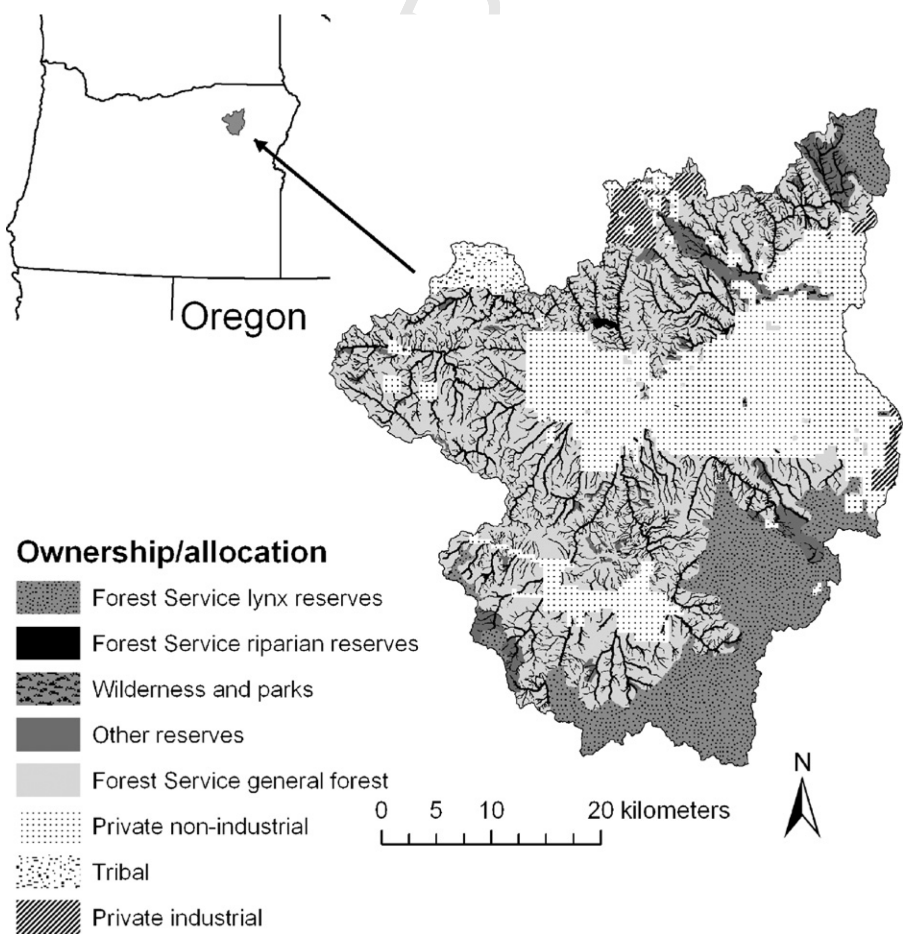

Fig. 1. The upper Grande Ronde Subbasin study area in northeast Oregon, USA.

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aquatic species at risk within the area (USDA and USDI, 2000).

Forest Service management in the area includes wilderness (no active management), riparian areas (managed to protect water quality and aquatic habitat), lynx habitat management areas, and general forest (managed for a variety of goods and services). Private lands tend to be managed for timber production and livestock forage, though this varies considerably by ownership.

3. Methods

Our approach projects the effects of natural disturbances and management treatments on vegetation by using state and transition models (STMs) (Fig. 2). The vegetative composition and structure defines each "state". These states are connected by transitions that indicate either the effect of successional vegetation development over time, or the effect of disturbance (Hemstrom et al., 2004). This approach builds on transition matrix methods that represent vegetation development as a set of transition probabilities among various vegetative states (e.g., Horn, 1975; Cattelino et al., 1979; Noble and Slatyer, 1980; Westoby et al., 1989; Laycock, 1991; Keane et al., 1996; Hann et al., 1997). For example, grass/forb-closed herblands might become dominated by small trees and shrubs after a period of time or might remain as grass/forb communities following wildfire. State changes along the successional, time-dependent paths are deterministic, and without disturbance or management, all the vegetation would ultimately accumulate in one state. Because disturbances or management activities can change the course of vegetative development at any point, very little or no vegetation may actually accumulate in the state representing the end point of succession.

We used the Vegetation Dynamics Development Tool (VDDT; Beukema et al., 2003) modeling program to project vegetation and disturbance conditions. This is a non-spatial model that allows building and testing STM for a set of environmental strata. It has been used in several landscape assessments and land management planning efforts in the interior northwestern United States (e.g., Keane et al., 1996; Hann et al., 1997; Merzenich et al., 2003). We also built spatially explicit versions of the VDDT models by using the Tool for Exploratory Landscape Scenario Analysis (TELSA; Kurz et al., 2000).

3.1. Vegetation data and state classes

Most of the vegetation data were developed by the Wallowa-Whitman and Umatilla National Forests and are typical of the kind used by national forests and other land managers in the Blue Mountains. Stand boundaries were delineated on 1:24,000 aerial photographs. Stand attributes were assigned based on aerial photo interpretation or field stand examinations. We also acquired vegetation data from private industrial forest land from the landowner, also developed from aerial photograph

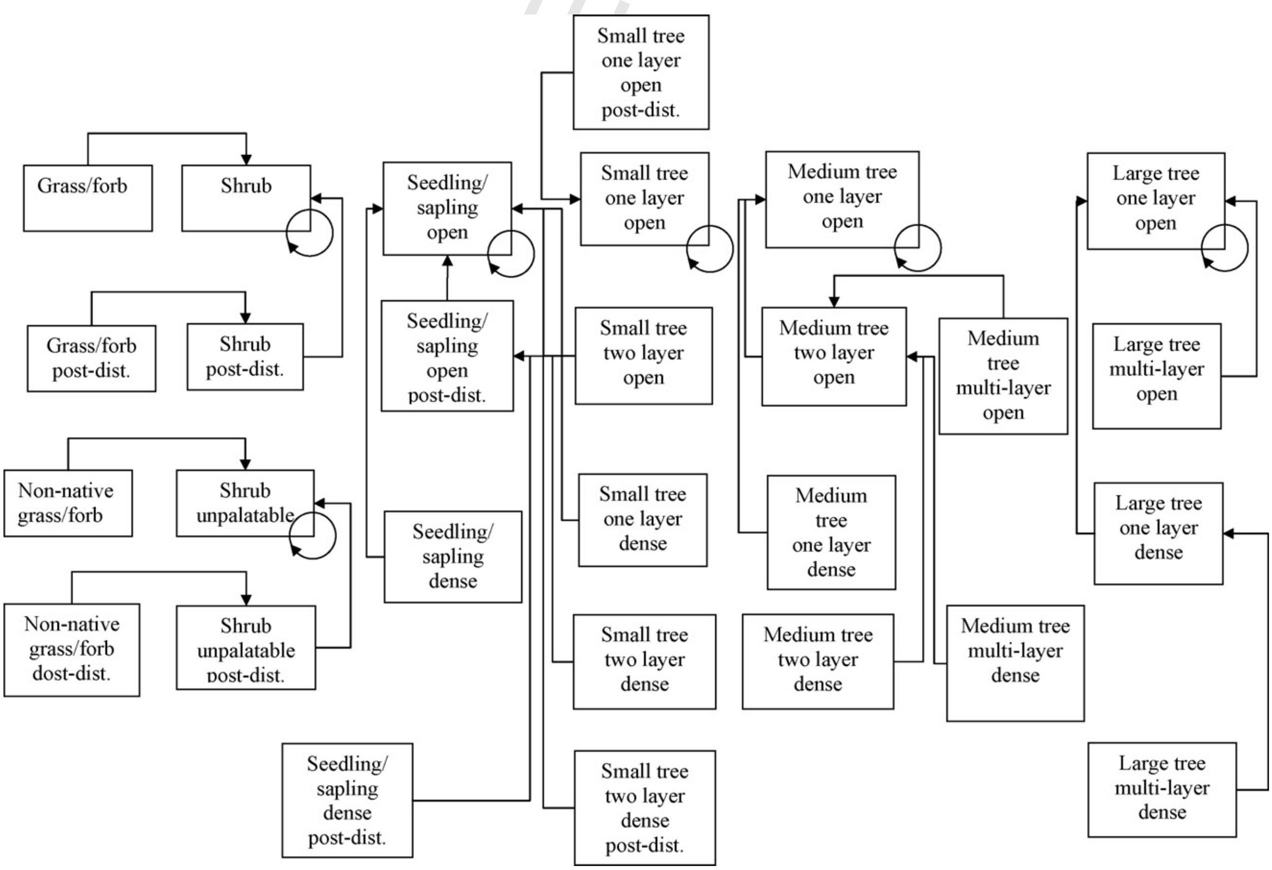

Fig. 2. Example state and transition model for surface and mixed-severity wildfire in warm-dry environments.

interpretation and field stand exams. Data that were particularly useful included tree species listed in order of abundance by canopy layer, tree size classes (diameter breast height—dbh) by canopy layer, total canopy cover, potential vegetation type, and life form (grass/forb and shrub) in non-forest vegetation. Forest structure classes are based on tree size, stand density (canopy coverage), and the presence of a single or multiple canopy layers. Separate classes also identify post-disturbance conditions created by high-severity wildfire and insect outbreaks. Attributes interpreted from aerial photographs and field stand examinations were used to identify the structure class. Six vegetation structure classes were based on the presence or absence of trees and the average dbh of dominant trees: (1) grass/forb dominated, (2) shrub dominated, (3) seedlings/saplings–dominant trees <12.5 cm dbh, (4) small trees–dominant trees 12.5 to <40 cm dbh, (5) medium trees–dominant trees 40 to <52.5 cm dbh, and (6) large trees–dominant trees ≥52.5 cm dbh. Tree canopy cover was divided into three classes: (1) tree canopy <15% cover was classified as grass/forb or shrub dominated, (2) tree canopy 15% to <40% (warm-dry forests) or 15% to <60% (cool-moist and cold forests) was open forest, and (3) tree canopy ≥40% (warm-dry forest) or ≥60% (cool-moist and cold forest) was dense forest. Finally, tree-dominated stands were divided into those with one or more than one canopy layers. Results presented in this paper were summarized using tree canopy layer classes rather than tree canopy cover classes, combining all canopy cover classes within single layered versus multi-layered forest structures.

Local land managers and ecologists often use potential vegetation to identify environment, disturbance regimes, and vegetation growth potential (Johnson and Clausnitzer, 1992). For the purposes of this analysis, we used only land areas that had forested potential vegetation types (about 80% of the landscape). We grouped potential vegetation types into three major forest environments (cold, cool-moist, and warm-dry) based on the potential natural vegetation classification by Johnson and Clausnitzer (1992). Cold forest environments comprise about 27% of the forest landscape. Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) dominate older forests in these environments, and lodgepole pine (*Pinus contorta*) frequently occurs following high-severity disturbances. Cool-moist forest environments occur at intermediate elevations and comprise approximately 30% of the forest landscape. Mixed forests of grand fir and Douglas-fir dominate older cool-moist stands, whereas western larch (*Larix occidentalis*), lodgepole pine, and ponderosa pine (*Pinus ponderosa*) dominate early seral stands. Warm-dry forests occupy about 42% of the forested land in the study area. Because of large variability in productivity and site potential, three VDDT models were used to represent warm-dry forests. These are distinguished with a dry site ponderosa pine model, a dry site Douglas-fir model, and a dry site grand fir/Douglas-fir model. Ponderosa pine is especially drought tolerant and occurs on the warmest and driest sites capable of supporting forests. It is also tolerant of the frequent surface fires that historically occurred on warm-dry sites. As a consequence, early seral ponderosa pine forests historically dominated warm-dry sites.

We used combinations of structure class (tree size, canopy cover, canopy layering), overstory species, disturbance history, and potential vegetation to assign the vegetation to 308 state classes that are included in our models. We did not include lands that do not potentially support forests in our models owing to lack of information about their fire and disturbance regimes.

3.2. Disturbances, transitions, and probabilities

Our models derive from those that Hann et al. (1997) developed for use in a broad-scale assessment of the interior Columbia River Basin. Their models were designed for use across very large landscapes (over 58 million ha) and with coarse-resolution data (1-km pixels). Our modifications are based on discussions with field managers, other experts, and the existing literature to allow better fit to higher-resolution vegetation data and more complex, localized transitions and state classes. Our models incorporate disturbances for wildfire, insect and disease agents, grazing by ungulates (deer, elk, and domestic cattle), stand growth and development processes, and various management treatments. Discussion and results of our ungulate grazing models are presented by Vavra et al. (this volume). In addition, probabilities for disturbances and treatments varied for several land allocation/ownership combinations: wilderness (national forest lands with no active management), riparian areas (national forest lands with low levels of silvicultural and fuels management to maintain water quality and aquatic habitat), lynx management areas (national forest lands managed to provide denning and foraging habitat for Canada lynx), general forest (national forest lands managed for a variety of goods and services), private industrial lands (private lands owned by large, industrial companies managed primarily for timber production), and private non-industrial lands (private lands owned by various owners managed less intensively for timber production).

3.2.1. Management treatments

Forest management activities included in the model were shelterwood harvest, group selection harvest, commercial thinning, pre-commercial thinning, mechanical fuel treatments, and prescribed fire. The annual probabilities for each of these were developed separately for cold forests, cool-moist forests, and warm-dry forests and were adjusted to reflect on-the-ground treatment rates within each structural stage and land ownership/allocation. We used a consensus process with local field experts (including those working on private industrial forest lands) to estimate the probabilities for each kind of management treatment by forest environment and scenario and the resulting change in state class. For example, we asked what change would occur in closed canopy lodgepole pine stands in cold forest environments as a result of shelterwood harvest in an active fuel treatment scenario. We considered prescribed fire to be a management activity.

3.2.2. Wildfire disturbances

We distinguished high-severity (e.g., stand replacement) from surface (a combined category of mixed-severity and low-severity fires) wildfires (Hessburg and Agee, 2003). In general,

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high-severity disturbances killed 75% or more of the overstory, mixed-severity disturbances killed 25–75% of the overstory, and low-severity fires killed less than 25% of the overstory. We had three sources of wildfire frequency and severity information: (1) data on actual fire occurrences over the last two to three decades that we could map and stratify by forest environment; (2) information, mostly on historical fire frequencies, from the literature; and (3) expert opinion from local fire managers. Unfortunately, data on fire occurrences do not include proportion by fire severity, so our estimates for proportion by severity class come from the opinions of local fire managers. Because our wildfire probabilities are based on recent fires and conditions in the study area, they reflect both the impacts of fire suppression on fire occurrence and severity and potentially enhanced rates of ignition from human activities.

3.2.2.1. Current fire probabilities. Our process for assigning current wildfire probabilities was to estimate a mean fire-return interval for all kinds of wildfire in each forest environment (warm-dry, cool-moist, cold) from data on actual fire occurrence and from the consensus of local fire managers. We stratified late seral warm, dry forest environments into those typically dominated by ponderosa pine, Douglas-fir, and a combination of Douglas-fir and grand fir (Johnson and Clausnitzer, 1992) because local fire managers thought fire frequencies and severities were substantially different in those strata. We then divided the annual probability for wildfires of any severity into probabilities for high-severity and surface wildfires (Table 1). Our division of high-severity versus surface wildfire probabilities reflected the opinions of local fire managers about what proportion of wildfires would occur in those two severity classes given combinations of forest environment, stand cover types, and stand structure. For example, local fire managers estimated that at least 94% of all wildfire in large tree, single-story, open-canopy, warm, dry forests dominated by ponderosa pine would be of surface severity at present (Table 1). This means that we assume high-severity wildfires to be uncommon in this forest type (<6% of all wildfires) under the current fire suppression regime. Given the estimate that current annual probability of all wildfires is 0.0111 (90-year average return interval) in this forest environment, high-severity wildfire was assigned an annual probability of 0.0004 (2250 years average return interval). This mean fire-return interval was more infrequent than we expected, but we had no other data on current fire occurrence by severity class in the study area. However, changing the annual probability for high-severity wildfire in this class to 0.002 (500-year average return interval) had minor effect on model outputs. Available data and the consensus of fire experts is that high-severity wildfire in single-storied, open canopy, large tree stands dominated by ponderosa pine is very rare at present. Fires are more often high severity in dense, multi-storied stands, especially of small trees (<40 cm dbh), and our wildfire probabilities reflect that tendency.

3.2.2.2. Historical fire probabilities. Mauroka (1994) found mean fire-return intervals of about 10–50 years in forest types where ponderosa pine is co-dominant with Douglas-fir and grand fir in the Blue Mountains. Heyerdahl et al. (2001) estimated that 90% of forests had mean fire-return intervals of <25 years in the southern half of the Blue Mountains, but only half had mean fire-return intervals of <25 years in the northern Blue Mountains. Our study area is mid-way between the northern and southern Blue Mountains as defined by Heyerdahl et al. (2001). High-severity fire was assumed by Heyerdahl et al. (2001) to have occurred in small (<0.4 ha) patches in warm-dry forests and at relatively infrequent, but unspecified, intervals that allowed development of large, old trees. They described wildfires on warm-dry sites as having been generally low in severity (i.e., surface). We assumed that our study area had a mean fire-return interval of surface wildfire on warm-dry sites of about 25 years or slightly longer, which matches well with this forest type in the Pacific Northwest (Agee, 1993). Because we had no local information on the return interval for high-severity wildfire in warm-dry environments, we assumed that high-severity fire was rare in open stands of large (e.g., ≥16 in. dbh) trees with a mean fire-return interval of 400 years or more (Table 1).

Heyerdahl et al. (2001) had difficulty in distinguishing wildfire from other disturbances in mesic forests near our study area. Apparently their study site had experienced many small-scale, non-wildfire disturbances over the historical period examined, perhaps mortality related to insect outbreaks. They found a median occurrence of one fire in the 1750–1900 time period in other mesic sites in northern Blue Mountain sites. Based on this information, we assumed that the overall fire-return interval in cool-moist forests was approximately 150 years prior to European settlement (Table 1).

Based on fire history studies in lodgepole pine, Engelmann spruce, and subalpine fir forests in other parts of the western United States (Agee, 1993), we assumed that fires in cold forests were generally high-severity events with a mean fire-return interval of about 200 years. We also assumed that open stands of large western larch, a fire-resistant species, had a longer mean fire-return interval for stand-replacement fire of 250 years and that dense stands of Engelmann spruce and subalpine fir, both fire-intolerant species, had a somewhat shorter mean fire-return interval for high-severity fire of about 190 years. We assumed the mean fire-return interval for surface wildfire was generally over 400 years.

3.2.3. Insect disturbances

Several insects may occur at endemic and outbreak levels in Blue Mountains forests (Ager et al., 2004), including Douglas-fir beetle (*Dendroctonus pseudotsugae*), Douglas-fir tussock moth, fir engraver (*Scolytus ventralis*), mountain pine beetle (*Dendroctonus ponderosae*), spruce beetle (*Dendroctonus rufipennis*), and western spruce budworm. Each species has specific preferences for host tree species and tree size as well as probability of occurrence and typical patch disturbance sizes and probabilities in the study area. We included endemic disturbances that reduce stand density but do not kill most of the susceptible host and outbreak disturbances that kill most or all the susceptible host over large areas. Outbreak insect disturbances were assigned an average duration and periodicity

Table 1
Assumed mean return intervals and mean annual probabilities for wildfires in large tree structure classes under historical conditions modeled in the upper Grande Ronde Subbasin, Oregon

Envt.[a]	CT[b]	S[c]	Current fire regime								Historical fire regime					
			All[d]		Surface[e]			High severity[f]			All		Surface		High severity	
			Prob.[g]	MRI[h]	Factor[i]	Prob.	MRI	Factor	Prob.	MRI	Prob.	MRI	Prob.	MRI	Prob.	MRI
dp	PP	Oo	0.0111	90	0.96	0.0107	94	0.04	0.0004	2250	0.0431	23	0.0427	23	0.0004	2250
dp	PP	Od	0.0143	70	0.90	0.0129	78	0.10	0.0014	700	0.0529	19	0.0514	19	0.0014	700
dp	PP	Mo	0.0111	90	0.80	0.0089	113	0.20	0.0022	450	0.0378	26	0.0356	28	0.0022	450
dp	PP	Md	0.0143	70	0.50	0.0071	140	0.50	0.0071	140	0.0357	28	0.0286	35	0.0071	140
dd	DF	Oo	0.0118	85	0.90	0.0106	94	0.10	0.0012	850	0.0435	23	0.0424	24	0.0012	850
dd	DF	Od	0.0133	75	0.50	0.0067	150	0.50	0.0067	150	0.0333	30	0.0267	38	0.0067	150
dd	DF	Mo	0.0118	85	0.80	0.0094	106	0.20	0.0024	425	0.0400	25	0.0376	27	0.0024	425
dd	DF	Md	0.0133	75	0.50	0.0067	150	0.50	0.0067	150	0.0333	30	0.0267	38	0.0067	150
dd	PP	Oo	0.0118	85	0.96	0.0113	89	0.04	0.0005	2125	0.0456	22	0.0452	22	0.0005	2125
dd	PP	Od	0.0133	75	0.50	0.0067	150	0.50	0.0067	150	0.0333	30	0.0267	38	0.0067	150
dd	PP	Mo	0.0118	85	0.80	0.0094	106	0.20	0.0024	425	0.0400	25	0.0376	27	0.0024	425
dd	PP	Md	0.0133	75	0.50	0.0067	150	0.50	0.0067	150	0.0333	30	0.0267	38	0.0067	150
dg	PP	Oo	0.0125	80	0.96	0.0120	83	0.04	0.0005	2000	0.0485	21	0.0480	21	0.0005	2000
dg	PP	Od	0.0167	60	0.50	0.0083	120	0.50	0.0083	120	0.0417	24	0.0333	30	0.0083	120
dg	PP	Mo	0.0125	80	0.80	0.0100	100	0.20	0.0025	400	0.0425	24	0.0400	25	0.0025	400
dg	PP	Md	0.0167	60	0.30	0.0050	200	0.70	0.0117	86	0.0317	32	0.0200	50	0.0117	86
dg	DF, GF	Oo	0.0125	80	0.90	0.0113	89	0.10	0.0013	800	0.0463	22	0.0450	22	0.0013	800
dg	DF, GF	Od	0.0167	60	0.30	0.0050	200	0.70	0.0117	86	0.0317	32	0.0200	50	0.0117	86
dg	DF, GF	Mo	0.0125	80	0.80	0.0100	100	0.20	0.0025	400	0.0425	24	0.0400	25	0.0025	400
dg	DF, GF	Md	0.0167	60	0.30	0.0050	200	0.70	0.0117	86	0.0317	32	0.0200	50	0.0117	86
cm	WL, LP	Oo	0.0083	120	0.85	0.0071	141	0.15	0.0013	800	0.0083	120	0.0071	141	0.0013	800
cm	WL, LP	Od	0.0100	100	0.50	0.0050	200	0.50	0.0050	200	0.0100	100	0.0050	200	0.0050	200
cm	WL, LP	Mo	0.0083	120	0.85	0.0071	141	0.15	0.0013	800	0.0083	120	0.0071	141	0.0013	800
cm	WL, LP	Md	0.0091	110	0.50	0.0045	220	0.50	0.0045	220	0.0091	110	0.0045	220	0.0045	220
cm	GF, ES	Oo	0.0111	90	0.60	0.0067	150	0.40	0.0044	225	0.0111	90	0.0067	150	0.0044	225
cm	GF, ES	Od	0.0111	90	0.40	0.0044	225	0.60	0.0067	150	0.0111	90	0.0044	225	0.0067	150
cm	GF, ES	Mo	0.0111	90	0.50	0.0056	180	0.50	0.0056	180	0.0111	90	0.0056	180	0.0056	180
cm	GF, ES	Md	0.0111	90	0.40	0.0044	225	0.60	0.0067	150	0.0111	90	0.0044	225	0.0067	150
cm	DF	Oo	0.0111	90	0.70	0.0078	129	0.30	0.0033	300	0.0111	90	0.0078	129	0.0033	300
cm	DF	Od	0.0111	90	0.50	0.0056	180	0.50	0.0056	180	0.0111	90	0.0056	180	0.0056	180
cm	DF	Mo	0.0111	90	0.70	0.0078	129	0.30	0.0033	300	0.0111	90	0.0078	129	0.0033	300
cm	DF	Md	0.0111	90	0.50	0.0056	180	0.50	0.0056	180	0.0111	90	0.0056	180	0.0056	180
cd	LP, WL	Oo	0.0067	150	0.40	0.0027	375	0.60	0.0040	250	0.0053	188	0.0013	750	0.0040	250
cd	LP, WL	Od	0.0067	150	0.20	0.0013	750	0.80	0.0053	188	0.0060	167	0.0007	1500	0.0053	188
cd	LP, WL	Mo	0.0067	150	0.40	0.0027	375	0.60	0.0040	250	0.0053	188	0.0013	750	0.0040	250
cd	LP, WL	Md	0.0067	150	0.20	0.0013	750	0.80	0.0053	188	0.0060	167	0.0007	1500	0.0053	188
cd	ES, AF	Oo	0.0067	150	0.40	0.0027	375	0.60	0.0040	250	0.0053	188	0.0013	750	0.0040	250
cd	ES, AF	Od	0.0067	150	0.20	0.0013	750	0.80	0.0053	188	0.0060	167	0.0007	1500	0.0053	188
cd	ES, AF	Mo	0.0067	150	0.40	0.0027	375	0.60	0.0040	250	0.0053	188	0.0013	750	0.0040	250
cd	ES, AF	Md	0.0067	150	0.20	0.0013	750	0.80	0.0053	188	0.0060	167	0.0007	1500	0.0053	188
cd	DF	Oo	0.0067	150	0.40	0.0027	375	0.60	0.0040	250	0.0053	188	0.0013	750	0.0040	250
cd	DF	Od	0.0067	150	0.40	0.0027	375	0.60	0.0040	250	0.0053	188	0.0013	750	0.0040	250
cd	DF	Mo	0.0067	150	0.40	0.0027	375	0.60	0.0040	250	0.0053	188	0.0013	750	0.0040	250
cd	DF	Md	0.0067	150	0.40	0.0027	375	0.60	0.0040	250	0.0053	188	0.0013	750	0.0040	250

[a] Forest environment. dp: warm, dry ponderosa pine forest potential; dd: warm, dry Douglas-fir forest potential; dg: warm, dry mixed grand fir and Douglas-fir forest potential; cm: cool, moist; cd: cold, dry forest potential.

[b] Cover type. PP: Ponderosa pine; DF: Douglas-fir; GF: Grand fir; WL: Western larch; LP: Lodgepole pine; ES: Engelmann spruce; SF: Subalpine fir.

[c] Forest structure. Oo: dominant and codominant trees at least 52.5 cm diameter breast height, one canopy layer, and canopy cover between 15% and 40%. Od: dominant and codominant trees at least 52 cm dbh, one canopy layer, and canopy cover over 40%. Mo: dominant and codominant trees at least 52 cm dbh, two or more canopy layers, and canopy cover between 15% and 40%. Od: dominant and codominant trees at least 52 cm dbh, two or more canopy layers, and canopy cover over 40%.

[d] All wildfire severities combined.

[e] Surface and mixed-severity wildfires combined.

[f] High-severity wildfires.

[g] Mean annual probability of occurrence.

[h] Mean return interval (years).

[i] Proportion of all wildfire in this severity class.

in years and an average probability for each insect species, using data from (Ager et al., 2004).

3.2.4. Stand growth transitions

Stand growth and succession transitions in our model are based on extensive review and adjustment of the models developed by Hann et al. (1997). Many model runs were done in conjunction with local silviculturists to adjust probabilities and correct transition linkages. Growth rates and successional trends reflect several major assumptions about forest behavior in the Blue Mountains:

1. Forest growth and successional rates depend on environment, being slower in dry and cold environments than in moist, productive sites.
2. Natural regeneration following disturbance is uncertain and may take several years to a decade or more, depending on the density of shrubs and competing vegetation and the average frequency of good seed crops and favorable climatic conditions. Grazing by large ungulates (deer, elk, and domestic livestock) substantially affects regeneration rates and tree density (Vavra et al., this volume). A high level of grazing reduces competing vegetation and creates favorable seedbeds, resulting in rapid regeneration or increased stand density (Riggs et al., 2000).
3. High-severity disturbances such as wildfire result in the preferential establishment of shade-intolerant conifers (e.g., ponderosa pine, western larch, and lodgepole pine). Disturbances that leave much of the canopy intact preferentially favor shade-tolerant species such as grand fir, Douglas-fir, Engelmann spruce, and subalpine fir.
4. High-severity disturbances in large-tree dominated stands result in high levels of snags and down wood that persist for 20 years or more unless salvage logging removes dead material. If salvage logging occurred, we also included artificial regeneration.
5. Forest growth and development transitions are generally time dependent and unidirectional in the absence of other disturbances. In the absence of disturbance, the forested landscape would become dominated by multiple-layered, large-tree forests of late seral species.

We tested our forest growth and succession assumptions with an independent modeling approach. We used stand-level simulations of tree growth from a stand growth model developed by Bettinger et al. (2004) and Graetz et al. (this volume) to check stand growth rates and transitions. Their model is essentially the Forest Vegetation Simulator (Crookston and Stage, 1999) stripped down for batch runs of large numbers of stands, producing outputs of thousands of stands that include lists of individual trees in search of optimal stand prescriptions. We classified tree lists from their model into our state classes, calculated average transition times owing to stand growth, and translated average transition times to annual transition probabilities. We found very few instances where calculated transitions from tree lists differed substantially from those estimated by field silviculturists. Where there were differences, we used calculated transitions and probabilities.

3.3. Disturbance and management scenarios

We modeled the long-term vegetation and disturbance conditions that might result from three scenarios: (1) background natural disturbances (no active management and no fire suppression), (2) fire suppression only, and (3) active fuels treatment. The scenarios represent existing and likely future combinations of management activities and natural disturbances, all beginning from current, existing vegetation conditions. The various land allocations and ownerships were either modeled individually (when disturbance probabilities varied by allocation or ownership) or were combined into a single land area. We assumed a constant management approach under the fire suppression only and active fuel treatment scenarios, no policy changes occurred to alter the probabilities of management activities on any land allocation or ownership. In addition, we assumed high levels of ungulate grazing for the fire suppression only and active fuel treatment scenarios, but low grazing effects in the background natural disturbance scenario (Vavra et al., this volume).

The background natural disturbance scenario did not include any management activities and was intended to represent the likely conditions under current climate with low ungulate grazing in the absence of present-day management activities. This scenario was modeled with the same natural disturbance probabilities across all ownerships and land allocations. The background natural disturbance scenario is generally similar to disturbance conditions assumed in various historical range of variability (HRV) analyses (Hann et al., 1997; Wimberly et al., 2000; Agee, 2003) but does not assume that model projections actually represent some past set of conditions. Rather, it produces simulations that represent potential conditions that might develop given current climatic conditions and natural disturbance probabilities as inferred from fire history and other disturbance studies. Because we included annual variability in disturbance probabilities (owing to local climatic fluctuations, fire ignitions, and insect outbreak cycles), our simulations also estimate variation in disturbance and vegetation conditions over time. We did not include global climate change trends. This means that the overall average probabilities of disturbance remain constant through time when averaged across our simulation periods (i.e., no long-term trends).

The fire suppression only scenario assumed no management activities other than fire suppression, high ungulate grazing, and low levels of salvage logging following stand-replacement disturbances on publicly owned lands regardless of land allocation. Active management probabilities from the active fuel treatment scenario were used for privately owned lands. Owing to a variety of environmental and social concerns, the probability of salvage following high-severity wildfire or insect outbreak is not particularly high (1%) compared to the entire area affected by those disturbances. Wildfire suppression and natural disturbance probabilities were included at current levels. Artificial regeneration (tree planting) was included following salvage logging.

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An active fuels treatment scenario was developed to approximate expected land management based on recently implemented fuel treatment policies. This scenario included levels and kinds of management activities designed to actively treat canopy and surface fuels to reduce wildfire risks in general forest lands within the first decade and to maintain relatively low levels of canopy and surface fuels across the landscape after the first decade. Because current fuel conditions are often high, initial fuel treatments were mostly accomplished through mechanical treatment rather than through prescribed fire. After initial mechanical treatment, prescribed fire was used to maintain fuels at relatively low levels, especially in warm-dry forest environments. Initial mechanical treatment rates were adjusted to allow treatment of stands in general forest allocations in warm-dry environments within the first decade, and then declined to maintenance levels. Rates of fuel treatment were lower in cool-moist and cold forest environments. Other management activities included precommercial thinning, commercial thinning, shelterwood harvest, and group selection harvest. The probabilities of all activities were adjusted to reflect different management objectives and activity levels by ownership and land allocation. For example, precommercial thinning is not used in lynx habitat areas, and silvicultural management is very limited in riparian areas. We distinguished riparian areas that are currently managed differently from uplands but did not distinguish natural vegetation or disturbances that might characterize riparian systems. Wildfire suppression probabilities continued at current levels. Natural disturbance probabilities other than fire remained at current levels.

3.4. Model projections and variability

Our VDDT models are relatively easy to run and execute quickly on a high-end desktop personal computer. The structure of the VDDT program allows runs of hundreds of years and many Monte Carlo simulations to generate averages and associated variability in state class abundance and disturbance occurrence (Beukema et al., 2003). We included annual variability for both wildfire and insect outbreaks by using a set of multipliers that we developed using expert opinion from local fire managers and forest pathologists and entomologists to reflect the frequency and severity of fire years and insect outbreaks over the period of record (generally 30 years from 1970 to 2000). The modeling process randomly assigns high, moderate, and low or normal fire and outbreak years for each Monte Carlo simulation. This means that our models include variation in fire occurrence and insect outbreaks over time. We assume this variation is caused by local climatic conditions, human activities, forest conditions, and cycles of insect activity. We used 30 Monte Carlo simulations for each model run to calculate average landscape conditions for each projected year and to assess variation. Because the process generates random sequences of high, moderate, and low or normal fire and outbreak years, the projections include sequences of years where wildfire, for example, is very high and nearly all the forests burn. Similarly, some year sequences include very little wildfire. Insect outbreaks tend to be more cyclical, depending on the insect species involved (Ager et al., 2004).

With the exception of the background natural disturbance scenario, we ran each VDDT projection for 200 years with 30 Monte Carlo simulations to estimate average conditions and variability. The simulation results we present are therefore yearly averages of the 30 Monte Carlo simulations. We ran the background natural disturbance scenario for 500 years with 30 Monte Carlo simulations to allow us to examine both the effects of a natural disturbance regime starting from existing conditions (short term, years 0–200) and long-term quasi-stable conditions (long term, years 201–500). We calculated several statistics for each disturbance regime simulation. The overall average for a landscape condition (e.g., area in a structure class) was calculated by combining all 30 Monte Carol runs and all simulation years. The average minimum and average maximum values were calculated by finding the minimum and maximum values, respectively, for the 30 Monte Carlo simulations of each year, then calculating the average of those yearly minima and maxima. The absolute minimum was the smallest value ever encountered in any year, and any Monte Carlo simulation and absolute maximum was the largest.

4. Results

4.1. Background natural disturbance scenario

Surface wildfire was the dominant disturbance across the landscape under the short-term background natural disturbance scenario (Fig. 3a). By the end of the 200-year simulation, insect outbreaks were affecting an average 0.2–0.5% of the potentially forested landscape annually. Wildfires of all severities affected an average of 1.5–4.7% annually. The area of surface wildfire was generally double or more the average annual amount of high-severity wildfires. The annual average proportion of the landscape affected by all disturbances varied from nearly none in some years to almost 10% in others.

Vegetation conditions under the background natural disturbance scenario changed substantially from current conditions over 200 years (Fig. 3d). At present, seedling/sapling and small-tree stands dominate the study area, occupying about 75% of the potentially forested landscape while large-tree forests comprise less than 10%. Small-tree single story, small-tree multi-story, medium-tree multi-story and large-tree multi-story stands all declined over 200 years. The decline in small-tree single story stands was particularly notable (37–17%). Grass/shrub, medium-tree single story, and large-tree single story structures increased. Increases in grass/shrub (6–23%) and large-tree single story stands (3–17%) were substantial. After about 200 years, average forest structure was relatively stable and dominated by grass/shrub (23%), seedling/sapling (24%), large-tree single story stands (17%), and small-tree single story stands (17%). Medium-tree stands and multi-layered forests of all sizes became relatively minor in comparison. Nearly all the large-tree forests were in warm-dry environments. Cool-moist and cold forest environments contained less than 10% large-tree forests over the long term.

Grass/shrub, seedling/sapling, large-tree single story, and small-tree single story forests dominated the landscape over

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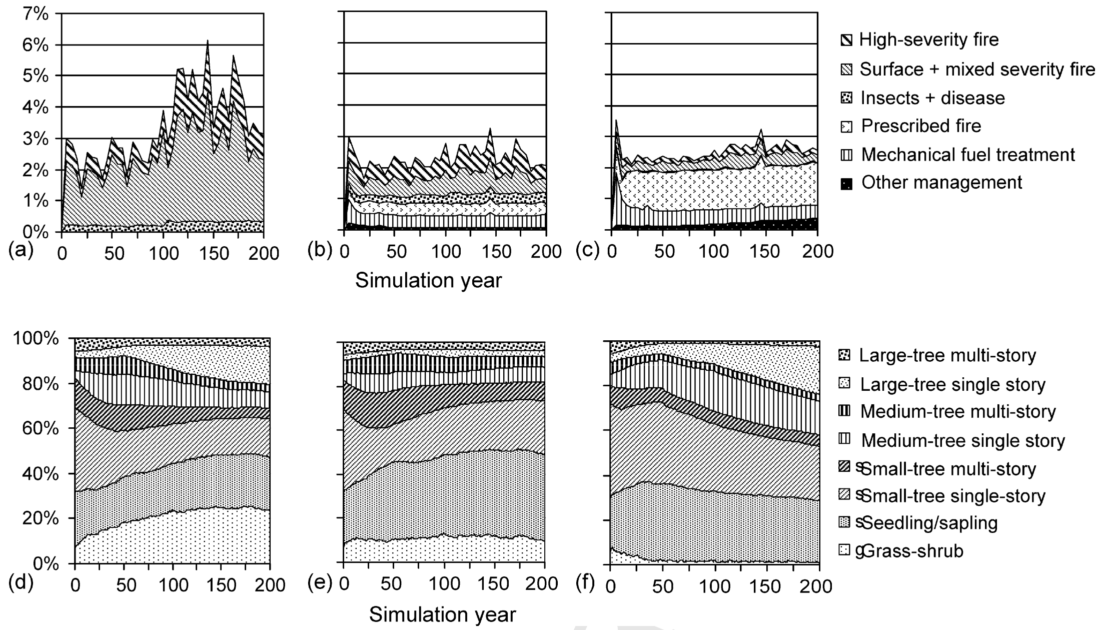

Fig. 3. Disturbances (a–c) and forest structure classes (d–f) for background natural disturbance, fire suppression only, and active fuel treatment simulation scenarios. Results are proportions of the study area from annual averages of 30 Monte Carlo simulations.

the long-term background natural disturbance scenario period (Fig. 4). There was considerable variation in these over our 30 Monte Carlo simulations. For example, some simulations contained as much as 25% large-tree single story conditions and some as little as 8%. Likewise, large-tree multi-story conditions ranged from an absolute maximum of 13% to an absolute minimum of 1%. In general, grass/shrub and large-tree single story classes are currently well below the simulated long-term conditions, whereas small-tree single story and small-tree multi-story forests are substantially above the simulated historical range. Large-tree multi-story forests, in contrast, are slightly above the overall long-term average.

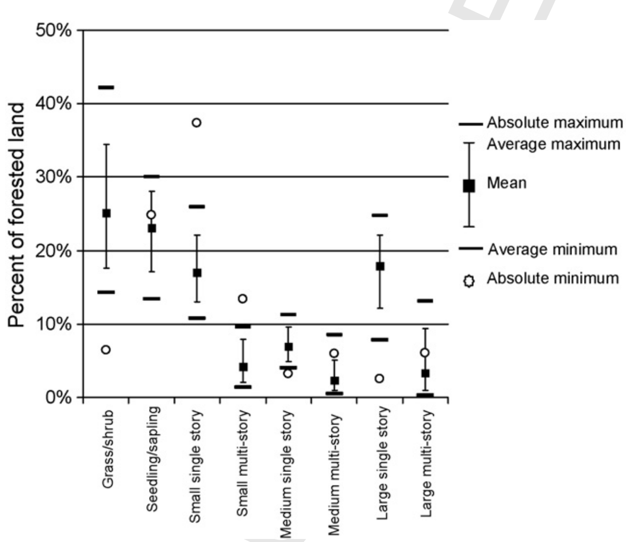

Fig. 4. Long-term mean and variation for area in different forest structure classes from years 201 to 500 of the background natural disturbance scenario. Mean, average minimum, average maximum, absolute minimum, and absolute maximum calculated from years 201 to 500 and 30 Monte Carlo simulations.

4.2. Fire suppression only disturbance scenario

The fire suppression only disturbance regime reflects relatively low overall levels of wildfire compared to the background natural disturbance regime, but a higher proportion of wildfires were of high severity (Fig. 3b). Wildfire burned between 1.3% and 1.4% of the landscape annually over decades 15–20. This was the only scenario in which high-severity wildfires burned as much area, on an annual average, as surface fires.

Insect outbreaks played an important role in landscape dynamics over 200 years and increased slightly at the end of the simulation period owing to increasing overall stand density. Cold forest environments had higher levels of insect activity than either warm-dry or cool-moist environments. In general, stand replacement by insects was generally similar to that under the background disturbance scenario. Mechanical fuel treatment, prescribed fire, and other management activity rates on privately owned lands reflected our modeling assumptions and were about 1% per year.

Dense multi-story forests, especially of smaller trees, were more abundant in the landscape under our fire suppression only disturbance scenario compared to the other scenarios. Fire suppression for 200 years produced a landscape with abundant seedling/sapling stands (Fig. 3e). Grass/shrub, seedling/sapling, and medium-tree single story structures all increased compared to current conditions. Seedling/sapling stands, in particular, increased from 25% to 36% of the potentially forested landscape area. Small-tree single story and small-tree multi-story forests decreased as small-tree forests, as a whole, dropped from 50% to 32% of the potentially forested landscape. Large-tree multi-story forests also declined (6–3%). Large-tree single story and medium-tree multi-story conditions remained relatively constant. The overall potentially forested landscape remained

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dominated by grass/shrub, seedling/sapling, and small-tree conditions, as it is at present.

4.3. Active fuel treatment disturbance scenario

Active fuel treatments produced notable changes in disturbances over 200 years compared to the other scenarios. Mechanical fuel treatment, prescribed fire, and other management activities affected 1.7–2.2% of the landscape area annually (Fig. 3c). Mechanical treatments were initially higher than prescribed fire (1.2% compared to 0.7% per year), but within 100 years, prescribed fire became dominant as initial mechanical treatment reduced fuel loads to levels that could be safely treated with prescribed fire. Most mechanical fuel treatments after 100 years occurred in cool-moist forests containing higher fuels and tree species more sensitive to underburning. The bulk of prescribed fire occurred in warm-dry forests where it is easiest to implement and highly effective. Other management activities, principally timber harvests on privately owned lands, occurred on about 0.1–0.2% of the landscape annually during the first 100 years. Shelterwood harvests in cool-moist and cold forests on national forest lands increased somewhat after 100 years as those forests grew to larger size classes.

Though quite variable on an annual basis (Fig. 3c), the forest area affected by wildfire declined from about 0.7% at year 0 to about 0.3% by 100 years, then slowly increased to about 0.7% again by year 200. This was substantially lower than the background average rate of 1.5–4.7% for wildfire disturbance under the background natural disturbance scenario and reflects combined effects from fire suppression and fuel reduction. It is also about half the annual occurrence of wildfire under the fire suppression only scenario. High-severity wildfire was heavily concentrated in cool-moist and cold forests, whereas surface wildfires dominated in warm-dry forests. Increasing tree size and stand density in upper elevation forests resulted in higher levels of stand-replacing wildfire toward the end of the simulation period. Insect outbreaks declined slightly over 200 years across the landscape but remained at relatively high levels in cold forests where fuel treatments and other stand-thinning activities were lowest. Insect outbreaks generated high-severity events on very little of the forest land per year in the active fuel treatment scenario and occurred at rates less than half those in the background natural disturbance and fire suppression only scenarios.

Grass/shrub, small-tree single story, small-tree multi-story, medium-tree multi-story, and large-tree multi-story structural conditions all declined over 200 years (Fig. 3f). The largest decrease occurred in small-tree single story stands, which declined from 37% to 24% of the potentially forested landscape area. Seedling/sapling, medium-tree single story, and large-tree single story stands all increased compared to current conditions. The increase in seedling/sapling stands was due to a combination of high-severity wildfire and insect outbreaks in cool-moist and cold forests while continued reburning in dense seedling/sapling stands limited growth into small-tree size classes. An increase in medium-tree single story stands was largely due to high levels of fuel treatments and prescribed burning in warm-dry forests. Large-tree stands, as a whole, increased from 8% to 24% over 200 years because small losses to large-tree multi-story stands from wildfire or insects were more than made up by large gains in single story stands owing to management (prescribed fire and thinning from below) and tree growth. Large-tree stands in warm-dry environments increased dramatically, but shifted strongly to single story conditions as a result of fuel treatment and prescribed fire activities, whereas those in cool-moist and cold forests nearly disappeared. The loss of large tree structures in upper elevation forests was due to a combination of insect outbreaks and stand-replacement wildfire, with a small additional loss as a result of relatively low levels of shelterwood harvest and fuel treatment.

5. Discussion

Our results beg two questions: Why are current conditions so different than those that might exist under a natural disturbance regime? and Can current conditions be maintained? We suggest that the path of forest disturbance, management treatments, and climate change over the last 100 years or more has produced current conditions that might be difficult to sustain. A long history of fire suppression, forest management, and high ungulate grazing (Vavra et al., this volume) has created forests of smaller trees, many of which might experience high-severity disturbance, especially as fire suppression and high ungulate grazing continue to increase stand densities. Management designed to maintain current conditions would have to carefully balance the generation and retention of large-tree stands (especially single story structures) while slowing high-severity disturbances from fire or insects. This might be especially difficult if abundant multi-layered large-tree forests are desired.

5.1. Lag time, variability, and key structural elements

The full influence of our alternative scenarios on forest structure took 150–200 years to develop. Decades to centuries were required for the growth of large trees and the establishment of a relatively stable long-term dynamic. In reality, climate change and other factors (e.g., changing political and management objectives) likely preclude forests from ever reaching a stable long-term dynamic at the spatial scale of our study area. The long timeframe required to generate relatively stable landscape conditions in our simulations resulted from the current low landscape abundance of large trees, the long time required to grow large trees, and the interaction of natural disturbances with stand development. Some tree species (e.g., ponderosa pine, western larch, and, to some extent, Douglas-fir) are long-lived and regenerate best in open, early seral conditions, whereas others (e.g., grand fir, subalpine fir) regenerate well in shaded environments and have shorter average longevity. Given the importance of large trees, the long timeframe required to grow them, and their potential longevity, and regeneration limitations for early seral tree species, we suggest that large ponderosa pine, western larch, and similar species are pivotal structural elements in this landscape.

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Our Monte Carlo simulations produced highly variable results for many landscape attributes (Fig. 4) because our models included annual variability in both wildfire and insect outbreak probabilities. Average annual conditions for various landscape characteristics reflect the most probable outcomes from the scenario. An average condition and variability are important and allow managers and others to evaluate likely trends. However, actual disturbances and, consequently, the amounts of habitats and structures over time follow a particular path that may or may not be the most likely path. For example, the amount of large-tree multi-story forest might average 3% of the landscape area under a natural disturbance scenario, but wildfire and insect outbreaks interacting with stand growth and development may produce wide fluctuations in the amount of large-tree multi-story forests over time in any one run.

5.2. Differences between current landscape and background natural disturbance conditions

The current landscape is outside the simulated long-term average minimum to average maximum ranges for several structural conditions compared to the background natural disturbance scenario (Fig. 4). Single story large-tree forests, for example, occupied an average of nearly 20% of the forested landscape under the background disturbance scenario, but less than 5% at present. Frequent, low-severity wildfire favored open stands of large, fire resistant trees under the background natural disturbance scenario while multi-storied large tree stands averaged less than 10% of the forested landscape. A combination of wildfire and insect outbreaks killed multi-story large-tree forests almost as fast as stands reached large, multi-storied condition in our simulations. This was particularly true in cool-moist and cold forests where infrequent wildfire allows high stand densities in early stand development. The currently existing structural conditions, driven by decades of fire suppression and various management activities, may be nowhere near a dynamic equilibrium. An expectation that the current landscape condition contains sustainable or stable amounts of various forest structures, and habitats may be unreasonable.

The decline of single story forests of large, fire-resistant trees and an increase in dense forests of smaller, fire-intolerant trees has been well documented in the interior Columbia basin (e.g., Everett et al., 1994; Hann et al., 1997; Hessburg et al., 1999; Hemstrom et al., 2001; Hessburg and Agee, 2003) and more generally in western North America (e.g., Covington and Moore, 1994; Peet, 2000). Several decades of fire suppression allowed fire-intolerant species such as grand fir to become established in the understory of previously open forests. In addition, timber harvest and insect activity reduced numbers of large ponderosa pine and other fire-tolerant species. Multi-story forests, on the other hand, have become more abundant in many places. Multi-story forests with large-trees have not increased, however, because large-trees in multi-story forests were lost to timber harvest and insect activity. Our background natural disturbance scenario results, indicating dominance by multi-story small- and medium-tree forests, agree well overall trends in the interior Columbia River basin.

5.3. Abundant multi-story large-tree forests may be difficult to sustain

The fire suppression only scenario did not produce large areas of multi-layered, dense large-tree forests, as might be expected when fires are suppressed for 200 years. Our models assumed that suppression of high-severity wildfires in dense forests is less effective than suppression of surface fires in open forests. Although we assume that fire suppression reduces the total amount of wildfire, high-severity wildfire was more common than in other scenarios. In addition, insect outbreaks disturbed more area, especially in cold forests, than in the other scenarios. Both trends were due to increases in dense, multi-layered forests on national forest lands as a consequence of fire suppression and no fuel management. In our simulations, insect outbreaks and wildfire converted many multi-layered large tree forests to grass/shrub and seedling/sapling stands about as quickly as trees reached large size. Large-tree forests, especially those with multi-layered structure, were less abundant at the end of 200 years than in any other disturbance scenario. Fire suppression alone might reduce the overall frequency of wildfire compared to historical conditions but is unlikely to generate large areas of multi-storied large-tree forest. In addition, wildfires would more often be of high severity and insect outbreaks would be conspicuous, leading to questions about the public acceptability of a fire suppression only scenario.

None of our scenarios produced abundant multi-story large-tree forests. In fact, those forests declined from current conditions under all three alternatives. The active fuel treatment scenario produced slightly more area in large-tree forests than the background disturbance scenario, and both produced considerably more area in large-tree forests than the fire suppression only scenario. The active fuel treatment scenario also generated more single story large- and medium-tree forests in warm-dry environments than the other scenarios. In all cases, large-tree dominated forests were less than 25% of the landscape area. Large trees take 150 years or more to grow in most areas of this landscape and, when lost, are difficult to replace. In addition, there is some question about the ability of stand thinning and fuel treatment to generate abundant stands of large, open ponderosa pine. Ager et al. (this volume) modeled stand-level effects of bark beetles and found that open stands of large ponderosa pine could become could suffer more mortality during a bark beetle outbreak. We were not able to fully account for this effect because their models did not include the suite of natural disturbances and management activities that occurred in our models. Perhaps the reduction in high-severity wildfire under the active fuel treatment scenario would offset increased insect mortality across the landscape. This possibility suggests the need for additional integration of stand-level disturbance models across large landscapes.

The relatively low levels of multi-story large-tree forests under all our scenarios indicate potential difficulties in management for wildlife species that are associated with multi-story large-tree forests. Wisdom et al. (2000) listed several species of conservation concern that are associated with multi-story older forests. Wales et al. (this volume) discuss the potential

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impacts of our scenarios on Lynx denning habitat. In effect, multi-story large-tree forests became more unstable as their abundance increased in our simulations. We suggest that at some level managing for high levels of multi-story large-tree forests may produce “boom and bust” conditions or other limitations on the sustainable amount of multi-story large-tree forests in this landscape.

5.4. Strengths and weaknesses of the state and transition models approach

State and transition models, in general, appear to us to have several strengths and weaknesses that are important to consider when contemplating their use for landscape simulation. We used STM because they allow integration of a wide variety of information from empirical data, other models, the literature, and expert opinion. The models are relatively easy to understand because ecological interactions are subsumed in states and transitions. However, portraying complex ecological interactions as boxes and arrows and using a combination of information from other models, the literature, and expert opinion means that very complicated interactions and different kinds of information are simplified and combined. Our projections, interpretations, discussion, and conclusions must be considered in this light.

Our simulation models are, as is true of all such efforts, a formalized set of assumptions about how we think the ecological processes (including human activities) in the study area interact to produce vegetation conditions, disturbances, and associated landscape characteristics. Although we used independent information from the literature and from stand-level silvicultural models to help build and calibrate our STM, our models still represent an integration of our assumptions. Our results, discussion, and conclusions are based on assumptions that may or may not represent actual ecological fact or “truth” and are, therefore, hypotheses about how this landscape might react to different management scenarios.

The composition and structure of vegetation through time is highly variable in this and landscapes of similar size, environment, and vegetation conditions. Our models were not-spatial; they did not simulate stand-level effects of management activities and disturbance. Results obtained from spatially explicit (e.g., patch-level) simulations provide important information about patch sizes, inter-patch distances, and other patch metrics that our models do not provide. Keane et al. (2002), using a spatially explicit (i.e., patch-level) landscape model found high levels of variability in community dynamics and patch metrics over time in comparable landscapes and suggested that simulation time periods should be at least 10 times the longest fire return interval to include rare but important events. They also suggest that landscapes should be large (e.g., >100,000 ha) to capture landscape patterns caused by large, rare fire events. Wimberly (2002), working in the Oregon Coast Range, found that even larger landscapes (e.g., >200,000 ha) were required to simulate the full range of historical wildfires. Our landscape was likely large enough to capture a representative range of forest and disturbance conditions in this environment, especially given the non-spatial nature of the models we used. We did not find very long simulation time periods to be necessary for the quasi-stable landscape condition to emerge under our scenarios, probably because our models were not spatially explicit and did not consider patch-level disturbance dynamics. We also found the non-spatial VDDT model much easier to calibrate than spatially explicit models. We expect that VDDT and similar models would be much easier to adapt for operational use by forest managers, particularly for large landscapes, than spatially explicit models. Perhaps a combination of approaches, using non-spatial models for general estimates of trends across large areas and spatially explicit models for local drill-down to patch characteristics would be work well for land managers.

6. Conclusions

State and transition models, in general, appear to us to have several strengths and weaknesses that pertain to interpreting our results. We used STMs because they allow integration of a wide variety of information from empirical data, other models, the literature, and expert opinion. The models are relatively easy to use and understand. This simplification, however, limited our ability to include detailed ecological relations and processes.

Several interesting landscape hypotheses emerge from our scenario simulations: (1) changes in management approach in landscapes the size of our study area may take decades or play out owing to the time required to grow large trees and the feedback loops among disturbances, (2) the current landscape is considerably different from that which might exist under a natural disturbance regime, (3) fire suppression alone does not mimic background natural disturbances and does not produce abundant large-tree structure, and (4) dense, multi-layered large-tree forests may be particularly difficult to maintain in abundance in this and similar landscapes owing to wildfire and insect disturbances.

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