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Forest Ecology and Management 85 (1996) 291–309

Forest Ecology and Management

Comparison of managed and pre-settlement landscape dynamics in forests of the Pacific Northwest, USA

David O. Wallin [a,b,*], Frederick J. Swanson [c], Barbara Marks [a], John H. Cissel [d], Jane Kertis [e]

[a] Department of Forest Science, Oregon State University, Corvallis, OR 97331, USA
[b] Current address: Center for Environmental Sciences, Huxley College of Environmental Studies, Western Washington University, Bellingham, WA 98225-9181, USA
[c] Forest Service-USDA, Pacific Northwest Research Station, 3200 Jefferson Way, Corvallis, OR 97331, USA
[d] Forest Service-USDA, Blue River Ranger District, Willamette National Forest, Blue River, OR 97413, USA
[e] Forest Service-USDA, Siuslaw National Forest, Corvallis, OR 97331, USA

Abstract

The landscape structure of forests subjected to wildfires fluctuates through time as a result of the episodic nature of these disturbances and long-term variation in the climatic conditions that influence the fire regime. These landscape dynamics influence a variety of important ecosystem processes. Before European settlement, landscape dynamics in Pacific Northwest forests were driven primarily by the patterns of wildfire, and an understanding of these dynamics can provide a unique frame of reference for evaluating current forest management policies and alternatives for the future. Tree-ring data and historical records of forest cutting were used to quantify the range of landscape conditions that existed on two large watersheds (4000 ha and 11 600 ha) between the late 1400s and 1990. A rule-based simulation model was used to generate landscape patterns that would result from five alternative future forest management scenarios. Our results demonstrate that conditions on these two watersheds in 1990 are outside the range of conditions that existed during most of the reconstructed pre-settlement era. Continued use of short (50 to 100 year) timber rotation lengths would push these watersheds even farther outside of this range. The use of much longer rotation lengths (200 + years) could bring these watersheds back to within or very near this range of pre-settlement conditions.

Keywords: Landscape dynamics; Managed forests; Pre-settlement; Rotation

1. Introduction

The coastal mountains of the Pacific Northwest of North America, extending from southeast Alaska to northern California, include some of the most productive forest lands in the world (Waring and Franklin, 1979). This region has provided a large share of the world timber supply for many decades (Waddell et al., 1989). Forest management policies for this region are currently undergoing major revisions. Throughout the United States portion of this region, and in many other parts of the western USA as well, forest cutting conducted since the 1940s on federally-managed land has used a dispersed ('staggered setting') system of 10–20 ha clearcut patches. Within the past decade, various concerns

* Corresponding author. Tel: (360) 650-7526 or -2844; fax: (360) 650-7284; e-mail: wallin@cc.wwu.edu.

0378-1127/96/$15.00 Published by Elsevier Science B.V.
PII S0378-1127(96)03765-6

have been raised over the effects of forest fragmentation that results from dispersed cutting (Harris, 1984; Rosenberg and Raphael, 1986; Franklin and Forman, 1987; Lehmkuhl et al., 1991; Saunders et al., 1991). These concerns led to a brief period of assessment of the consequences of changing from dispersed to more aggregated cutting patterns, under the assumption that cutting rates would remain unchanged (Franklin and Forman, 1987; Swanson and Franklin, 1992; Wallin et al., 1994). Two parallel developments were underway at this time: regional ecological assessments and planning efforts led to species conservation strategies which dramatically reduced the cutting rate and set landscape pattern by reserve designation (Thomas et al., 1990; Johnson et al., 1991; FEMAT, 1993; Noss, 1993); and the history of past landscape patterns and disturbance events was being used as a basis for designing managed landscape patterns (Swanson et al., 1993; Cissel et al., 1994). This historical approach has also been suggested for use in managing boreal forests (Hunter, 1993). The rationale for this approach is that the flora and fauna present at the time of European settlement are adapted to the range of conditions at that time so maintaining the system somewhere within this range would provide a high probability of producing conditions suitable for most, if not all, of these species. This includes unrecognized species and species with poorly understood habitat requirements. A key premise of this approach is that maintaining some level of natural habitat patterns and ecological processes is essential for maintaining ecosystem productivity. This approach has been referred to as 'ecosystem dynamics' or 'the range of natural (pre-settlement) variability'.

Given the profound alterations in the environment since European settlement, the relevance of using the range of pre-settlement variability as a model for developing management plans has been questioned. In many places, for example, future management options are severely constrained by the substantial reduction in the area of natural vegetation since European settlement because much of the land base has been usurped for buildings, agriculture or intensive, short-rotation industrial forestry. The pre-settlement disturbance regime included wildfires that, under extreme conditions, could consume over 1000 km^2 of forest in a single event (Agee, 1993). Disturbances of this size are not likely to be socially acceptable nor could the remaining forested land base absorb disturbances of this size without profound and unprecedented impact on ecosystem structure and function. Hence, it is not likely to be either practical or acceptable to implement management scenarios that mimic the full range of landscape conditions that existed under the pre-settlement disturbance regime. Nevertheless, given our rudimentary understanding of how to manage ecosystems and sustain native species, it seems prudent to use the range of pre-settlement conditions as a frame of reference for evaluating potential future management scenarios.

One of the primary obstacles to more widespread consideration of natural variability in the design of forest management plans is the difficulty of quantifying pre-settlement conditions. Detailed information will probably never be available, but, even a simplified reconstruction of the range of pre-settlement conditions could be quite useful. The work presented here represents a preliminary attempt to define the range of conditions that existed over the last 500 years on two study landscapes (4000 and 11 000 ha). We then discuss current landscape conditions relative to these historical conditions and examine landscape conditions created by five potential future management scenarios.

1.1. Pre-settlement disturbance regimes

Under the pre-settlement disturbance regime, landscape dynamics in these forests were driven primarily by the patterns of wildfire (Agee, 1991a; Agee, 1993). Other disturbance processes, such as landslides (Swanson et al., 1990), windthrow (Ruth and Yoder, 1953; Gratkowski, 1956), insects (Rudinsky, 1962; Powers, 1995) and volcanic eruptions (Yamaguchi, 1993) also play roles in some cases. Our analysis focuses on the effects of fire as the primary force driving landscape dynamics. During the pre-settlement era, forest fires were initiated by lightning, and accidental and intentional ignitions by Native Americans were also important (Burke, 1980; Boyd, 1986). Distinguishing between fires ignited by lightning and by Native Americans is problematic, and we have made no attempt to do so in our analysis.

1.2. Early European influences on the fire regime

Europeans initially began to influence the fire regime indirectly by reducing Native American populations and, in turn, reducing the frequency of ignitions by Native Americans. Initial contact between European traders and Native Americans began in the late 18th century. By the early 1830s European diseases had resulted in a 75 to 90% reduction of Native American populations in western Oregon (Scott, 1924; Cook, 1955). The large-scale influx of European settlers into the region began in the 1840s and by the 1850s substantial European settlements existed within 20 to 50 km of our study areas (Burke, 1980). At about this time, Europeans became an increasingly important source of fire ignitions. Concerted efforts to suppress fire began in about 1910 (Burke, 1980; Pyne, 1982).

1.3. Recent land-use effects on landscape dynamics

Landscape dynamics in these forests are currently driven by forest cutting practices. Forest cutting on private lands throughout the region have been underway for well over a century (Robbins, 1988), however, substantial harvesting on public forests did not begin until the late 1940s and the 1950s (Harris, 1984). Over the past few decades, most planning for public forests has been based on rotation lengths of 80 to 100 years and 10–20 ha cutting units (USDA Forest Service, 1990; Spies et al., 1994). On private forest lands, much larger clearcuts are used and rotation lengths of about 50 years are common (Spies et al., 1994). The first rotation of dispersed cutting is now 20 to 40% complete on much of the public lands outside the designated wilderness and other protected areas (Ripple et al., 1991; Spies et al., 1994). New timber sales on public lands have been curtailed in recent years (USDA Forest Service and USDI Bureau of Land Management 1994) as new approaches to forest management are being considered (FEMAT, 1993).

1.4. Approaches to reconstructing the fire regime

The characteristics of a fire regime, or any other disturbance process, can be described in terms of the spatial pattern, frequency and severity of events (Pickett and White, 1985). Each of these parameters can be estimated by using a combination of air-photo records (for recent fires), and tree origin and fire scar dates archived in tree rings. Techniques for using tree origin dates and fire scar dates to reconstruct the fire regime have been well described (Arno and Sneck, 1977; Barrett and Arno, 1988; Morrison and Swanson, 1990; Arno et al., 1993). Fire scars can provide more accurate dating of events since tree regeneration can occur over a period of several decades after a fire event (Means, 1982; Huff, 1995). The spatial extent of a fire is determined by matching sites with fire scar or tree regeneration evidence for the same date. For any given fire, severity can be inferred from the relative abundance of fire scars and tree regeneration; a high proportion of scarred trees within a fire polygon suggests an event of low to moderate severity and a high proportion of tree regeneration suggests a high severity (stand-replacement) event.

1.5. Differences between logging and fire

The stand-level effects of wildfire and clearcutting are profoundly different (Hansen et al., 1991). Historically, clearcutting in these forests involved the felling of all overstory trees and removal of 25 to 75% of the woody material from the site (Harmon et al., 1996). In contrast, following a wildfire the surviving trees and standing dead trees contribute a great deal to the structural character of the post-fire stand. Many old-growth stands have experienced multiple fires of low to moderate severity within the life-span of the oldest trees in the stand. These structural differences between stands that regenerate after a wildfire or a clearcut will not be considered in our analysis. Instead, we will focus on differences in the landscape patterns generated by wildfire and forest cutting. These landscape-level differences have not been previously quantified for these forests. A complete evaluation of the ecological differences between pre- and post-settlement forests will require consideration of both stand- and landscape-level characteristics, and this is beyond the scope of our paper.

2. Methods

2.1. Study areas

The work presented here is based on fire history data from two study areas in the central Oregon Cascade Range on the Willamette National Forest (Fig. 1). Teensma (1987) collected fire history data for an 11 607 ha study area centered on the Lookout Creek watershed, including the H.J. Andrews Experimental Forest. Connelly and Kertis (1992) collected data for a 7600 ha study area centered on the Augusta Creek watershed, 30 km to the south of Lookout Creek. The current analysis will use data from all

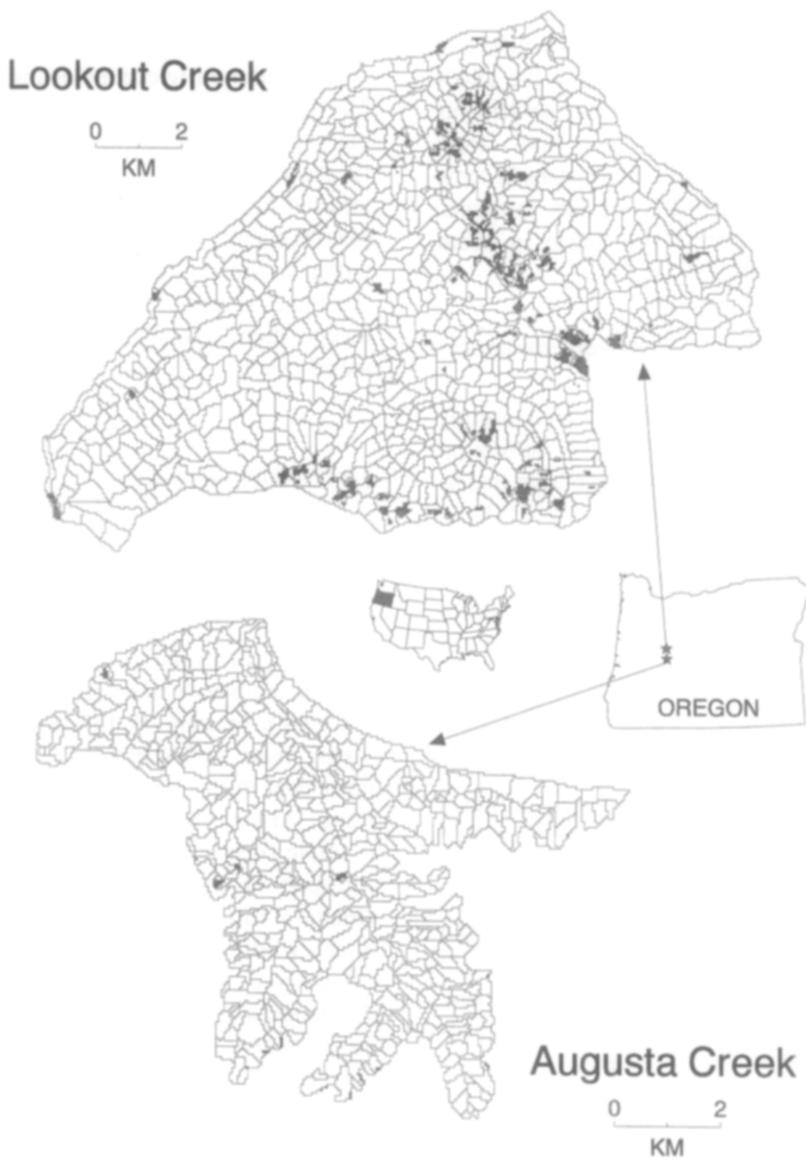


Fig. 1. Lookout Creek and Augusta Creek watersheds (Blue River Ranger District of the Willamette National Forest). Areas in black are permanent openings in the forest canopy (rock outcrops and meadows). Polygons within each watershed represent 'potential harvest units' (see Methods: Model Description).

of the Lookout Creek study area and from a 4013 ha subset of the Augusta Creek study area. Those parts of the larger Augusta Creek study area that are within a roadless area and a Congressionally designated wilderness area were excluded due to limited sampling in these areas.

Topography in both study areas is steep and highly dissected, with elevations ranging from 370 to 1600 m at the Lookout Creek study area and 660 to 1755 m at Augusta Creek. Most of these study areas lie within the Western Hemlock (*Tsuga heterophylla*) Zone (Franklin and Dyrness, 1973). Both study areas include a series of relatively permanent openings in the forest canopy (Fig. 1), including rock outcrops and meadows. These features were mapped based on interpretation of recent aerial photographs (G. Lienkaemper, pers. comm.) and they cover 3.0 and 0.3% of the Lookout and Augusta Creek study areas, respectively.

2.2. Fire history reconstruction

The methods used to reconstruct the fire history at each site are described in Teensma (1987) and Connelly and Kertis (1992). At both sites, fire history reconstruction was based primarily on examination of tree rings exposed on stumps in recent clearcuts. All cutting was in natural — i.e. previously unharvested — stands. In each clearcut, six to ten large stumps were examined for evidence of fire scars and pitch rings. Origin dates for each tree and the dates for any scars or pitch rings were determined by ring counts conducted in the field. Harvest records were later used to correct these counts for the number of years since the tree was cut and the origin date was also corrected for stump height (described in Teensma, 1987; Morrison and Swanson, 1990). In selected locations where no clearcut was present, increment cores were used to establish tree origin dates. No attempt was made to collect fire scar data from cored trees. Increment cores were lightly sanded before rings were counted under a dissecting microscope. These ring counts were also corrected for height at coring to establish a tree origin date. Tree origin date was sometimes estimated when cores did not reach the center of large trees or when the cores missed the tree center. Most of the trees selected for examination, either as stumps or using increment cores, were Douglas-firs (*Pseudotsuga menziesii*), a shade-intolerant species that typically regenerates after high-severity fires. Selection of sample sites was aided by referring to recent aerial photographs. Care was taken to sample along stand age-class discontinuities that were apparent in these photographs.

Ring counts conducted in the field are subject to some error resulting from difficulties in resolving sections of very closely spaced rings, deterioration of the stumps and physical damage during harvesting. More accurate counts can be obtained by sawing sections off stumps and returning them to the lab for cleanup and more careful analysis. This approach takes more time and reduces both the spatial extent of the sampling effort and the number of samples that can be obtained. Both Teensma (1987) and Connelly and Kertis (1992) opted to count rings in the field.

These counts were usually accurate to within ± 10 years and the fire return interval in this system is generally much greater than this. Nevertheless, because the counts do have some error associated with them, fires cannot be accurately dated to a single year. Multiple scars a few years apart on a single tree sometimes provided unambiguous evidence of multiple, closely spaced, fires. Interpretation of tree origin dates is also difficult because seedling establishment may lag a fire event by many years (Means, 1982; Huff, 1995). Since assigning fires to a single year is problematic, and sometimes, inappropriate, Teensma (1987) and Connelly and Kertis (1992) identified fire episodes that span several years. These fire episodes were longer for earlier episodes because older trees had more extensive development of compressed growth rings and therefore more opportunity for counting error.

Establishing the occurrence of a fire episode relied on several pieces of evidence. A single scar was not sufficient because physical damage resulting from the fall of an adjacent tree or other processes can create scars very similar to those created by fire. Establishing a fire episode required evidence from five or more trees, at different sites, and the evidence had to include both fire scars and tree origin dates. Sites used to establish an episode had both spatial and temporal coherence, and the boundary for the fire episode was mapped by using topographic information and knowledge of fire behavior. Age-class discontinuities apparent on aerial photographs were used to help define the boundaries of more recent fire episodes.

The natural fire rotation length (Heinselman, 1973) was calculated for selected time periods at each site. The natural fire rotation is the length of time it would take to burn an area equivalent to the area under study. It is calculated as the duration of the time period divided by the total proportion of the study area burned during this period.

2.3. Reconstructing pre-settlement landscape dynamics

Each of the fire-episode maps is interpreted as representing a patch of moderate to high severity fire (Teensma, 1987; Connelly and Kertis, 1992). Mapping the variability in fire severity within each fire polygon is problematic. The fact that scars on surviving trees are available to date these fires demonstrates that they were not complete stand-replacement events. For this reason, a detailed analysis of the forest age-class distribution created by these fires is inappropriate for these data. Nevertheless, each fire episode depicted by these maps can be treated as an event that opened up the canopy to some degree. For this analysis, open-canopy conditions are defined to persist for 40 years after a fire. In low to moderate elevation forests on the west side of the Cascade Range in Oregon, 40-year-old stands typically have conifer crowns covering at least 60% of the area with tree heights of 15 m or more (Curtis et al., 1982; Schoonmaker and McKee, 1988). The set of fire episode maps for each site was used to develop a simplified, multitemporal, representation of landscape patterns. Starting with the earliest interpreted fire episode at each study area, the sequence of fire episode maps was overlaid in a GIS and time since the previous fire was tracked for each 50×50 m grid cell. Initially, unburned grid cells were treated as background; however, by the mid-1500s nearly all grid cells in both study areas had burned at least once. The fire year is defined as the mid-point of the fire episode. At 20-year time steps, and for each fire year, a map of open- and closed-canopy forest was produced. All permanent openings (Fig. 1) were mapped into the open-canopy forest category.

A simplified representation of landscape patterns at any point in time can be obtained by describing the distribution, shape and abundance of patches of open- and closed-canopy forest. For each map depicting the distribution of open- and closed-canopy forest, landscape patterns were quantified using FRAGSTATS (McGarigal and Marks, 1995). This program quantifies landscape patterns by using a variety of indices. From among the dozens of indices available to describe landscape condition (O'Neill et al., 1988; McGarigal and Marks, 1995), we selected three simple, ecologically meaningful indices: (1) density of edges (meters per hectare) between open- and closed-canopy forest; (2) the percentage of the landscape in closed-canopy forest; and (3) the percentage of closed-canopy forest area more than 100 m from the edge of an open-canopy patch (a measure of the amount of interior, closed-canopy forest). Chen et al. (1992) quantified variation in several physical and biological factors along transects from recent clearcuts into old-growth Douglas-fir forest stands in Oregon and Washington. Measurable edge effects persist for hundreds of meters from clearcut edges for some response variables. Chen et al. arbitrarily defined the edge influence as extending to the point where a response variable returns to two-thirds of the value for the interior forest environment. Using this definition, they found depth-of-edge influence ranging from 0 to 137 m for various response variables. For this study, we used 100 m as the boundary for edge influence.

2.4. Reconstructing post-settlement landscape dynamics

Forest cutting began in the early 1950s at Lookout Creek and in the 1960s at Augusta Creek. All cutting units were mapped, and the year of harvest for each unit was obtained from USDA Forest Service records. Harvest records were compiled through 1990. The landscape patterns generated by forest cutting were reconstructed using these cutting-unit maps in the same way as the fire-episode maps. The cutting maps were sequentially overlaid in a GIS and time-since-cutting was tracked for each 50×50 m grid cell. At 10-year time steps, a map of closed- and open-canopy forest (including permanent openings; Fig. 1) was generated and landscape patterns were evaluated using FRAGSTATS. When combined with the maps generated by using the fire episodes, this produced a continuous record of landscape pattern dynamics for each study area from the late 1400s to 1990.

2.5. Modeling alternative future landscape dynamics

2.5.1. Model description

The effects of alternative future forest cutting scenarios on landscape patterns were examined by using the CASCADE model (Wallin et al., 1994). The CASCADE model simulates landscape pattern dynamics in response to forest cutting and subsequent regrowth. Forest regrowth is indexed as time since disturbance. The model operates in a simple, gridded landscape. The user specifies a cutting rate (rotation length), minimum stand age eligible for harvest (equal to the rotation length in this application), adjacency constraints on cutting and either an aggregated or dispersed distribution of cuts. The model requires a map that defines 'Potential Harvest Units' (PHUs) for the study area (Fig. 1). The shape, size and position of the PHUs were defined by a forest engineer, based on regulatory, engineering and other logistical constraints on log removal, road placement and environmental protection. Topography and the position of the perennial stream network are the primary basis for defining these constraints. As cutting proceeds, individual PHUs are selected and cut completely; cutting only a portion of a unit is not permitted.

The spatial distribution of cuts is quantified by using a dispersion index developed by Clark and Evans (1954):

$$R = 2p^{1/2}r,$$

where R is the dispersion index, p is the mean patch density (number of patches per unit area), and r is the mean nearest neighbor distance (see also Pielou, 1977 p.155; Sinclair, 1985). A value of $R < 1$ indicates an aggregated distribution of cuts and $R > 1$ indicates a dispersed or uniform distribution of cuts. Nearest neighbor distances are calculated using the centroids of individual PHUs.

To select a PHU for cutting, all those eligible for cutting are examined in turn and the dispersion index is calculated using a single candidate PHU and all those containing open-canopy forest. After examining each candidate, the one that either maximizes (dispersed cutting) or minimizes (aggregated cutting)

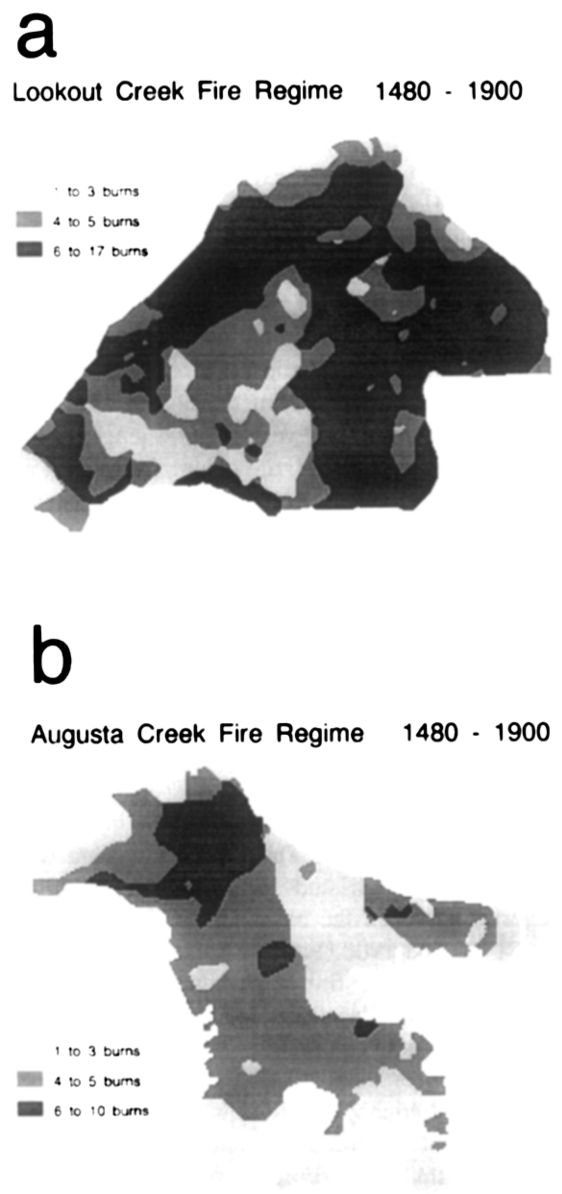


Fig. 2. Fire frequency at (a) Lookout Creek and (b) Augusta Creek study areas. These maps were produced by overlaying all fire episodes between 1480 and 1900 at each study area. Light gray: < 4 fires; medium gray: 4 to 5 fires; dark gray: > 5 fires. Percentage of the study area in the low, medium, and high frequency fire categories for Lookout Creek and Augusta Creek are 17, 28, 55 and 39, 47, 14, respectively.

the dispersion index is selected. This process is repeated until the cutting quota for the time step is reached. At the end of each 10-year time step, a map of open- and closed-canopy forest is output by the model and landscape conditions are quantified by using FRAGSTATS. For both study areas, 400-year model runs were generated for each scenario. Additional details on the model are provided in Wallin et al. (1994).

2.5.2. Scenario description

Five management scenarios were modeled to examine the effect of variation in rotation length and the spatial distribution of cuts. The first scenario is intended to provide a simplified representation of current management on industrial forest lands in the region. These lands are intensively managed for wood production and 50-year rotation lengths (20% of the area cut per decade) are common (Spies et al., 1994). The second scenario, with a 100-year rotation length (10% of the area cut per decade), is similar to management practices used on public forest lands in the PNW during the 1980s (USDA Forest Service, 1990). The remaining three scenarios incorporate aspects of more recently proposed forest management plans (Johnson et al., 1991; FEMAT, 1993). One of these scenarios used a 200-year and another used a 333-year rotation (5% and 3% of the area cut per decade, respectively). The final scenario used of 100-, 200- and 333-year rotation lengths in different parts of the study area. The fire history data were used to stratify each study area into three different categories based on the observed fire frequency between 1480 and 1900 (Fig. 2). Areas with low (1–3 fires), medium (4–5 fires) and high (> 5 fires) fire frequencies were placed on 333, 200 and 100 year rotations, respectively. These rotation lengths are somewhat longer than the mean fire-return interval for each fire-frequency class; however, most of these fires were not stand-replacement events, and these rotation lengths more accurately reflect the dominant age-class of forests present on these sites.

The 100-year rotation scenario used dispersed cutting, and adjacent PHUs were not eligible for harvest until the next 10-year time step. The other four scenarios used aggregated cutting with no adjacency constraint. All simulations were initiated using 1990 conditions.

3. Results

Teensma (1987) identified 31 individual fire episodes between 1482 and 1935 for the Lookout Creek study area, and Connelly and Kertis (1992) identified 17 fire episodes for the Augusta Creek study area between 1469 and 1920 (Fig. 3). The Lookout Creek study area is nearly three times larger than the Augusta Creek study area, which may explain this difference in the number of observed fires. In all but one case, fire episodes in each study area did not overlap in time. At Augusta Creek the 1785 and 1787 fire episodes did overlap in time, but these two fires were spatially discontinuous.

Fire episode size and frequency varied considerably throughout the period of record at both study areas (Fig. 3). At Lookout Creek, very extensive fire episodes in 1482 and 1532 covered 96 and 61%, respectively, of the study area. Each of the subsequent recorded fire episodes burned between 0.4 and 31% of this study area. Fire episodes were less frequent during the period between 1600 and 1750 (five episodes in 150 years) than during the period after 1750 (17 in 150 years). At Augusta Creek, very extensive fire episodes in 1499, 1523 and 1539

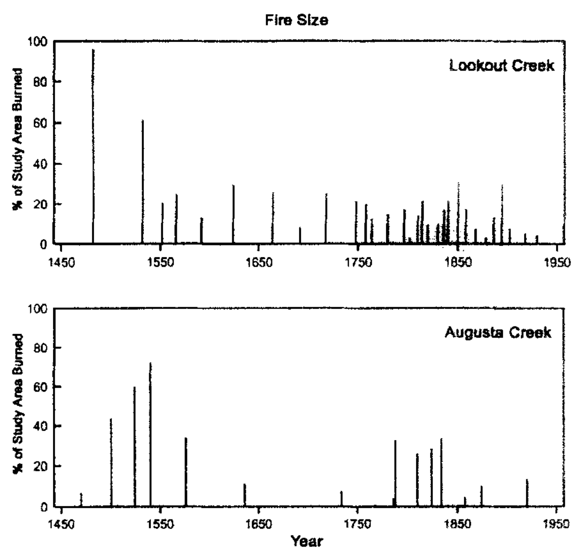


Fig. 3. Fire size in the (a) Lookout Creek (11607 ha) and (b) Augusta Creek (4013 ha) study areas. Fire size is expressed as a percentage of the size of the total study area. The mean fire size was 19% (2262 ha) and 23% (915 ha) of the study area at Lookout and Augusta Creeks, respectively.

covered 44, 60 and 72%, respectively, of this study area. The cumulative effect of these three fire episodes was to burn 93% of the Augusta Creek study area. Fire episodes between 1592 and 1785 were much less extensive; each of these fire episodes burned between 0.4 and 11% of the Augusta Creek study area. From 1787 through 1834, fires were more extensive and more frequent. Fire frequencies were lower and fire sizes were smaller after 1834.

The combined effect of the observed variation in fire episode size and frequency was to generate substantial variability in the disturbance rates (percentage of the study area disturbed per decade). Major disturbances were usually followed by a period of many decades with no disturbance (Fig. 3). During the pre-settlement era (1480–1850), no disturbances were recorded during 18 and 25 of the 37 decades at Lookout and Augusta Creeks, respectively. The overall, pre-settlement era decadal disturbance rates were 9.2% at Augusta Creek and 12.6% at Lookout Creek (Table 1). These figures correspond to a natural fire rotation (Heinselman, 1973) of 108 and 79 years, respectively. At both sites, the disturbance rate was much lower during the fire-suppression era (1911–1950). The disturbance rate was also low during the post-settlement era (1851–1910) at Augusta Creek, but at Lookout Creek post- and pre-settlement disturbance rates were comparable.

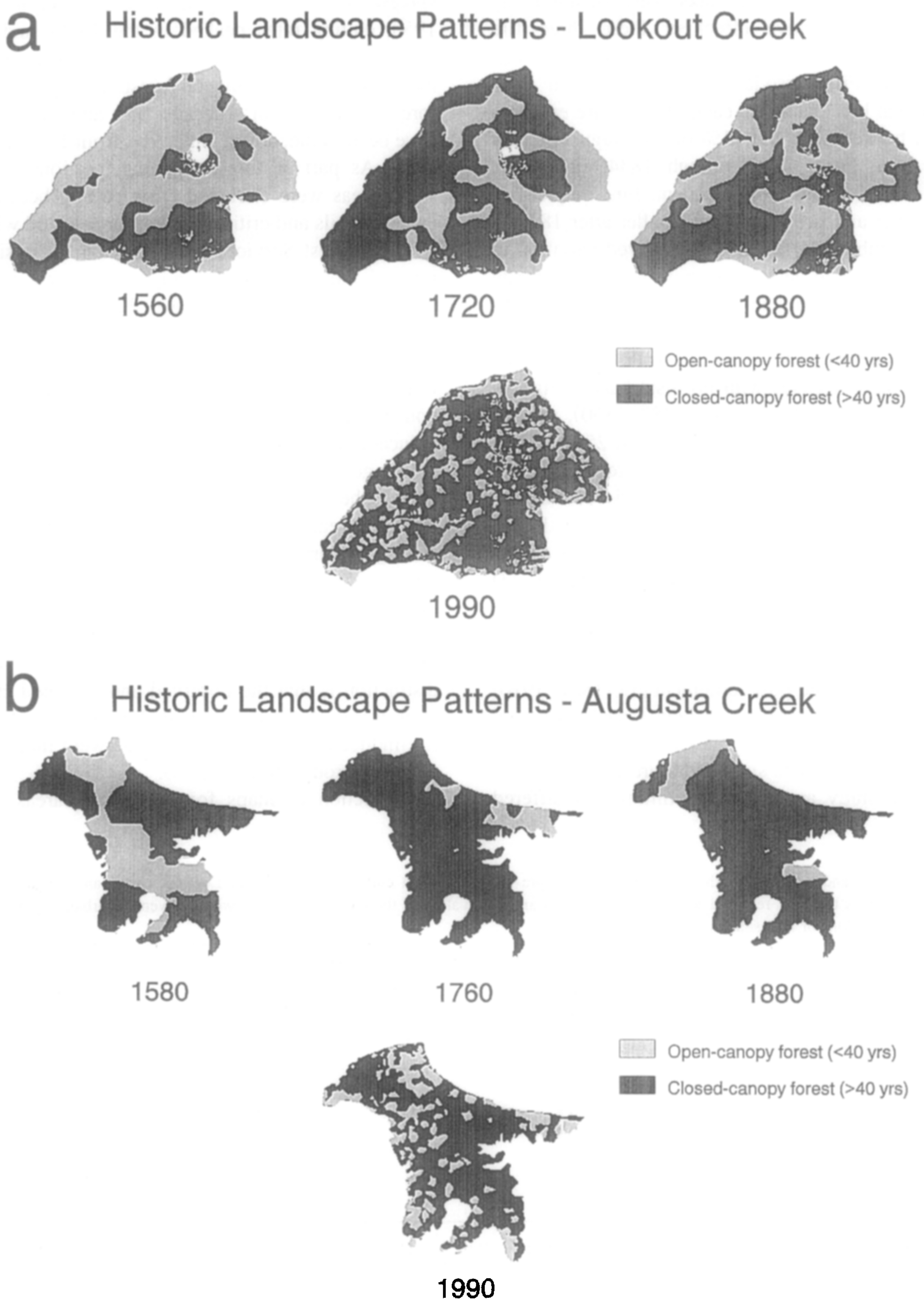
Forest cutting rates since 1950 are comparable in these two study areas, but the timing has differed somewhat. Forest cutting at Lookout Creek began in 1950, cutting rates peaked in the 1960s and have been very low since then because most of it is managed as an experimental forest. In Augusta Creek, forest cutting began a decade later and cutting rates have been stable, at just under 9%, from 1970 through 1990. As part of forest planning, portions of both study areas were excluded from cutting because of unstable soils and critical habitat for selected wildlife (USDA Forest Service, 1990). For our analysis, we ignored these exclusions and, because our calculations used a larger land base, the cutting rates reported in Table 1 are somewhat lower than those called for in the National Forest management plan. Using our calculations, the forest cutting rates at both areas were lower than the pre-settlement, fire-induced, disturbance rates; however, they were within the range of fire-induced disturbance rates observed during the pre- and post-settlement eras (Fig. 3). Similarly, each of the five alternative future forest cutting scenarios use harvest rates that are within the range of pre-settlement disturbance rates. At times, the modeled harvest rates drop below the targets because of shortages in stands that are at or above the minimum allowable ages for harvest. A key distinction is that the observed and modeled cutting rates were relatively constant through time, but the fire-induced disturbance rates underwent major fluctuations through time (Fig. 3).

Inspection of maps for both study areas at se-

Table 1
Disturbance rate and rotation length resulting from fire (pre-1950) or forest cutting (post-1950) at Lookout Creek and Augusta Creek. Rotation length is the time required to burn or harvest an area equivalent to the entire study area with a given disturbance rate (i.e. the 'natural fire rotation' of Heinselman (1973))

Time period	% of study area disturbed per decade		Rotation length (years)	
	Lookout	Augusta	Lookout	Augusta
1480–1850 (Pre-settlement)	12.6	9.2	79	108
1851–1910 (Post-settlement)	13.0	2.4	77	418
1911–1950 (Fire suppression)	2.4	3.4	410	298
Timber harvest				
1951–1960	4.8	0.0	209	-
1961–1970	9.9	7.2	101	139
1971–1980	6.0	8.9	165	112
1981–1990	6.7	8.8	149	114
1951–1990 (Overall timber harvest rate)	6.9	6.2	146	161

Note: Some of the forest in both study areas is not considered suitable for cutting because of unstable soils or status as critical habitat for wildlife. These areas were included in our calculations so the actual rotation lengths on lands available for cutting are somewhat higher than those shown here. See text for details.


a
Historic Landscape Patterns - Lookout Creek
1560
1720
1880
Open-canopy forest (<40 yrs)
Closed-canopy forest (>40 yrs)
1990
b
Historic Landscape Patterns - Augusta Creek
1580
1760
1880
Open-canopy forest (<40 yrs)
Closed-canopy forest (>40 yrs)
1990

lected dates suggests that landscape patterns generated by the pre-settlement fire regime were quite different from conditions in 1990 (Fig. 4). In both watersheds, the patterns in 1990 are the result of forest cutting and fire suppression, and the earlier patterns were generated exclusively by fire. In general, the forest patterns appeared much more fragmented in 1990 than for the earlier dates. The five management scenarios produced a wide range of landscape patterns after 100 years (Fig. 5). The time series of historical and modeled future landscape patterns were evaluated in terms of three landscape indices calculated using FRAGSTATS (McGarigal and Marks, 1995).

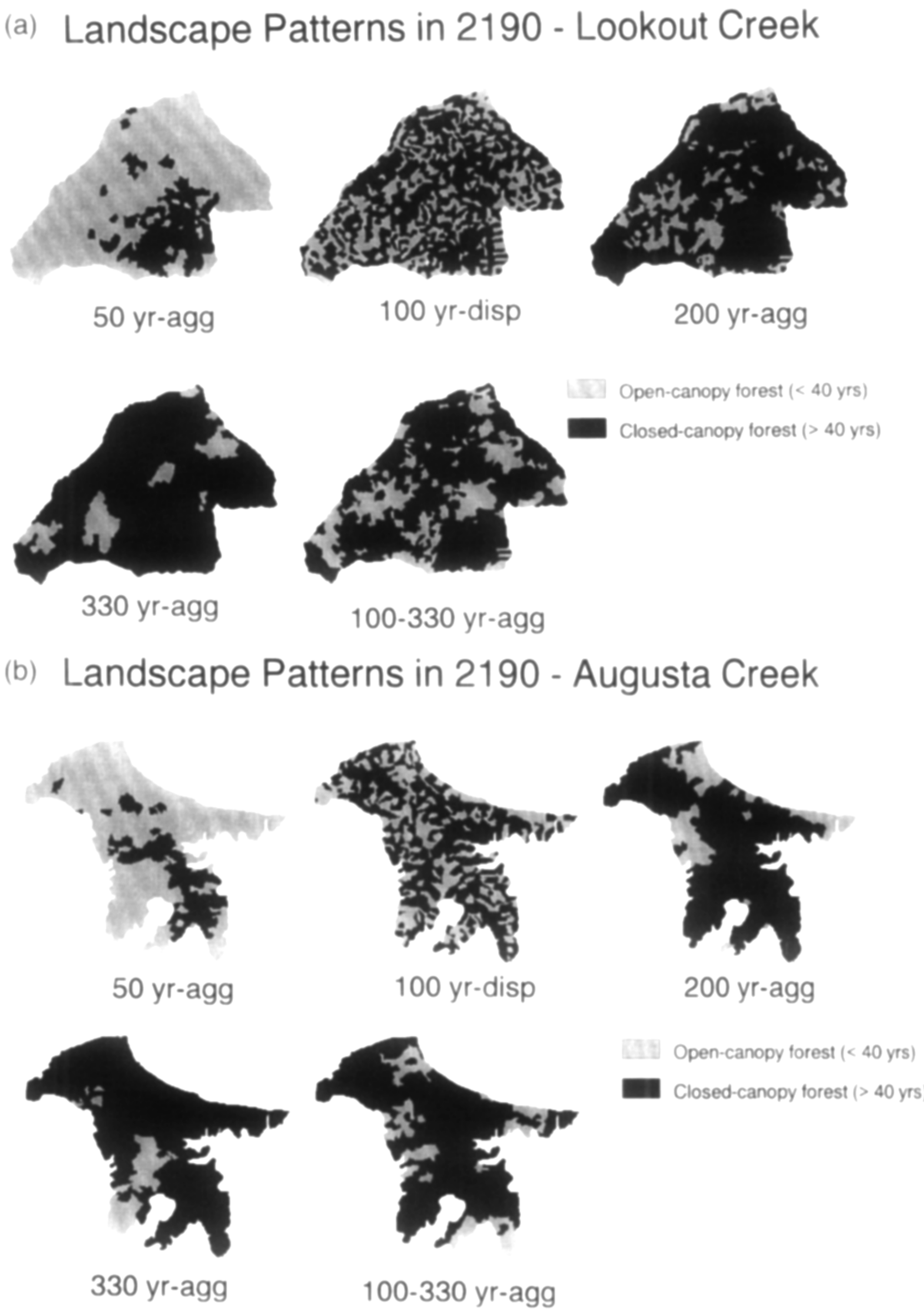
Edge density at Augusta Creek (Fig. 6b) underwent substantial fluctuations during the pre-settlement era, in response to variation in the size and frequency of fires. The series of large fires between 1499 and 1539 (Fig. 3b) reduced the amount of closed-canopy forest in the study area and resulted in very low edge densities. During most of the 1600s and early 1700s, edge density remained low for the opposite reason; the infrequent, small fires during this period produced very little open-canopy forest. During the late 1500s and after the mid-1700s, fires of intermediate size generated somewhat higher edge densities. The edge density at Lookout Creek (Fig. 6a) was also low initially, as a result of the very large 1482 fire episode (Fig. 3a). After recovering from this event, edge density remained fairly stable during the remainder of the pre-settlement era. This stability, relative to Augusta Creek, may in part result from the larger size of the Lookout Creek study area.

Edge density in both study areas increased dramatically with the onset of forest cutting (Fig. 6). The edge densities observed for each study area in 1990 are well outside the range of conditions that have existed in these watersheds over the previous 500 years. Conditions in 1990 are the product of a dispersed pattern of forest cutting with relatively short (~ 100 year) rotation lengths. Results of our simulations demonstrated that dispersed and aggregated cutting patterns produced very different edge densities. The continuation of dispersed cutting on a 100-year rotation after 1990 increased edge density even further and maintained these very high levels indefinitely. The four scenarios that used aggregated cutting patterns resulted in large reductions in edge density. At Lookout Creek, the 50- and 300-year rotations appear to maintain conditions comparable with pre-settlement conditions. The 200-year and mixed-length rotations maintained edge densities that are somewhat higher than those of the pre-settlement era. At Augusta Creek, the 200- and 330-year rotations maintained conditions comparable to those of the late 1500s and early 1800s. The 50-year and mixed-length rotations maintained conditions somewhat higher than pre-settlement edge densities.

At both sites, edge densities generated by the use of a 50-year rotation length were roughly comparable with those generated by the 200- 330-year and mixed-length scenarios. For the 50-year rotation length, edge densities were relatively low because of the scarcity of closed-canopy forest. In the other three scenarios, edge densities were low for the opposite reason — a scarcity of open-canopy forest. When cutting was dispersed, edge densities were high because cuts were in small patches with high edge-to-area ratios.

During most of the pre-settlement era over 60% of both study areas was covered by closed-canopy forest (Fig. 7). The abundance of closed-canopy forest was reduced during the late 1400s and much of the 1500s as a result of the very large fires during this period. Similarly, the frequent intermediate-sized fires of the early 1800s at Augusta Creek and the late 1800s at Lookout Creek also caused a brief reduction in the amount of closed-canopy forest. In both areas, conditions in 1990 appeared to be well within the range of pre-settlement conditions. Four of the five future management scenarios appeared to maintain

Fig. 4. Historical landscape patterns in (a) Lookout Creek and (b) Augusta Creek for three arbitrarily selected dates in the past and for the year 1990. Light areas are open-canopy stands; dark areas are closed-canopy stands. White areas in the 1560 and 1720 maps for Lookout Creek are areas that had not burned since 1482. All disturbance in the 1990 coverages results from forest cutting; disturbance for the other dates results from fire.


(a) Landscape Patterns in 2190 - Lookout Creek
50 yr-agg
100 yr-disp
200 yr-agg
330 yr-agg
100-330 yr-agg
Open-canopy forest (< 40 yrs)
Closed-canopy forest (> 40 yrs)
(b) Landscape Patterns in 2190 - Augusta Creek
50 yr-agg
100 yr-disp
200 yr-agg
330 yr-agg
100-330 yr-agg
Open-canopy forest (< 40 yrs)
Closed-canopy forest (> 40 yrs)

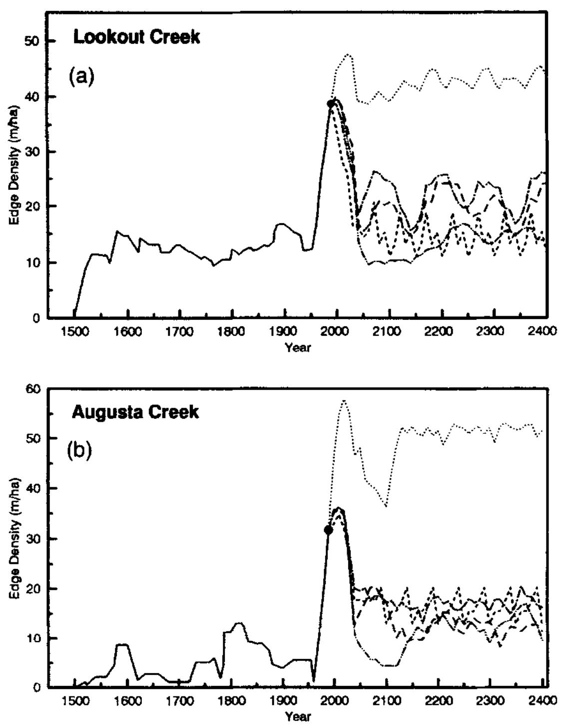


Fig. 6. Edge density (m ha^{-1}) in (a) Lookout Creek and (b) Augusta Creek: (—) historical conditions; (●) conditions in 1990. Landscape dynamics after 1990 were generated by using the CASCADE model with rotation lengths of 50 years (---); 100 years (· · ·); 200 years (– – –); 330 years (-· · -); and a mixture of 100, 200, and 330 years (– · –). The simulations based on the 100-year rotation used dispersed cutting, and all others used aggregated cutting.

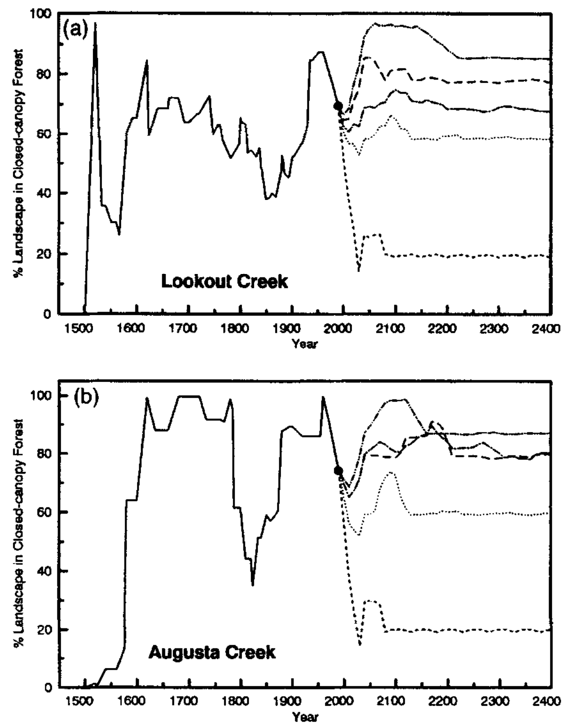


Fig. 7. Percentage of the landscape in closed-canopy forest in (a) Lookout Creek and (b) Augusta Creek: (—) historical conditions; (●) conditions in 1990. Landscape dynamics after 1990 were generated by using the CASCADE model with rotation lengths of 50 years (---); 100 years (· · ·); 200 years (– – –); 330 years (-· · -); and a mixture of 100, 200, and 330 years (– · –). The simulations based on the 100-year rotation used dispersed cutting and all others used aggregated cutting.

conditions that are within the range of conditions that persisted during most of the pre-settlement era. Conditions created by aggregated cutting on a 50-year rotation were not unprecedented, but were only observed during a very brief period in the early 1500s. Differences among these five scenarios were directly related to the rotation lengths — the 50-year rotation length left the smallest amount of closed-canopy forest and the 330-year rotation length left the most. The mixed-length scenario produced somewhat different results in each study area because of differences in the proportions of each study area that were placed on short, medium and long rotations (Fig. 2).

The initial increase in the abundance of closed-canopy forest at both sites under the 330-year rotation occurred as the supply of 330-year-old stands was exhausted and cutting was curtailed until stands of sufficient age became available. Differences among study areas in the duration of this increase resulted from the 1990 stand age-class distributions. Augusta Creek had more area in old stands in 1990 than did Lookout Creek.

After the recovery from the large fires of the late 1400s and early 1500s, the percentage of closed-canopy forest area that was more than 100 m from an edge remained between 70 and 90% at both study

Fig. 5. Landscape patterns in the year 2190 for (a) Lookout Creek and (b) Augusta Creek generated by the CASCADE model, using five alternative management scenarios. See text for description of the scenarios. Light areas are open-canopy stands; dark areas are closed-canopy stands.

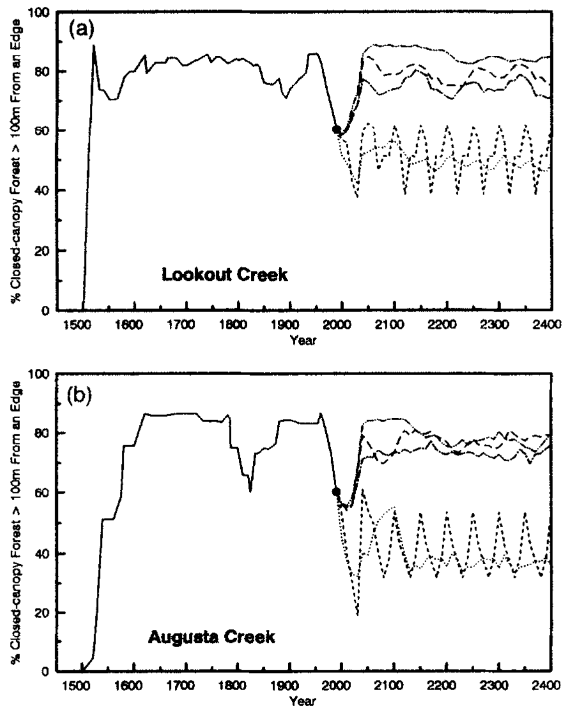


Fig. 8. Percentage of closed-canopy forest area more than 100 m from an edge in (a) Lookout Creek and (b) Augusta Creek: (—) historical conditions; (●) conditions in 1990. Landscape dynamics after 1990 were generated by using the CASCADE model with rotation lengths of 50 years (- - -); 100 years (· · ·); 200 years (— — —); 330 years (- · · -); and a mixture of 100, 200, and 330 years (– · –). The simulations based on the 100-year rotation used dispersed cutting, and all others used aggregated cutting.

areas during most of the pre-settlement era (Fig. 8). Conditions in 1990 at both study areas were somewhat below this level. Management, using either a 50- or 100-year rotation length, would result in further reductions and would maintain both sites at levels that are well below those that were typical during the pre-settlement era. The other three management scenarios maintain conditions comparable with those that occurred during most of the pre-settlement era. The differences among these three scenarios appeared to be modest.

4. Discussion

A variety of studies have examined the effect of fire suppression and climatically driven variation in wildfire frequency on stand-structure and composition in temperate forests (Cwynar, 1987; Agee et al., 1990; Clark, 1990; Overpeck et al., 1990), but these studies have not examined landscape-level patterns. The effect of wildfire and fire suppression on landscape pattern have been quantified in Yellowstone National Park, Wyoming (Romme, 1982) and the Boundary Waters Canoe Area, Minnesota (Heinselman, 1973; Baker, 1989; Baker, 1992). Because both of these areas are protected, the researchers were not able to examine the effects of forest cutting on landscape patterns. Several studies have used satellite data to quantify recent changes in landscape pattern in response to forest cutting (Hall et al., 1991; Ripple et al., 1991; Spies et al., 1994; Cohen et al., 1995; Turner et al., 1996; Zheng et al., 1996), but these studies lack a pre-settlement frame of reference for interpreting these changes. Our results present the first analysis of long-term records of pre-settlement, wildfire-generated landscape patterns relative to those generated by recent historical and alternative future forest cutting.

The size and frequency of wildfires at both study areas varied substantially between the late 1400s and 1950 (Fig. 3), and this variability is consistent with results from previous studies (Heinselman, 1973; Romme, 1982; Baker, 1989, 1992; Agee et al., 1990; Clark, 1990; Overpeck et al., 1990). The large fires of the late 1400s and early 1500s are consistent with regional climate reconstructions indicating that warmer and dryer conditions persisted during this time (Miller, 1969; Burbank, 1981; Heikkinen, 1984). Our data are biased, however, because recent, high-severity, fires obliterate the record of older fires at many sites and this reduces the number of sample points for the earlier time periods. This lower sampling density makes it difficult to detect smaller fires and fires that are detected must be mapped based on fewer sample points. A decline in fire size with time has been reported for other fire history studies (Agee et al., 1990; Agee, 1991b), and it is not clear whether this is an artifact of sampling bias or a real effect of climatic change. Reductions in the wildfire disturbance rate during the 1600s and 1700s are consistent with the cooler and wetter conditions that persisted during this period (Miller, 1969; Burbank, 1981; Heikkinen, 1984; Graumlich and Brubaker, 1986). Fire severity may have also varied substantially among centuries, although available data do not permit us to address this point. The decade-to-decade variability in the disturbance rate has declined somewhat since 1900 with the onset of fire suppression and forest cutting. Each of the five future management scenarios strive to maintain a constant cutting rate, and these model runs assume completely effective fire suppression. Unplanned and uncontrollable wildfires would add some unknown degree of variability to these modeled disturbance rates. The model runs also use a rigid constraint on the minimum stand age eligible for cutting. Although stands of different ages do vary in their susceptibility to wildfire (Agee and Huff, 1987), stands of any age can burn, given the proper conditions. When combined with the constant cutting rate, this rigid constraint on the minimum cutting age will lead to a uniform stand age-class distribution that remains constant through time. All but the longest cutting rotations modeled here would also reduce or eliminate mature and old-growth forest structure from these landscapes. The use of some level of green-tree retention in harvested stands could provide for some old-growth characteristics in these landscapes (Swanson and Franklin, 1992).

A reduction in the abundance of older stands and the reduction in the spatial and temporal variability in the age-class distribution may have important ecological consequences. High variability in the stand age-class distribution is likely to generate spatial and temporal fluctuations in the abundance of habitat for species that depend on a particular seral stage. These fluctuations are likely to play an important role in maintaining biological diversity (Southwood, 1977; Seagle, 1986; Urban and Smith, 1989; Hansen et al., 1991; McLaughlin and Roughgarden, 1993). One additional consequence of a constant harvest rate is that it requires maintaining a road network in perpetuity. Road networks can promote the spread of exotic species (Mack, 1986; Forcella and Harvey, 1988; Schmidt, 1989; Tyser and Worley, 1992), and they can also profoundly alter surface and groundwater hydrology (Megahan, 1987; Jones and Grant, 1996). Our results, combined with these earlier studies, suggest that episodic harvest events followed by road removal might be more appropriate for these systems.

In some systems subjected to periodic disturbances, the patch mosaic structure of small areas may fluctuate widely through time while on a larger scale, the proportion of the area in a particular state remains constant. This pattern has been referred to as the 'shifting-mosaic steady state' (Bormann and Likens, 1979). A modeling study has suggested that a stable mosaic can be expected on areas that are about 50 or more times larger than the mean size of the disturbance (Shugart and West, 1981). The area in Minnesota studied by Heinselman (1973) and Baker (1989, 1992) was 87 times larger than the mean fire size and showed no evidence of a stable mosaic. The area in Wyoming studied by Romme (1982) and both of our study areas were considerably smaller — nine and five times larger, respectively, than the mean fire size — and these areas also showed no evidence of a stable landscape mosaic. Baker (1989) concluded that the concept of the shifting-mosaic steady state is not likely to be applicable in fire-prone, temperate forests. He pointed out that environmental heterogeneity in these systems leads to variability in the susceptibility to burning, which generates spatial heterogeneity in the age-class distribution. Within small areas of relatively uniform fire susceptibility, fire sizes are simply too large to permit a stable mosaic. Baker argued that this combination of environmental heterogeneity and the mismatch between environmental grain size and disturbance grain size dictates that these systems are more likely to consist of a 'mosaic of different non-steady state mosaics.' The area in Minnesota studied by Baker has low topographic relief and a much more homogeneous environment compared to the highly dissected topography and heterogeneous environment of our study areas in Oregon. This difference suggests that the forest in our study areas are even more likely than Baker's study area to undergo large temporal variation in landscape conditions, regardless of the size of the area under study.

Variability in the wildfire disturbance rate generated a wide range of landscape conditions in our study areas between the late 1400s and 1850. Romme (1982) and Baker (1992) also reported large variation in pre-settlement landscape conditions. In their study areas, the effect of fire suppression was modest because of the short duration of effective fire suppression compared with the natural fire rotation length. In our study areas, conditions in 1990 were well outside the range of pre-settlement conditions for edge density (Fig. 6) and near the extremes of this range for the abundance of interior closed-canopy forest (Fig. 8). Landscape conditions in 1990 are the product of both fire suppression and 30 to 40 years of dispersed forest cutting. We believe that the patterns of forest cutting have played a much more important role than has fire suppression in shaping these landscape patterns. Previous modeling work has demonstrated that landscape patterns created by dispersed cutting are quite different from those created by aggregated cutting (Franklin and Forman, 1987; Li et al., 1993; Wallin et al., 1994). Landscape patterns created by dispersed cutting have also been shown to be extremely resistant to change, and erasing these patterns requires a substantial reduction in the harvest rate (Wallin et al., 1994). The results of our current study are consistent with these findings. The three management scenarios that used aggregated cutting and a reduced harvest rate (longer rotations) did the best job of changing landscape conditions to within or very near the pre-settlement range. The two management scenarios that used the shortest rotation lengths maintained landscape conditions well outside the range of conditions that persisted for most of the pre-settlement era. Although we lack sufficient knowledge to fully analyze the ecological consequences of these landscape conditions, it seems prudent to conclude that the sustained use of either of these two plans would profoundly alter ecosystem structure and function. On federally-managed lands, stand and landscape conditions resulting from these two management plans would also violate various federal statutes (USDA Forest Service and USDI Bureau of Land Management, 1994).

In these analyses, we have treated clearcutting and wildfire as equivalent disturbances. In these ecosystems, however, their effects at the stand-level are profoundly different. After a wildfire, surviving trees, snags, and coarse woody debris from the previous stand contribute a great deal to the structural complexity of young, natural stands (Franklin et al., 1981; Spies et al., 1988; Hansen et al., 1991). After a traditional clearcut, little or no standing structure is retained from the previous stand, and this lack of structural legacy is one of the key differences between intensively managed and natural stands (Hansen et al., 1991). Recent changes in forest cutting practices are aimed at increasing the amount of structure retained after forest cutting (Swanson and Franklin, 1992). If these changes are fully implemented, this would reduce some of the differences between natural and managed stands.

This study demonstrates an approach to quantifying the range of pre-settlement variability for a variety of landscape-level attributes. By ignoring important differences in within-stand structural complexity, our results are likely to underestimate the ecological differences between the pre-settlement landscapes and those generated by forest cutting. A more complete analysis of the ecological consequences of the landscape patterns presented here will require more explicit consideration of structural characteristics at the stand level. Nevertheless, our results do provide unique insights on some of the differences between managed and pre-settlement landscapes. This information can be invaluable in the development and evaluation of alternative forest management plans.

Acknowledgements

Funding for this work was provided by the USDA Forest Service New Perspectives Program and the National Sciences Foundation through the H.J. Andrews Long-Term Ecological Research (LTER) Program (NSF Grant DEB 90-11663). J.K. Agee, M. Brookes, M. Eng, S. Garman, J. Jones and A. McKee provided useful comments on earlier drafts of this manuscript. We are grateful to G. Lienkaemper and M. Murillo for their invaluable technical assistance. P. Teensma, and K. Connelly graciously provided us with the fire history data used in this analysis.

References

Agee, J.K., 1993. Fire Ecology of Pacific Northwest Forests. Island Press, Washington, DC.

Agee, J.K., 1991a. Fire history of Douglas-Fir forests in the Pacific Northwest. In: L.F. Ruggiero, K.B. Aubry, A.B. A.B. Carey and M.H. Huff (Editors), Wildlife habitat relationships in old-growth Douglas-Fir forests. USDA Forest Service, Pacific Northwest Research Station, General Technical Report PNW-GTR 285, Portland, OR, pp. 25–34.

Agee, J.K., 1991b. Fire history along an elevational gradient in the Siskiyou Mountains, Oregon. Northwest Sci., 65: 188–199.

Agee, J.K., Finney, M. and De Gouvenain, R., 1990. Forest fire history of Desolation Peak, Washington. Can. J. Forest Res., 20: 350–356.

Agee, J.K. and Huff, M.H., 1987. Fuel succession in a western hemlock/Douglas-fir forest. Can. J. Forest Res., 17: 697–704.

Arno, S.F. and Sneck, K.M., 1977. A method for determining fire history in coniferous forests of the mountain west. USDA Forest Service, General Technical Report INT-42, Intermountain Forest and Range Experiment Station, Ogden, UT.

Arno, S.F., Reinhardt, E.D. and Scott, J.H., 1993. Forest structure and landscape patterns in the subalpine lodgepole pine type: a procedure for quantifying past and present conditions. USDA Forest Service, General Technical Report INT-294, Intermountain Research Station, Ogden, UT.

Baker, W.L., 1989. Landscape ecology and nature reserve design in the Boundary Waters Canoe Area, Minnesota. Ecology, 70: 23–35.

Baker, W.L., 1992. Effects of settlement and fire suppression on landscape structure. Ecology, 73: 1879–1887.

Barrett, S.W. and Arno, S.F., 1988. Increment-borer methods for determining fire history in coniferous forest. USDA Forest Service, General Technical Report INT-244, Intermountain Research Station, Ogden, UT.

Bormann, F.H. and Likens, G.E., 1979. Catastrophic disturbance and the steady state in northern hardwood forests. Am. Sci., 67: 660–669.

Boyd, R., 1986. Strategies of Indian burning in the Willamette Valley. Can. J. Anthropol., 5: 65–86.

Burbank, D.W., 1981. A chronology of late Holocene glacier fluctuations at Mount Rainier, Washington. Arctic Alpine Res., 13: 369–386.

Burke, C.J., 1980. Historic fires in the central western Cascades, Oregon. M.S. Thesis, Oregon State University, Corvallis, OR.

Chen, J., Franklin, J.F. and Spies, T.A., 1992. Vegetation response to edge environments in old-growth Douglas-fir forests. Ecol. Applic., 2: 387–396.

Cissel, J.H., Swanson, F.J., McKee, W.A. and Burditt, A.L., 1994. Using the past to plan the future in the Pacific Northwest. J. Forest., 92 (8): 30–31, 46.

Clark, J.S., 1990. Fire and climate change during the last 750 years, northwestern Minnesota. Ecol. Monogr., 60: 135–159.

Clark, P.J. and Evans, F.C., 1954. Distance to nearest neighbor as a measure of spatial relationships in populations. Ecology, 35: 445–453.

Cohen, W.B., Spies, T.A. and Fiorella, M., 1995. Estimating the age and structure of forests in a multiownership landscape of western Oregon, USA. Internat. J. Remote Sensing, 16: 721–746.

Connelly, K.P. and Kertis, J.A., 1992. Augusta Creek fire history. unpublished report to the Blue River Ranger District of the Willamette National Forest.

Cook, S.F., 1955. The epidemic of 1830-33 in California and Oregon. University of California Publications in Am. Archaeol. Ethnol., 43 (3): 303–326.

Curtis, R.O., Clendenen, G.W., Reukema, D.L. and Demars, D.J., 1982. Yield tables for managed stands of coast Douglas-Fir. USDA Forest Service, Pacific Northwest Forest and Range Experiment Station, General Technical Report PNW-GTR-135, Portland, OR.

Cwynar, L.C., 1987. Fire and the forest history of the North Cascade Range. Ecology, 68: 791–802.

FEMAT, 1993. Forest Ecosystem Management: an Ecological Economic, and Social Assessment, Report of Forest Ecosystem Management Assessment Team, July 1993, Portland, OR.

Forcella, F. and Harvey, S.J., 1988. Patterns of weed migration in northwestern U.S.A. Weed Sci., 36: 194–201.

Franklin, J.F., Cromack, K., Denison, W., McKee, A., Maser, C., Sedell, J., Swanson, F.J. and Juday, G., 1981. Ecological characteristics of old-growth Douglas-fir forests. USDA Forest Service General Technical Report PNW-GTR-118, Portland, OR.

Franklin, J.F. and Dyrness, C.T., 1973. Natural vegetation of Oregon and Washington. USDA Forest Service General Technical Report PNW-GTR-8, Portland, OR.

Franklin, J.F. and Forman, R.T.T., 1987. Creating landscape patterns by forest cutting: ecological consequences and principles. Landscape Ecol., 1: 5–18.

Gratkowski, H.J., 1956. Windthrow around staggered settings in old-growth Douglas-fir. Forest Sci., 2: 60–74.

Graumlich, L.J. and Brubaker, L.B., 1986. Reconstruction of annual temperature (1590–1979) for Longmire, Washington, derived from tree rings. Quatern. Res., 25: 223–234.

Hall, F.G., Botkin, D.B., Strebel, D.E., Woods, K.D. and Goetz, S.J., 1991. Large-scale patterns of forest succession as determined by remote sensing. Ecology, 72: 628–640.

Hansen, A.J., Spies, T.A., Swanson, F.J. and Ohmann, J.L., 1991. Conserving biodiversity in managed forests. Bioscience, 41: 382–392.

Harmon, M.E., Garman, S.L. and Ferrell, W.K., 1996. Modeling the historical patterns of tree utilization in the Pacific Northwest: implications for carbon sequestration. Ecological Applications, in press.

Harris, L.D., 1984. The Fragmented Forest. University of Chicago Press, Chicago, IL.

Heikkinen, O., 1984. Dendrochronological evidence of variations of Coleman Glacier, Mount Baker, Washington, USA. Arctic Alpine Res., 16: 53–64.

Heinselman, M.L., 1973. Fire in the virgin forest of the Boundary Waters Canoe Area, Minnesota. Quatern. Res., 3: 329–382.

Huff, M., 1995. Forest age structure and development following wildfires in the western Olympic Mts., Washington. Ecol. Applic., 5 (2): 471–483.

Hunter, M.L., 1993. Natural fire regimes as spatial models for managing boreal forests. Biol. Conserv., 65: 115–120.

Johnson, K.N., Franklin, J.F., Thomas, J.W. and Gordon, J., 1991. Alternatives for management of late-successional forests of the Pacific Northwest: A report to the agricultural committee and the merchant marine and fisheries committee of the U.S. House of Representatives. (unpublished report, 59 pp.) October 8, 1991.

Jones, J.A. and Grant, G.E., 1996. Peak flow response to clearcutting and roads in small and large basins, western Oregon. Water Resour. Res., in press.

Lehmkuhl, J.F., Ruggiero, L.F. and Hall, P.A., 1991. Landscape-scale patterns of forest fragmentation and wildlife richness and abundance in the southern Washington Cascade range. In: L.F. Ruggiero, K.B. Aubry, A.B. Carey and M.H. Huff (Tech. Coord.), Wildlife and vegetation of unmanaged Douglas-fir forests. USDA Forest Service, Pacific Northwest Research Station, Portland, Oregon, General Technical Report PNW-GTR-285, pp. 425–442.

Li, H., Franklin, J.F., Swanson, F.J. and Spies, T.A., 1993. Developing alternative forest cutting patterns: a simulation approach. Landscape Ecol., 8: 63-75.-

Mack, R.N., 1986. Alien plant invasion into the intermountain west: a case history. In: H.A. Mooney and J.A. Drake (Editors), Ecology of Biological Invasions of North America and Hawaii. Springer-Verlag, NY, pp. 192–213.

McGarigal, K., and Marks, B.J., 1995. FRAGSTATS: Spatial pattern analysis program for quantifying landscape structure. Pacific Northwest Research Station, U.S. Department of Agriculture Forest Service, General Technical Report, PNW-GTR-351, Portland, OR, 123 pp.

McLaughlin, J.F. and Roughgarden, J., 1993. Species interactions in space. In: R.E. Ricklefs, and D. Schluter (Editors), Species diversity in ecological communities: historical and geographical perspectives, University of Chicago Press, Chicago, IL, pp. 89–98.

Means, J.E., 1982. Developmental history of dry coniferous forest in the central western Cascade Range of Oregon. In: J.E. Means (Editor), Forest succession and stand development research in the Northwest: Proceedings of a symposium; March 26, 1981, Corvallis, OR; USDA Forest Service, Portland, OR, pp. 142–158.

Megahan, W.F., 1987. Effects of forest roads on watershed function in mountainous areas. In: Balasubramaniam et al. (Editors). Environmental Geotechnics and Problematic Soils and Rocks. Balkema, Rotterdam, pp. 335–347.

Miller, C.D., 1969. Chronology of neoglacial moraines in the Dome Peak area, North Cascade Range, Washington. Arctic Alpine Res., 1: 49–66.

Morrison, P.H. and Swanson, F.J., 1990. Fire history and pattern in a Cascade range landscape. Pacific Northwest Research Station, U.S. Department of Agriculture Forest Service, General Technical Report, PNW-GTR-254, Portland, OR.

Noss, R.F., 1993. A conservation plan for the Oregon Coast Range: some preliminary suggestions. Nat. Areas J., 13: 276–290.

O'Neill, R.V., Krummel, J.R., Gardner, R.H., Sugihara, G., Jackson, B., DeAngelis, D.L., Milne, B.T., Turner, M.G., Zygmunt, B., Christensen, S.W., Dale, V.H. and Graham, R.L., 1988. Indices of landscape pattern. Landscape Ecol., 1: 153–162.

Overpeck, J.T., Rind, D. and Goldberg, R., 1990. Climate-induced changes in forest disturbance and vegetation. Nature, 343: 51–53.

Pickett, S.T.A. and White, P.S. (Editors), 1985. The ecology of natural disturbance and patch dynamics. Academic Press, NY.

Pielou, E.C., 1977. Mathematical Ecology. Wiley, NY.

Powers, J.S., 1995. Spatial and temporal dynamics of the Douglas-fir bark beetle (*Dendroctonus pseudotsugae*, Hopk.) in the Detroit Ranger District, Oregon: a landscape ecology perspective. M.S. Thesis. Forest Science Department, Oregon State University, Corvallis, OR.

Pyne, S.J., 1982. Fire in America: a cultural history of wildland and rural fire. Princeton University Press, Princeton.

Ripple, W.J., Bradshaw, G.A. and Spies, T.A., 1991. Measuring forest landscape patterns in the Cascade range of Oregon, USA. Biol. Conserv., 57: 73–88.

Robbins, W.G., 1988. Hard times in paradise, Coos Bay, Oregon, 1850–1986. University of Washington Press, Seattle, WA, 194 pp.

Romme, W.H., 1982. Fire and landscape diversity in subalpine forests of Yellowstone national park. Ecol. Monogr., 52: 199–221.

Rosenberg, K.V. and Raphael, M.G., 1986. Effects of forest fragmentation on vertebrates in Douglas-fir forests. In: J. Verner, M.L. Morrison and C.J. Ralph (Editors), Wildlife 2000: Modeling habitat relationships of terrestrial vertebrates. Univ. of Wisconsin Press, Madison, WI, pp. 263–272.

Rudinsky, J.A., 1962. Ecology of Scolytidae. Ann. Rev. Entomol., 7: 327–347.

Ruth, R.H. and Yoder, R.A., 1953. Reducing wind damage in the forests of the Oregon coast range. Res. Pap. 7. U.D. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station, Portland, OR.

Saunders, D.A., Hobbs, R.J. and Margules, C.R., 1991. Biological consequences of ecosystem fragmentation: a review. Conserv. Biol., 5: 18–32.

Schmidt, W., 1989. Plant dispersal by motor cars. Vegetatio, 80: 147–152.

Schoonmaker, P. and McKee, A., 1988. Species composition and diversity during secondary succession of coniferous forest in the western Cascade Mountains of Oregon. Forest Sci., 34 (4): 960–979.

Scott, H., 1924. History of the Oregon Country. Riverside Press, Cambridge.

Seagle, S.W., 1986. Generation of species-area curves by a model of animal-habitat dynamics. In: J. Verner, M.L. Morrison and C.J. Ralph (Editors), Wildlife 2000: Modeling habitat relationships of terrestrial vertebrates, Univ. of Wisconsin Press, Madison, WI, pp. 281–285.

Shugart, H.H. and West, D.C., 1981. Long-term dynamics of forest ecosystems. Am. Sci., 69: 647–652.

Southwood, T.R.E., 1977. Habitat, the template for ecological strategies? J. Anim. Ecol., 46: 337–365.

Spies, T.A., Franklin, J.F. and Thomas, T.B., 1988. Coarse woody debris in Douglas-fir forests of western Oregon and Washington. Ecology, 69: 1689–1702.

Spies, T.A., Ripple, W.J. and Bradshaw, G.A., 1994. Dynamics and pattern of a managed coniferous forest landscape. Ecol. Applic., 4: 555–568.

Swanson, F.J. and Franklin, J.F., 1992. New forestry principles from ecosystem analysis of Pacific Northwest forests. Ecol. Applic., 2: 262–274.

Swanson, F.J., Franklin, J.F. and Sedell, J.R., 1990. Landscape patterns, disturbance and management in the Pacific Northwest, USA. In: I.S. Zoneveld and R.T.T. Forman, (Editors), Changing landscapes: an ecological perspective. Springer-Verlag, NY, pp. 191–211.

Swanson, F.J., Jones, J.A., Wallin, D.O. and Cissel, J., 1993. Natural variability: Implications for ecosystem management. In: M.E. Jensen and P.S. Bourgeron (Editors) Eastside forest ecosystem health assessment—Volume II: Ecosystem management: principles and applications. Portland, OR: U.S. Dept. of Agriculture, Forest Service, Pacific Northwest Research Station, pp. 89–103.

Teensma, P.D.A., 1987. Fire history and fire regimes of the central western Cascades of Oregon. Ph.D. Dissertation, University of Oregon, Eugene, OR, 188 pp.

Thomas, J.W., Forsman, E.D., Lint, J.B., Meslow, E.C., Noon, B.R. and Verner, J., 1990. A conservation strategy for the northern spotted owl. USDA Forest Service and USDI National Park Service. Portland, OR, 427 pp.

Turner, M.G., Wear, D.N. and Flamm, R.O., 1996. Land ownership and land cover change in the Southern Appalachian Highlands and Olympic Peninsula. Ecol. Applic., in press.

Tyser, R.W. and Worley, C.A., 1992. Alien flora in grasslands adjacent to road and trail corridors in Glacier National Park, Montana (USA). Conserv. Biol., 6: 253–262.

Urban, D.L. and Smith, T.M., 1989. Microhabitat pattern and the structure of forest bird communities. Am. Nat., 133: 811–829.

USDA Forest Service, 1990. Land and resource management plan: Willamette national forest. Pacific Northwest Region, USDA Forest Service, Portland, OR.

USDA Forest Service and USDI Bureau of Land Management, 1994. Record of decision for amendments for Forest Service And Bureau of Land Management planning documents within the range of the northern spotted owl. USDA Forest Service and USDI Bureau of Land Management, Portland OR, 74 pp.

Waddell, K.L., Oswald, D.D. and Powell, D.S., 1989. Forest statistics of the United States, 1987. Pacific Northwest Forest and Range Experiment Station, U.S. Department of Agriculture Forest Service, Resource Bulletin PNW-RB-168, Portland, OR.

Wallin, D.O., Swanson, F.J. and Marks, B., 1994. Landscape pattern response to changes in pattern generation rules: land-use legacies in forestry. Ecol. Applic., 4: 569–580.

Waring, R.H. and Franklin, J.F., 1979. Evergreen coniferous forests of the Pacific Northwest. Science, 204: 1380–1386.

Yamaguchi, D.K., 1993. Forest History, Mount St. Helens. Nat. Geogr. Res. Explor., 9 (3): 294–325.

Zheng, D., Wallin, D.O. and Hao, Z., 1996. Detecting rates and patterns of landscape change in the Changbai Mountain area between 1972 and 1988. Landscape Ecol., in press.