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Fire and Weather Disturbances in Terrestrial Ecosystems of the Eastern Cascades

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Eastside Forest Ecosystem Health Assessment

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ABSTRACT

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Fire has been an important ecological process in eastside Cascade ecosystems for millennia. Fire regimes ranged from low severity to high severity, and historic fire return intervals ranged from less than a decade to greater than 300 years. Fire history and effects are described for grassland and shrubland ecosystems, and the range of forested communities by plant series: Ponderosa Pine, Douglas-fir/White fir/Grand fir, Lodgepole pine, Western hemlock/Western redcedar, and subalpine fir/Mountain hemlock. The riparian zones within these communities may be more or less impacted by fire. The effects of extreme weather events, including unusual temperature, wind, or moisture have generally had less significant impact than fire. Management practices, including fire suppression, timber harvesting, and livestock grazing, have altered historical fire regimes, in some cases irreversibly. The management issues for the 1990s include both management and research issues, at a grand scale with which we have little experience. Ecosystem and adaptive management principles will have to be applied.

Keywords: Forest fire, fire history, *Juniperus occidentalis, Pinus ponderosa, Pseudotsuga menziesii, Abies grandis, Abies lasiocarpa, Pinus contorta.*

Contents

INTRODUCTION

Fire has been an important disturbance process for millennia in forested wildlands east of the Cascade crest in Oregon and Washington (fig. 1). Records from early explorers and dates of frequent fires measured on many older scarred trees suggest that fires burned at frequent intervals in many eastside forests and grasslands. Historical fire regimes are important to understand because they were part of a set of ecosystem processes and states that appear relatively more sustainable compared to the state of these ecosystems today.

Fire, in these forests, has been described as both "benign" and "catastrophic." To make such judgments requires an understanding of how fire interacts with wildlands and whether this interaction is desirable. Solutions to the eastside forest health situation will require making both scientific and value decisions. This paper focuses on the scientific aspect of fire and weather disturbances in eastside forests, with discussion in disturbance regimes over the past century or more. Fire has been a variable in both space and time: low to high intensity, frequent to infrequent, and of small to large extent.

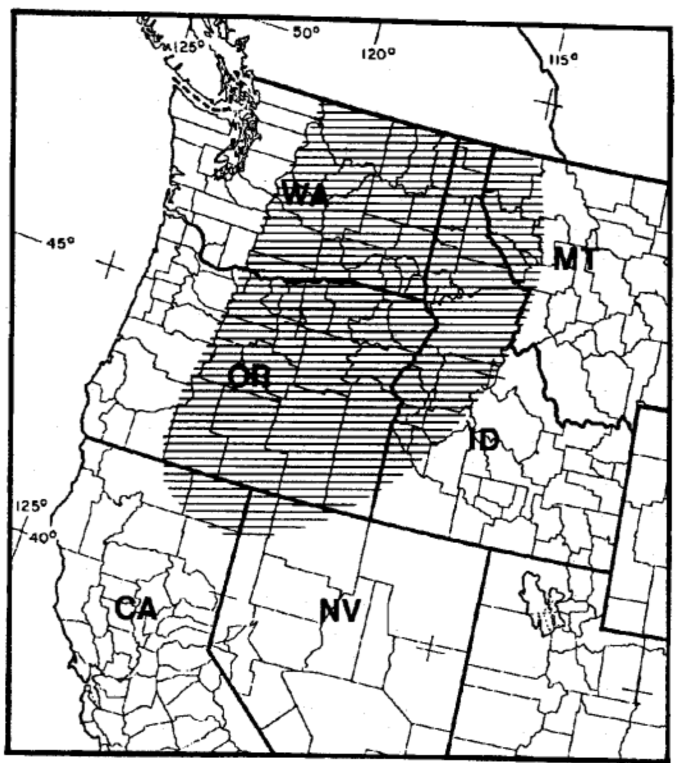


Figure 1. The focus of this review of fire is on ecosystems east of the Cascade crest.

In prehistoric times, some forests burned every 5 to 10 years, but others had fire-return intervals of a 100 years or longer. Fire patterns have changed markedly during the past century. A policy of suppressing all fires, selectively harvesting fire-resistant tree species, and livestock grazing has altered historical fire frequencies, intensities, and extents-collectively known as a fire "regime." Changes in fire regime, more dramatic in some eastside forest ecosystems than in others, are associated with changes in ecosystem health and sustainability. Using fire in ecosystem management strategies will require addressing some public perceptions and misconceptions about fire and improving forest managers' ability to predict the effects of fire on landscapes and ecosystem components.

Extreme weather events are much less manageable than. fire, but may still have significant impact on stand structures and desired future conditions for eastside forests. Some weather effects, however, are a function of stand structure, so that appropriate management can reduce detrimental effects.

FIRE AS A DISTURBANCE PROCESS

Although natural disturbances of many types have been present in wildland ecosystems for millennia, only recently have scientists begun to quantify their importance in ecosystem structure and function (White and Pickett 1985). During most of the 20th century, the concept of ecosystem disturbance supposed that disturbances must be major and catastrophic. Because they were assumed to originate in the physical environment, disturbances were viewed as an external agent of ecosystem change. In the past two decades ecologists recognized that disturbances spanned a wide gradient of intensities, and many disturbances were at least partly a function of the biotic state of the ecosystem: fuel buildup that affected fire intensity or stressed trees that were more vulnerable to insect attack. Some disturbances, such as insect or disease outbreaks, originate within systems, so disturbance can be an internal agent of ecosystem change.

Fire as a Disturbance Agent

Fire is a classic disturbance agent: discrete in time, affecting ecosystem function and structure, altering the physical environment. Gross generalizations about fire effects are risky, but predictions are possible if fire is described in a specific ecosystem and quantified by its characteristics: frequency, intensity, extent, seasonality, and its relation to other disturbances. Unfortunately, these parameters are not well understood for most ecosystems, including those on the eastside of Oregon and Washington.

The characteristics of fire are important in understanding its direct and indirect effects. Fire frequency, its return interval, is measured by counting years between fire scars on trees or by analyzing age classes of forest stands remaining after fires. Predictability is the variation in frequency. Together, frequency and predictability are important fire characteristics that determine species presence and dominance on the landscape (fig. 2).

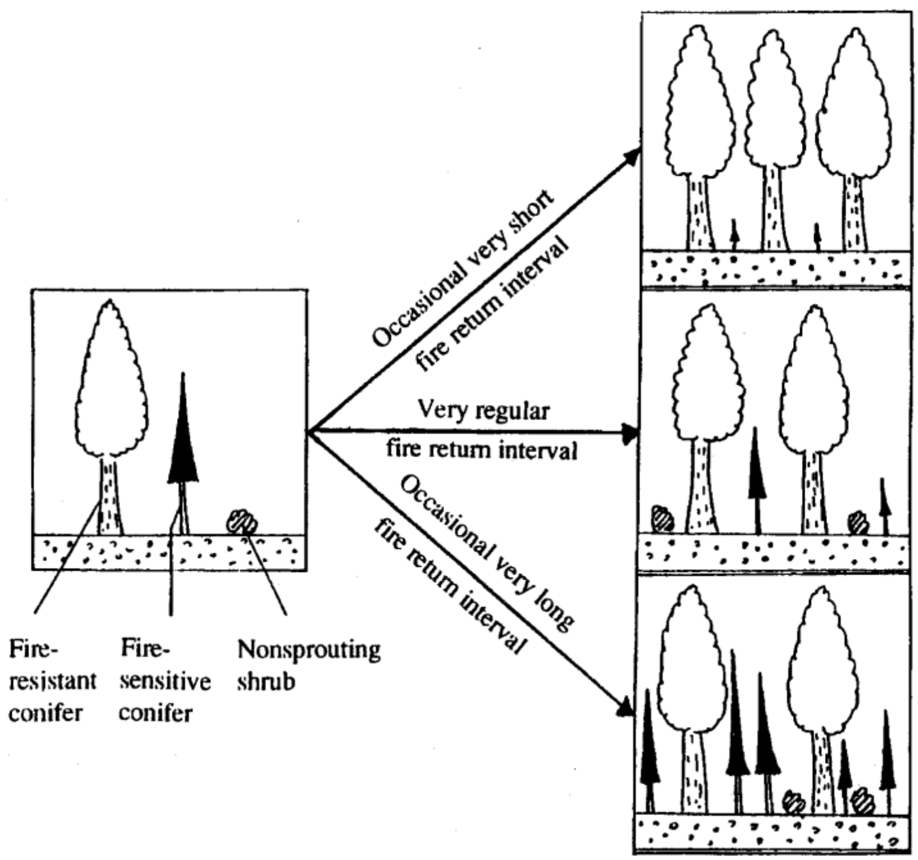


Figure 2. Variation in fire return intervals can result in different vegetation response, even when the mean fire return interval is the same. Short fire return intervals reduce nonsprouting species, and long intervals allow some species to pass through a fire-intolerant stage when young.

The magnitude of a fire is most commonly described by its intensity. Fireline intensity, the heat release rate along a unit length of fire front, is directly related to flame length. In this paper, three categories will be used to describe fireline intensity (FLI). **Surface** fire is the lowest intensity category, with flame lengths to 1 m and FLI below 400 kW m^{-1}. **Understory** fire is of intermediate intensity, with flame lengths between 1 and 3 m and FLI between 400 and 1600 kW m^{-1}. Heat and smoke can be excessive, and fire behavior can be erratic. **Crown** fire is the highest intensity category, with flame lengths above 3 m and FLI above 1600 kW m^{-1}. Many historical (pre-1850) eastside forest fires were in the surface fire category, but historical fires span the entire range of intensities, including crown fires. The ability of some species to either survive or regenerate after fire is significantly related to fire intensity.

The geographic extent of historical fires is poorly known. Cross-dated fire scars can be used to develop maps of fire extent in ecosystems with frequent low-intensity fires. Even this technique is incomplete because such fires would not have scarred every tree within the fire perimeter. In ecosystems that historically burned with high fire intensity, fire extent may be obvious for a century or more from the mosaic pattern of different-aged stands. Most fires east of the Cascade crest must have burned during the summer when lightning strikes were common and fuels were dry. Without fire suppression, fires may have burned sporadically from time of ignition until October or November.

Fire often interacts synergistically with other disturbances. Low-intensity frequent underburning may discourage bark beetle outbreaks by controlling stand density and reducing competitive stress on residual trees. Conversely, fire may encourage bark beetle attack on damaged trees. By creating open landscapes, intense fires increase the effect of rain-on-snow events by making more of the snow-covered landscape susceptible to the impact of warm raindrops. Stand-replacement fires can increase erosion by reducing fine-root biomass that held marginally stable soil in place. Fires inhibit some fungi and dwarf mistletoe through the effects of smoke (Parmeter and Uhrenholdt 1976), but encourage butt decay by opening wounds for entry of decay organisms. Synergistic effects of fire and other disturbance factors make fire a fascinating and complex disturbance.

The Fire Regime

A fire regime is a generalized way of integrating various fire characteristics. The organization may be according to the characteristics of the disturbance (for example, Heinselman 1973), dominant or potential (climax) vegetation on the site (Davis and others 1980), or fire severity, the magnitude of effects on dominant vegetation (Agee 1990). In this paper, fire regimes will be defined at historical scales by the potential climax vegetation (for example, the grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.), and by fire severity within each climax series (fig. 3, 4). Changes to fire regimes resulting from management activities will also be discussed.

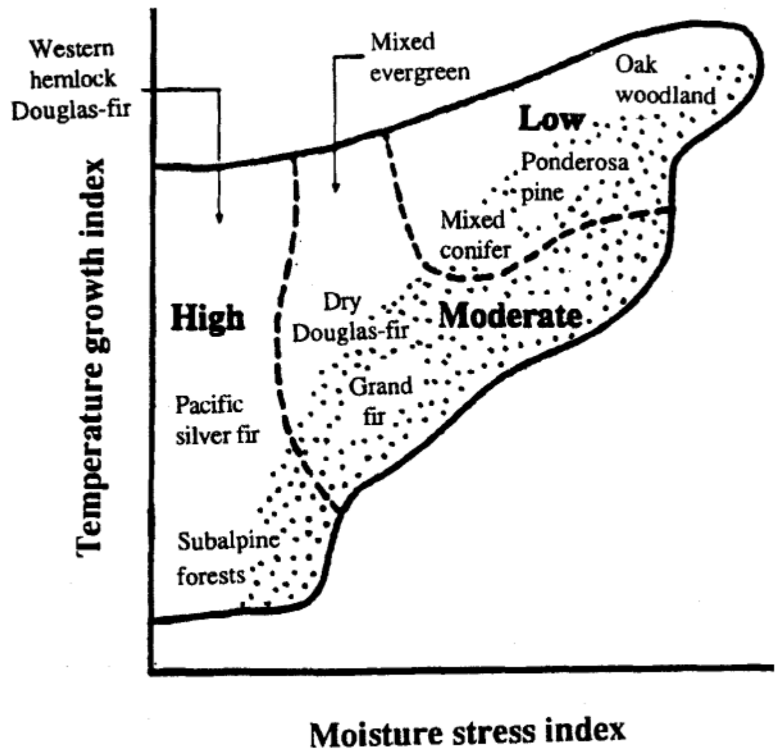


Figure 3. The fire severity regimes of the Pacific Northwest (Low, Moderate, High) can be displayed in the matrix of Pacific Northwest forest types (based on potential climax vegetation) ordinated by growing-season temperature and moisture stress (Agee 1990). Eastside ecosystems are indicated by a stippled pattern.

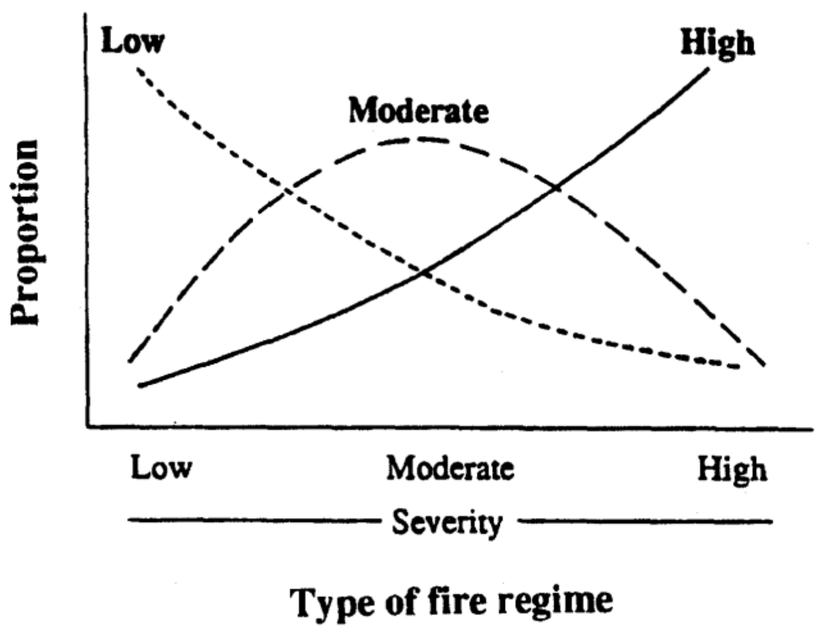


Figure 4. Within each fire severity regime is a range of fire severities, either in the same fire or between fires (Agee, in press).

In the low severity fire regime, the predominant fire severity level results in tree basal area reductions of 20 percent or less. Low severity surface fires are most common, but moderate and high severity fires are possible, although uncommon. At the other end of the scale, in the high severity fire regime, the predominant fire severity level results in tree basal area reductions of 70 percent or more. A high proportion of fires are of understory and crown intensity. The middle of the scale is the moderate severity fire regime, with a complex mixture of low, moderate, and high severity fires. Historically, eastside forests had all three natural fire regimes.

Fire severity, being defined in part by its effect on ecosystems, is also a function of plant responses to fire. Many plants have adapted to particular fire regimes by developing survival mechanisms.

Fire Adaptations of Plants

Fire has predictable effects on each plant species and time-temperature sequences can be estimated from fireline intensities. These estimates have been used to predict crown scorch height (Van Wagner 1973) and cambial damage (Peterson and Ryan 1986). Predicting root damage is more difficult because of variations in rooting patterns and soil properties. Fire has interacted with individual plant species for millions of years, plants and trees have developed a variety of adaptations to fire, some of which allow them to persist even in the presence of fire; others allow the population to persist even though individuals may be killed (Kauffman 1990). Many plants in eastside forests have such adaptations.

Thick bark is a common adaptation to fire found in trees, particularly where fires were frequent and of low-to-moderate intensity. In eastside forests, mature ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) , western larch (*Larix occidentalis* Nutt.), and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) are insulated for an hour or more against lethal temperatures because of their thick, corky bark, which is a better insulator than asbestos of equal thickness (Martin 1963). This adaptation is most effective in low-intensity fires, which do not scorch the crown but may smolder at the base.

Many shrubs and deciduous trees respond to fire by releasing dormant buds under the bark of fire-scorched stems and branches. This epicormic sprouting occurs when fire is intense enough to kill the live foliage but does not persist long enough to create temperatures lethal to the vascular cambium. Mountain dogwood (*Corpus nuttallii* Audubon) and Oregon ash (*Fraxinus latifolia* Benth.) are examples of such species. Basal sprouting also occurs in many epicormic sprouting species; in other species, such as snowberry (*Symphoricarpos albus* (L.) Blake), ninebark (*Physocarpus malvaceus* (Greene) Kuntze), Scouler's willow (*Salix scoulerana* Barratt ex Hook.), huckleberries (*Vaccinium* spp.), pinegrass (*Calamagrostis rubescens*) and rabbitbrush (*Chrysothamnus* spp.), basal sprouting occurs from root crowns or rhizomes. New growth of some grasses, such as Sandberg's bluegrass (*Poa sandbergii* Vasey) and bluebunch wheatgrass (*Agropyron spicatum* (Pursh) Scribn. & Smith), is well-protected from frequent, low-intensity fires by older foliage.

Other species are adapted to long-term survival, even though current populations are killed by fire (Kauffman 1990). The important seral shrub ceanothus (*Ceanothus spp.*) and the herbaceous lupine (*Lupinus* spp.) produce seeds that can lie dormant for decades awaiting scarification by fire. After fire, high densities may be found in locations where no live individuals existed at the time of the fire. A similar seed-bank strategy is exhibited by the serotinous cones of some varieties of lodgepole pine (*Pinus contorta* Dougl. ex Loud), except that the seeds are stored in the canopy of the tree. Seeds are protected beneath resin-sealed cone scales and can remain viable until a fire passes through and melts the resin seal. Trees may die but seeds are undamaged and fall to a newly fertilized ashbed. Some plants are stimulated to flower after burning, such as Great Basin wildrye (*Elymus cinereus*) or pinegrass. Others have wind-blown seed that rapidly invades burned sites, such as fireweed (*Epilobium angustifolium* L.). These adaptations ensure that some vegetative recovery or reestablishment is likely after fire passes through an ecosystem (Stickney 1990).

The Fire Environment

Eastside forest ecosystems exhibit numerous adaptations to a classic "fire environment"-a set of environmental conditions conducive to the recurring presence of fire. Fire behavior can be predicted based on fuels, weather, and topography. Ignition from lightning and by Native Americans over the past few millennia, together with factors influencing fire spread and behavior, have shaped landscape composition and structure. A brief summary of important fire behavior components is presented to help interpret fire effects information for eastside plant series.

A source of fire ignitions with an extended evolutionary track is lightning. Eastside forests are "hotspots" for lightning storms (fig. 5). Based on historical data, variation in fire activity would be expected even within plant associations, depending on location. Higher frequency lightning locations should have shorter fire return intervals than locations with lower lightning potential. August has the highest potential for lightning ignitions (USDA Forest Service 1981).

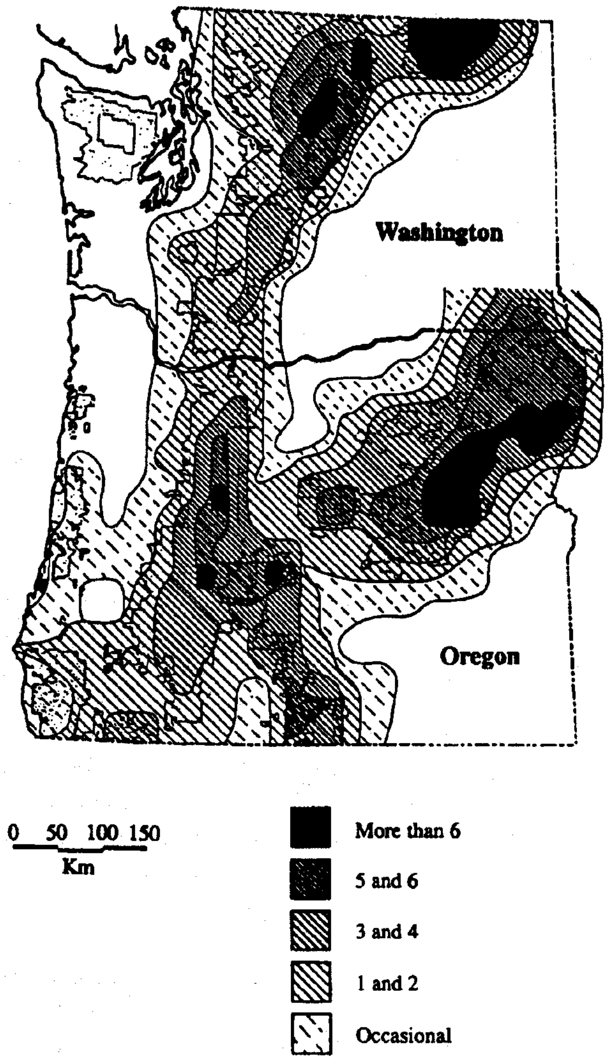


Figure 5. Lightning activity across Oregon and Washington (from Morris 1934). The Blue Mountains, north central-Washington Cascades, and Okanogan Highlands are "hot spots" for lightning.

Once a fire occurs, fire behavior is a function of fuels, weather, and topography. Throughout the low elevation grasslands, woodlands, and forests, grasses and forbs were important fine fuels that allowed surface fires to spread. After European settlement began around 1850, heavy livestock grazing significantly decreased these fuels. At all elevations, dead conifer needles on the forest floor remain important vectors for fire spread. At high elevation, fire spread is augmented by live shrub and tree foliage, but these fuels are only available late in the growing season or during droughts (Williams and Rothermel 1992). Because changes in vegetation correspond with changes in elevation, the plant series is a useful way to describe fuel profiles, but the location and extent of a plant series often require local interpretation (in addition to the lightning patterns described above). For example, the historical effects of fire in a small patch of grand fir forest surrounded by cooler, wetter subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) forest will be markedly different than if the small patch were surrounded by drier, warmer ponderosa pine forest.

Weather is important also for patterns of precipitation and wind that affect fire behavior. Protected from Pacific storms by the Coast and Cascade mountain ranges, the eastside forests receive much less precipitation than mountain ranges to the west. Forests near the Cascade crest, adjacent to the Columbia River and in the northern Blue Mountains, are more maritime (not as cold or hot; more precipitation) than other eastside forests, which are more continental (hotter and colder; less precipitation). Most of the annual precipitation, which ranges from 10 to 250 cm, is received between late fall and early spring, while summer precipitation is usually light.

Local winds associated with differential heating of the landscape are important throughout eastside forests: up-valley winds during the day, down-valley at night (Schroeder 1970). Topographic influences interact with weather, but have direct effects on fire as well. Steep slopes are more prone to burn than flat ones, southerly aspects more than northerly, and ridgelines more than valley bottoms. Warm, dry foehn or Chinook winds are common in eastside forests. They occur several times a month during the summer and fall and can quickly dry the fine fuels that carry fire. Such winds can also be strong. Valleys that trend east-west, with low saddles at their crests are more often affected by these wind patterns than are north-south valleys or areas with more topographic relief.

An important exception to typical up-valley afternoon winds occurs in major drainages that flow from the Cascade crest to the east. Daytime heating of the Columbia basin creates a thermal low which draws air from the relatively higher pressure areas of the adjacent Cascade mountains. This configuration creates a down-canyon afternoon wind, which can significantly affect fire behavior (Schroeder 1961). Older trees in the Entiat and Chelan valleys of north-central Washington have fire scars on their down-canyon sides. Because fire scars usually occur on the lee sides of trees when a fire is windblown, and because most fires spread fastest in the afternoon, such scars are evidence of historical afternoon down-valley winds. A recent example of afternoon down-valley winds occurred at the Prophecy fire near Crater Lake, when a thermal low east of the incident area created a strong afternoon wind drawn across the Cascade crest to the east, allowing a fire to burn down-valley at high intensity across several hundred hectares.

Forest fires do not normally consume all aboveground biomass, although they can in ecosystems composed primarily of fine fuel, such as grasslands or chaparral. In forested ecosystems, a higher proportion of dead fuel than live fuel is consumed because moisture contents of the dead fuels are generally lower. Normally, more small than large fuel is consumed, although some exceptions occur (see section on lodgepole pine forests). Biomass available as fuel may be as little as 15 percent of the total aboveground for westside Cascade fires (Fahnestock and Agee 1983) but might be somewhat higher for eastside forests. Most fuel consumption is aboveground, but old stumps and roots will commonly burn out in eastside ecosystems.

Each fuel size class has a different surface-area-to-volume ratio (SA/V), and is thereby able to gain and lose moisture at a unique rate. This ability is characterized in a concept known as timelag class, a descriptor of the time required for a cylindrical fuel particle to move 63 percent of the way to a new equilibrium moisture content. Fine fuels, those below 0.63 cm diameter, have a large SA/V, requiring only 1 h to move 63 percent of the way to a new equilibrium moisture content. They are therefore called 1-h timelag fuels. For example, a 1-h timelag fuel at 10 percent moisture, placed in a drier environment where the equilibrium moisture content is 5 percent, will be at about 7 percent moisture after 1 h. Fuel size classes and their timelag equivalents are: 1-h timelag, 0-0.63 cm; 10-h timelag, 0.64-2.54 cm; 100-h timelag, 2.55-7.62 cm; 1000-h timelag 7.63-20.32 cm. In the field, environmental conditions are always changing, as are moisture contents of the various timelag classes of fuel. During dry summer months, 1-h and 10-h timelag fuels are typically at 5 to 10 percent moisture by dry weight, 100-h timelag fuels at 10 to 15 percent, and 1000-h timelag fuels at 13 to 18 percent.

Live fuels also contribute to fire behavior. Waxy-leaved shrubs often burn in surface fires, and coniferous tree crowns may combust under severe fire weather or dense stand conditions. New leaves typically have more than 200 percent moisture (2 g water/g of tissue), but as the growing season progresses, cell walls thicken and moisture content decreases, so that moisture of late-season leaves is around 100 percent. Older foliage and small twigs remain at that percentage. Foliage above 120 percent moisture rarely burns (Chandler and others 1983). Unusual weather or fire conditions can drop live fuel moisture below 100 percent, and foliar flammability seems to increase once fuels are Z percent moisture content (Agee, in press).

The density of fuels affects oxygen supply and can affect fire behavior. Fuelbeds can either be too dense or too loose to provide optimum oxygen flow or heat transfer. The “packing ratio,” or proportion of th fuelbed volume occupied by fuel particles, can range from grassland (0.001) to tightly packed litter (0.1).

Computerized models of fire behavior have been developed that integrate the effects of fuels, weather, and topography on spread, heat release, and intensity of wildland fires (Rothermel 1983). A variety of standardized fuel models characterize typical grass, shrub, or timber-dominated fuels (Albini 1976). When one of these models is selected or a site-specific fuel model is developed (Burgan and Rothermel 1984), fuel moisture of the timelag classes (below 1000-h) can be entered, as well as live fuel moisture, slope, and windspeed. Outputs include rate of spread and fireline intensity (Burgan and Rothermel 1984); fireline intensity is directly related to scorch height (Van Wagner 1973).

The components of fire behavior influence fire effects. Current fire models focus on the flaming front, but they do not provide sufficient resolution for modeling precise fire effects, such as tree mortality or smoke production. They are helpful in interpreting the eastside forest landscape, however, and suggest how alternative management uses of fire might be modeled in the future.

HISTORICAL FIRE REGIMES OF EASTSIDE ECOSYSTEMS

The historical fire regime is described by those combinations of fire severities that occurred before significant European influence, generally before 1850. We can argue that the influence of horses imported to the continent by Europeans in the 1700s might require an earlier date, but little evidence exists in eastside forests that significant European influence was exerted until after 1850. Historical fire regimes are influenced by a long-standing history of Native American ignitions.

Evolution of Present-Day Vegetation

Long-term climatic changes have caused individual plant species to migrate across landscapes, and fire regimes have changed correspondingly. Little paleoecological exploration has been done in eastside areas compared to other areas of North America, but enough is known to understand major plant migration during the late Holocene (Barnosky and others 1989). The pollen record of the Okanogan Highlands in eastern Washington, near the terminus of the continental ice sheet at the end of the last glaciation, provided evidence of widespread sagebrush-grass vegetation at that time with isolated pockets of forest tree species, including haploxylon pine (the soft or white pines, such as whitebark (*Pinus albicaulis* Engelm.) or western white pine (*P. monticola* Dougl, ex D. Don) and spruce and fir. Present-day relict species, such as the small riparian population of Alaska-cedar (*Chamaecyparus nootkatensis* (D.Don) Spach) at the Cedar Grove Botanical Area on the Malheur National Forest, were probably more widely distributed. Earlier hypotheses that the Columbia Basin was a lodgepole pine parkland (Hansen 1947) are now dismissed because of the low percentages of diploxylon (hard pines such as ponderosa or lodgepole) pine pollen at most late-glacial sites.

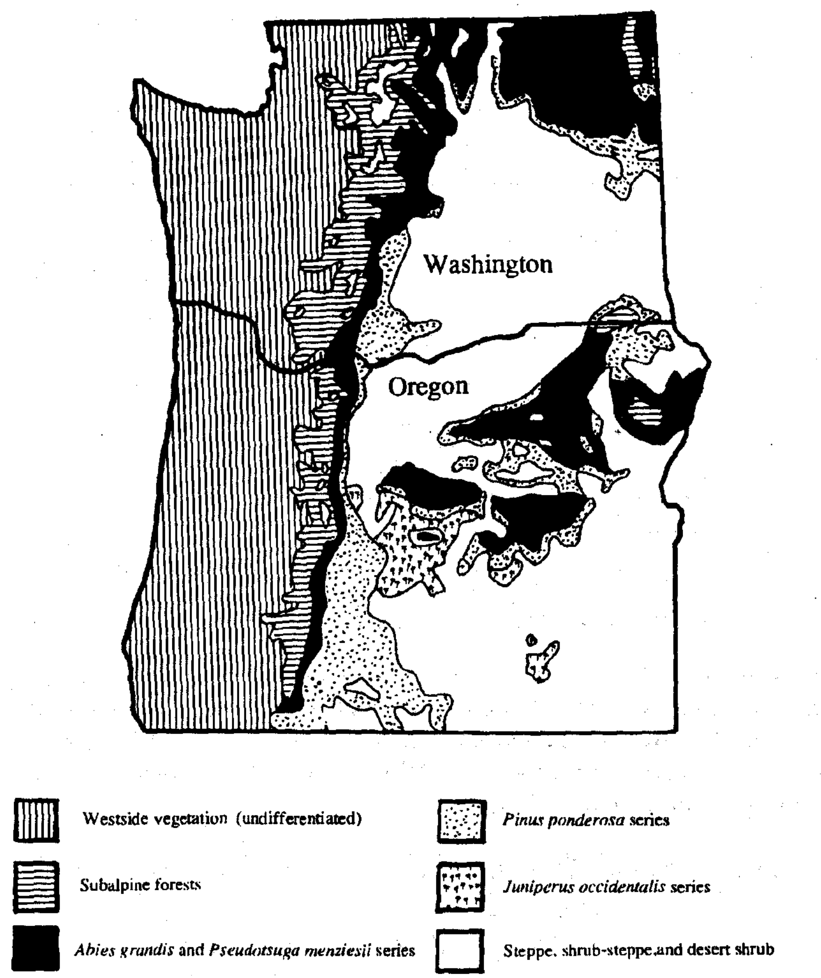
Most sites show evidence of increased summer drought beginning between 11 and 9 millennia B.P. The warmer, drier period commenced and ended at different times across the region. At Carp Lake, a lowland site 150 km west of the Blue Mountains, ponderosa pine replaced steppe vegetation 8500 years ago, and it persists to the present (Barnosky 1985). Similar inferences may be extended to much of the current ponderosa pine series. In more southerly areas of the Columbia Plateau, the range of western juniper has expanded and contracted over the past 5000 years (Mehringer and Wigand 1987), suggesting a rough balance in climatic shifts over that time. In subalpine areas, one of the closest regional sites is Lost Trail Bog Pass in the Bitterroot Mountains, Montana, which changed from sagebrush-grass vegetation before 12,000 B.P. to lodgepole pine/Douglas-fir by 7000 B.P. (Mehringer and others 1977). During this warming, many boreal species disappeared from eastside landscapes, or moved upslope to form isolated patches, such as Alaska-cedar in the Blue Mountains and the Siskiyou Mountains (Whittaker 1961).
The implication for fire regimes is that individual species have coevolved with fire longer than with any

particular plant community. Today's plant communities and their fire regimes represent the environment and species mix of the last few millennia at most. A corollary to this is that if global climate changes, these past fire regimes cannot be projected very far into the future as the operative pattern.

Current Vegetation and Fire Regimes

Today's eastside forests include a wide array of ecosystems, from xeric sagebrush-steppes to alpine meadows (fig. 6). In this analysis, fire history and effects will be described for each of the major vegetation groupings. The plant association concept will be the model by which fire history and effects will be organized. A single plant association will encompass several plant communities on a successional trajectory, but the association is named for the successional endpoint, or climax, community. The climax community is defined by the most shade-tolerant overstory species and understory species that will eventually dominate the site in the absence of disturbance. Disturbances, such as fire, result in seral community dominance; protection from disturbance allows community progressions towards climax.

Figure 6. Major vegetation types of eastside ecosystems (adapted from Franklin and Dyrness



19731. The oak woodland type is not mapped but extends in a 50-km-wide band north and south of the Columbia River into the edge of the shrub-steppe type. Nonforested alpine areas are shown as small nonshaded enclosures within the subalpine forest types.

A plant series consists of all the plant associations ultimately dominated by a single, shade-tolerant climax species. Two communities, for example, may be dominated by ponderosa pine in the presence of repeated low-intensity fires. If protected from fire, one might remain dominated by ponderosa pine, although the structure of the community would change; this community would be part of the ponderosa pine series. The other community might become dominated by Douglas-fir and would be part of the Douglas-fir series. The implications of presence or absence of disturbance on species composition, horizontal and vertical structure, and function are well described by such a classification system.

The tree species occurring across eastside forests are found in more than one of the major plant series (fig. 7). Their adaptations to fire (table 1), in combination with characteristics of historical fires of the past, have resulted in the fire regimes of eastside forests.

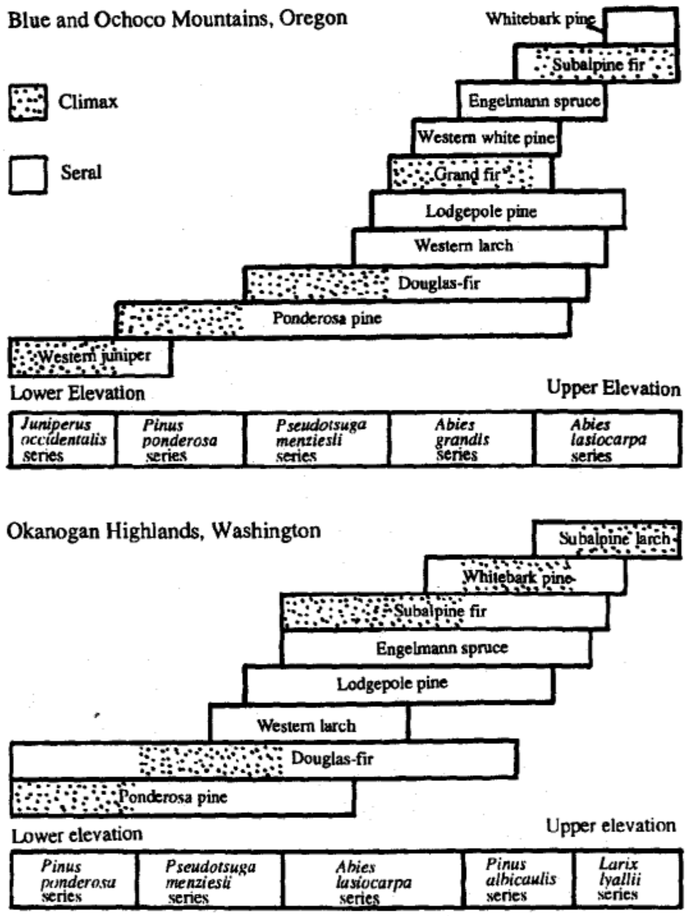


Figure 7. Environmental orientation of the major tree species of two eastside ecosystems. Length of bar denotes range of species, with stippled areas that portion of the range in which a particular species is climax. Top. The Blue and Ochoco Mountains (from Johnson and Clausnitzer 1991). Bottom: The Okanogan Highlands (Williams and Lillybridge 1983). Individual species such as ponderosa pine are found over a broader range of plant series than the one in which it may be a climax dominant.

Table 1—Major tree species of the Eastern Cascades and their response to fire

Tree species	Response to fire
Western juniper	Avoider -- easily killed at young or mature stages
Oregon white oak	Resister -- has thick bark, can also sprout new crown if scorched (endurer)
Ponderosa pine	Resister -- has thick bark that developes at an early age
Douglas-fir	Resister -- has thick bark when mature but suseptible to fire when young
Western larch	Resister -- has thick bark and develops at an early age
Grand fir	Avoider -- thin bark when young but moderately resistant when mature
Lodgepole pine	Evader -- thin bark even when mature, but has serotinous cones (var. latifolia)
Quaking aspen	Endurer -- thin bark, easily top-killed, but sprouts readily after burning
Subalpine fir	Avoider -- thin bark, shallow-rooted, almost always killed by fire
Engelmann spruce	Avoider -- same as subalpine fir
Mountain hemlock	Avoider -- same as subalpine fir
Whitebark pine	Moderate resister -- thin bark but usually grows in fuel-limited environments with patchy fire

The major fire regimes are discussed in related groups. The grassland, shrubland, oak woodland, and the western juniper (*juniperus occidentalis* Hook.) series are grouped. The ponderosa pine series and lodgepole pine series are discussed separately. The western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) and western redcedar (*Thuja plicata* Donn ex D. Don) series, which are not widely distributed across eastside landscapes, the Douglas-fir, white fir (*Abies concolor* (Gord & Glend.) Lindl. ex Hildebr.), and grand fir series, and the subalpine fir and mountain hemlock (*Tsuga mertensiana* (Bong.) Carr.) series, are discussed as combined groups. Riparian areas are discussed as a "group" because of their importance in eastside landscapes, although they are intermixed as linear corridors within the series mentioned above.

Fire History Methods for Eastside Ecosystems

Almost all of the vegetated land east of the Cascades has burned repeatedly over the past 1000 years. Delineating that fire history is sometimes difficult because most of the information must be salvaged from living trees. Grasses retain little evidence of fire, and long-lived shrubs may be useful in dating only the last fire by estimating the age of germinated plants or sprouts from the root crown. More quantified estimates of fire return intervals are available for forested sites.

Two types of fire interval estimation have been used to determine forest fire history: point and area estimates (Agee, in press). Point intervals are used in low-severity fire regimes where fire-scarred residual trees are common. For example, the scar record of an individual tree is an estimate of point frequency, although not all fires will scar every tree. The record of an individual tree is a conservative estimate of fire return interval. Fire scar records of nearby trees may be combined (if crossdated to be synchronous) to produce a better estimate of point frequency, called a "composite fire interval," though this combined record is now distributed over an area rather than a point (table 2). If too wide an area (generally over 10 to 15 ha is selected, the composite fire interval loses its relevance as an estimate of point frequency. Where fire regimes are of moderate-to-high severity, various area frequency methods are used to determine fire history. These methods rely on reconstructing past fires by age classes of stands over the landscape (natural fire rotation, Heinselman 1973) or by using the present age-class distribution (negative exponential or Weibull distribution, Johnson and Van Wagner 1985).

Table 2—Fire return intervals for the eastside Cascades major plant series. Some of the cited studies are from related vegetation types in nearby regions

Fire return interval, years	Method [1]	Location	Source
Western juniper series			
15-20	CFI	Nevada	Young and Evans (1981)
7-17	P	N. California	Martin and Johnson (1979)
25	CFI (260)	S.W. Idaho	Burkhardt and Tisdale (1976)
Ponderosa pine series			
16-38	CFI (16)	E. Oregon	Bork (1985)
7-20	CFI (16)	E. Oregon	Bork (1985)
11-16	P	E. Oregon	Weaver (1959)
3-36	P	E. Oregon	Soeriaatmadja (1966)
Douglas-fir series			
7-11	CFI (20)	E. Washington	Wischnofske and Anderson (1983)
10	P	Blue Mountains	Hall (1976)
10-24	P	E. Washington	Finch (1984)
14	CFI (40)	E. Washington	Finch (1984)
8-18	P	E. Washington	Agee (unpublished data)
White fir series			
9-42	CFI (1)	S. Oregon	McNeil and Zobel (1980)
9-25	CFI (16)	C. Oregon	Bork (1985)
9-18	CFI (1)	C. California	Kilgore and Taylor (1979)
Grand fir series			
16.6	CFI (25)	E. Washington	Schellhaas (pers. comm.)
47	P	E. Oregon	Weaver (1959)
33-100	CFI (30)	E. Washington	Wischnofske and Anderson (1983)
17 [2]	P	Montana	Arno (1976)
100-200 [3]	--	Montana	Antos and Habeck (1981)
Lodgepole pine series			
60	P	S. Oregon	Agee (1981)
60	P	S. Oregon	Stuart (1984)
Western hemlock/Western redcedar series			
50-200+	CFI (100)	N. Idaho	Arno and Davis (1980)
50-100 [2]	--	Montana	Davis and others (1980)
150-500 [3]	--	Montana	Davis and others (1980)
Subalpine fir series			
25-75 [2]	--	Montana	Barrett and others (1991)
109-137	NFR	C. Washington	Agee and others (1990)
140-340 [3]	--	Montana	Barrett and others (1991)
250	--	E. Washington	Fahnestock (1976)
50-300 [4]	CFI	Montana	Arno (1980)
29 [4]	CFI (100)	Montana	Morgan and Bunting (1990)

[1] P=point or individual tree, CFI=composite fire interval, with size of area in parentheses, NFR=natural fire rotation

[2] Intermediate intensity fire return interval

[3] Stand-replacement fire return interval

[4] Stands dominated by whitebark pine

(1) * Stands are 70 percent Abies grandis series, 20 percent Pseudotsuga menziesii series, 8 percent Tsuga heterophylla series, and 3 percent Abies amabilis series

Many fire history studies have been completed in eastside forests or in neighboring areas where information is applicable. These studies will be discussed by plant series. An important limitation of the fire return interval is the analytical method used. Several unpublished studies are also included in the discussion.

Natural landscape-scale patterns of the various vegetation types are described. At the landscape scale, these patterns are largely unquantified for eastside forests, so I present these sections as unreferenced and speculative. The speculations consist of a set of hypotheses that I think are reasonable given the empirical evidence for patterns of stands. The “natural landscape” is arbitrarily defined here as pre-1850, to contrast changes resulting from management over the past century.

GRASSLAND, SHRUBLAND AND WOODLAND ECOSYSTEMS

Fire History

Little knowledge of historic fire frequency in grasslands, shrublands, and woodlands of the Columbia Basin is available. The scabland plant series, such as the stiff sagebrush (*Artemisia rigida* (Mutt.) Gray)/ Sandberg's bluegrass, Sandberg's bluegrass/oatgrass (*Danthonia unispicata*), *Eriogonum douglasii*/ Sandberg's bluegrass, and *E. strictum*/Sandberg's bluegrass plant associations have such low biomass productivity (125-335 kg/ha, Johnson and Simon 1987) that they will not carry fire and probably rarely burned. The green fescue (*Festuca viridula*) (1000-1100 kg/ha), Idaho fescue (*Festuca idahoensis*) (400-1350 kg/ha), and bluebunch wheatgrass (425-950 kg/ha) plant series (Johnson and Simon 1987) have enough fuel to burn annually but probably did not because of low flammability early in the season and lack of fire starts across grasslands late in the season. Native Americans were probably an important ignition source because they were observed in the 1830s setting fire to the prairies of the Blue Mountains (Shinn 1980). In 1826, the Ogden party noted Native Americans setting fire to prairie within 10 m of their camp in the upper Crooked River area near present-day Paulina, Ore. (Shinn 1980). Fires must have also spread from adjacent forest, at least from those forests where fires were frequent.

Little is known about burning in eastside shrub communities. Occasional types such as the *Eriogonum* spp./ *Physaria oregana* have too little fuel to carry fire, but many of the rest burned periodically. Long fire return intervals are suggested by accounts cited in Vale (1975), in which early travelers in the Walla Walla area noted 15-cm-thick sagebrush large enough to be used as fuel. Because sagebrush is very fire sensitive, a long fire return interval is inferred from plants this large. Daubenmire (1970) claimed that no record of Native American burning existed for the sage-steppe region of eastern Washington, but Shinn's (1980) accounts suggest such ignitions were at least locally important. Some shrubfields resulted from highintensity fires on forested sites; these areas will be discussed as early seral communities of those series.

No information is available on oak woodland fire regimes east of the Cascade crest. Information in westside oak woodlands is largely from sites where Douglas-fir is climax, and in eastside oak woodlands north and south of the Columbia Gorge, where ponderosa pine and Douglas-fir are usually the climax species. Native American burning is well documented for such sites (Boyd 1986), and similar burning must have occurred in eastside oak stands. Burning every several years kept pine from encroaching into oak woodlands, allowing easy acorn gathering. Fire return intervals, though unknown, must have been short and intensities low. Extent of the fires is unknown but may have been substantial given the dry conditions and historical abundance of grassy understories.

Western juniper has expanded its range into grassland-shrubland communities over the past century. Most juniper older than 100 years are found on fuel-limited sites, such as rimrock, and most fire history data are from these rimrock sites. Historical fire return intervals in the western juniper series range from a minimum of 10 to 25 years (table 2) to more than 100 years in places where older juniper with no evidence of fire can be found. The intensity and extent of these fires depends in part on how much sagebrush, juniper, and grass is present. Grassy sites are more likely to burn completely over a wider range of conditions than sites with more shrub and tree cover because of the fuel continuity provided by grasses (Clark and others 1985). Most fires are severe enough to kill nonsprouting sagebrush and western juniper because both species are fire sensitive. The extent of historic fires is not well known, but they had the potential to cover wide areas.

Fire Effects

Grassland-Three major plant series are present in the grassland zone: green fescue, Idaho. fescue, and bluebunch wheatgrass. Most of the fire effects described below are inferred from information in Johnson and Simon (1987). In general, grassland communities exhibit increases of forbs with fire, and most shrub communities show increases of grass and forb cover after fire.

Green fescue (*F. viridula* Vasey) is dominant in the high elevation green fescue series. Hot, late-season fires will damage green fescue. Weedy forbs, such as yarrow (*Achillea millefolium* L.), aster (*Aster integrifolius* Nutt.), penstemon (*Penstemon globosus* (Piper) Pennell & Keck) and pokeweed (*Polygonum phytolaccaefolium* Meisn.) are favored by fires, which probably occurred at infrequent intervals.

The Idaho fescue series is a widely distributed plant series with more than 10 plant associations in eastside landscapes. As with the green fescue series, hot, late-season fires favor forbs in these plant associations. Late-autumn fires are often less damaging to Idaho fescue than mid-to-late summer fires. Fires tend to burn within the accumulated fine needle-like culms at the base of the plant and produce temperatures sufficient to kill some of the basal meristematic tissue. In Idaho, return to preburn cover took up to 30 years after summer fires (Harniss and Murray 1973). Fires probably created conditions that favored plant diversity on these sites. Balsamroot (*Balsamorhiza sagittata* (Pursh) Nutt.), lupines, Kentucky bluegrass (*Poa pratensis* L.), and yarrow are favored by burning, but harsh paintbrush (*Castilleja hispida* Benth.) and prickly lettuce (*Lactuca serriola* L.) are not adversely affected. Prairie junegrass (*Koeleria cristata* Pers.), a common codominant with Idaho fescue in four plant associations, appears to do well after a fire.

The bluebunch wheatgrass plant series is dominated by bluebunch wheatgrass. This grass is more tolerant of fire than Idaho fescue, and fire stimulates bluebunch wheatgrass flowering and seedset. Associates such as Sandberg's bluegrass, red threeawn (*Aristida longiseta* Steud.), and milkvetches (*Astragalus* spp.) are also favored by burning. Wyeth's buckwheat (*Eriogonum heracleoides* Nutt.), a late seral associate in the higher elevation bluebunch wheatgrass plant associations, is weakened by fire, and prickly pear (*Opuntia polyacantha* Haw.) can be damaged after fire by grazing if its prickles are burned off.

Shrublands-Fire effects depend on the adaptive strategies of individual species. Those that can sprout will grow back quickly and share dominance with grasses and forbs for a shorter period of time than those that do not sprout. Stiff sagebrush, big sagebrush (*Artemisia tridentata* Nutt.), low sagebrush (*A. arbuscula* Nutt.), and curlleaf mountain-mahogany (*Cercocarpus ledifolius* Nutt.) are nonsprouters that recolonize burned areas slowly. Bitterbrush (*Purshia tridentata* (Pursh) DC), a weak sprouter, is generally killed outright by summer or autumn fires (Clark and others 1982). All of these species increase with protection from fire, but also become decadent over time.

Other shrub species are moderate to strong sprouters and unless fires are repeated and intense, these other species will regain dominance over 5 to 10 years. Ninebark, snowberry, spiraea (*Spiraea betulifolia* Pall.), and rabbitbrushes are common shrubland dominants that increase after burning.

Oak woodlands-Oregon white oak (*Quercus garryana* Dougl. ex Hook.) is the dominant overstory tree in the presence of regular underburning; its acorns do not need stratification, which may be a limiting factor to oak outside of the maritime influence of the Columbia River corridor. Acorns may germinate in relatively warm, moist autumns, and later freeze during winter. Most acorns are eaten by small or large mammals, so few are left to germinate.

Although Oregon white oak, like most other coastal white oaks, is thought to be a less vigorous sprouter than other oak species, more recent research and observations suggest that sprouting is an important form of reproduction for this species (Kertis 1986, Sugihara and Reed 1987). Intense burning associated with log corridors seems to create favored sites for acorn establishment (Agee, unpublished data). These sites, with substantial disturbance, are also favorable sites for many alien species such as velvetgrass (*Holcus lanatus* L.), tansy ragwort (*Senecio jacobea* L.), and St. John's wort (*Hypericum perforatum* L.). The acorns may sprout and oak becomes established because the acorns are buried and thus protected from predators, because of less initial competition from herbaceous vegetation, or because fire sanitizes the site from fungi.

Because oak sites regenerate from both seedlings and sprouts, stand development patterns are not easily interpreted. Historically, seedlings may have replaced dead overstory oaks once the stems fell and burned in fires, which would probably result in a uniform distribution of oaks on the landscape because of moisture competition inhibiting oak establishment near live trees. More recently, cutting of oaks has resulted in more stump sprout and root sucker establishment, which might be more clumped because of multiple sprouts per stump and more root suckers within the circumference of the rooting platform of the older tree.

Western juniper-Western juniper plant associations contain many of the grassland or shrubland species mentioned above as codominants. In rimrock areas, western juniper has remained prevalent over past centuries because of the rocky substrate and lack of fuel needed to carry intense fires. These locations commonly contain older juniper, many of which are fire-scarred, suggesting a moderate severity fire regime. Western juniper has remained dominant in shifting sand dune ecosystems in south-central Washington, migrating with the dunes over millennia (Long and others 1979). Fire is likely less a factor in the stand dynamics of dune junipers.

Historically, fires at 10- to 25-year intervals confined western juniper to protected microsites, although it also established itself around these sites and in grasslands. Young junipers appear to establish best on “safe sites” (sensu Harper 1977): under shrubs, such as bitterbrush, or under the skeletons of dead and down junipers where they are shaded for part of the day. Such shade can result in daytime temperatures up to 3 to 4° C cooler than on nearby bare ground (Burkhardt and Tisdale 1976). The process of tree invasion into the sage-grasslands was slow; established junipers grow slowly (6-9 cm yr^{-1}) and the sapling stage may last 30 to 40 years (Eddleman 1987). Fires moving across the sage-grasslands at 10- to 25-year intervals would have eliminated western juniper from unprotected microsites.

The crown of an established juniper expands slowly over time, and as it does, herbaceous production declines from shading, litterfall, and soil moisture competition; thus, juniper may create its own fuelbreak. I have seen large trees at John Day Fossil Beds National Monument that have survived multiple fires because of the lack of surface fuels surrounding trees, but smaller trees in a matrix of sage-grassland were killed.

Fire at intervals from 10- to 25-years will stimulate bluebunch wheatgrass, Sandberg’s bluegrass, and numerous forbs; with Idaho fescue, such intervals will have neutral or negative effects. Nonsprouting shrubs would be temporarily eliminated and cover of sprouting shrubs would be temporarily reduced.

Natural Landscape Patterns

The natural landscape patterns of grasslands, shrublands, and woodlands in eastside ecosystems are complex. We know that some shrublands burned infrequently, and that shrub-steppe, true steppe (dominated by grasses), and meadow steppe (dominated by broad-leaved forbs and grasses) vegetation all occurred. Fire created seral grassland communities in the shrub steppe types, but shrubs were not part of and did not invade other parts of the steppe because of soil and climatic limitations (Daubenmire 1970). Oak woodlands likely covered similar areas as they do today, but with significantly different structure: fewer oaks and many fewer conifers. These woodlands were true savannas, with scattered mature oaks (perhaps 40 ha^{-1}) in a grass-forb dominated landscape. In contrast, juniper woodlands were absent across much of the landscape they dominate today, existing as rimrock, canyon-edge communities that did not support continuous grass cover.

THE PONDEROSA PINE SERIES

Fire History

The dry forest ponderosa pine series is widely distributed in eastside ecosystems, although ponderosa pine is more common as a seral dominant on cooler, moister sites of the Douglas-fir, white fir, or grand fir plant series. The most comprehensive fire history for the Pacific Northwest ponderosa pine series is based on data from the vicinity of Bend, Oregon (Bork 1985). Composite fire intervals (16-ha areas) ranged from 7 to 20 years at Pringle Butte and 16 to 38 years at Cabin Lake. Fire return intervals of less than 5 years on individual trees have been documented in Arizona (Dieterich 1980), but such frequent burning has not been documented in the Pacific Northwest. The intensity of these fires appears to have been low. Munger (1917) noted that because of the open nature of ponderosa pine woods of the eastern Cascades, fires were relatively easy to check with a 12-inch-wide fireline. The area covered by individual fires in ponderosa pine forests was probably large, because fuel was available on the forest floor: long-needled pine litter and extensive cured grass in the understory. Bork (1985) was not able to show that fires were extensive: most did not scar trees over more than 16-ha areas, but this lack of scarring could be due to low intensity fires occurring frequently among thick-barked trees.

Significant interactions exist between fires and other disturbance processes in the ponderosa pine series. Post-fire insect attack is common if fire severity results in substantial basal scorch or crown scorch. A risk-rating system developed for bark beetle attack (Keen 1943) and based on four age and four vigor classes appears to work well for fire-related bark beetle attack in ponderosa pine (Swezy and Agee 1991). Vigorous residual trees may expand their growth rates after fires (Weaver 1959), and young trees can survive up to 75 percent crown scorch with less than 25 percent mortality. Older, low-vigor trees may show poor survival (Swezy and Agee 1991). Fire historically reduced dwarf mistletoe infection by pruning dead branches and consuming individual tree crowns that had low-hanging witches' brooms (Harrington and Hawksworth 1990; Koonce and Roth 1980). Little decay is associated with fire scars in ponderosa pine (Morris and Mowat 1958), although belowground root scarring from burning logs has not been studied.

Fire Effects

Between 10 and 15 plant associations are in the ponderosa pine series with both grass and shrub understories. These understory species provided significant soil moisture competition for small ponderosa pine, with surface soils most affected (Riegel and others 1992). Some shading of seedlings by live or dead trees may be important for protection from heat and frost (Cochran 1970), but height growth of established seedlings was reduced by competition with adjacent mature trees (Barrett 1973). Frequent underburns before 1900 killed most of these small understory trees, which had colonized the sites during brief fire-free intervals, maintaining an open, parklike appearance in ponderosa pine forests. Mature trees were protected from the light fires by high crowns and thick bark.

Stand development of ponderosa pine forests is associated with the shade intolerance of the pine, good seed years, and frequent fire (Cooper 1960). Forest pattern is uneven-aged at the landscape scale, but even-aged at the stand or group scale (Cooper 1960). Gaps are thought to be created by the death of old, even-aged groups of trees (0.06-0.13 ha), which fall and scarify the soil after branches and boles are consumed by subsequent fires. Pines became established on these sites, and lack of pine needle fuel in the gap may have caused it to be missed by the next fire or two, allowing the small trees to develop a little more size and fire resistance before a surface fire eventually moved through and thinned the young landscape patch. The same fires would kill regeneration under mature tree canopies. This regeneration was generally smaller because of the competition from larger trees. It was also subjected to hotter fires because of the accumulated litter from those larger trees. The even-aged pattern was thus maintained within mature groups and new groups formed only in openings. New clusters of trees would be thinned by fire over time and eventually become a "yellow-belly" mature group of ponderosa pine.

Cooper's stand development hypothesis, developed in Arizona, has been slightly altered by White (1985), who found a much broader range of tree ages and more variation in clump size in other Arizona pine stands. In the eastern Cascades, West (1969) found clump sizes of about 0.25 ha in ponderosa pine forests, with regular, uniform spacing of trees within clumps. He hypothesized that elliptical groups were due to fires scarifying the soils along the axes of trees downed by west winds. Morrow (1985) found ponderosa pine cluster sizes from 0.02 to 0.35 ha near Bend, Ore. Surviving cohorts of trees in the 1980s were associated with longer fire-free intervals of the past, suggesting saplings had an increasing probability of survival if they were protected from fire for a cycle or two.

Production of other understory species in these forests is inversely related to tree density and cover. Historically, these open, parklike stands had substantial grass and forb cover (Wickman 1992). Nonsprouting shrubs such as bitterbrush that used to be much more limited in cover because of frequent underburning are now widespread. Although bitterbrush can sprout after light spring burning (Martin and Driver 1983), fire often kills it. Clark and others (1982) showed that spring burned bitterbrush will sprout, with about 15 percent surviving for 1 year, but fall-burned bitterbrush is almost totally killed. Bitterbrush will recolonize sites, often from rodent-cached seed, but fires held the shrub understory in check. Frequent light burning allowed bunchgrasses and most forbs to recover rapidly (Wright and others 1979), so herbaceous vegetation dominated the understory.

Natural Landscape Patterns

The natural landscape pattern of ponderosa pine forests was a seemingly unbroken parkland of widely spaced tree clumps and continuous herbaceous understory. Dutton (1887) romantically described the ponderosa pine forests of Arizona: "The trees are large and noble in aspect and stand widely apart ...Instead of dense thickets where we are shut in by impenetrable foliage, we can look far beyond and see the tree trunks vanishing away like an infinite colonnade. The ground is unobstructed and inviting ....from June until September there is a display of wildflowers which is quite beyond description." The stable patch dynamics of ponderosa pine forests were largely a result of frequent low-intensity fire. Disruption of this pattern occurred at small scales (less than the 0.35-ha patch size) when trees in patches became senescent or when mistletoe-infested trees torched. Of all the eastside forest vegetation types, the ponderosa pine type was the most stable in landscape pattern.

THE DOUGLAS-FIR, WHITE FIR, AND GRAND FIR SERIES

Mixed-conifer forests are transitional between the drier, lower elevation forest or woodland types, and higher elevation subalpine forest types. The white fir series in the eastern Cascades is a southern extension of the grand fir series, and in fact the two firs hybridize extensively through central and southern Oregon. The white fir series occurs from about Bend, Oregon, south to Crater Lake National Park. The species is more widely represented in the Klamath and Siskiyou mountains and the Sierra Nevada in California. The more mesic white fir series of the Siskiyou mountains is not included in this discussion. The Douglas-fir and grand fir series may locally be absent in eastside forest transects; when they both occur, the grand fir series is found on the cooler, moister sites.

Fire History

The mixed conifer forests of the Douglas-fir, white fir, and grand fir series show the most frequent fire activity of all eastside forests, although cooler, wetter sites of the grand fir series have longer fire return intervals (table 2). In this respect, they are not transitional or intermediate to the lower and higher elevation forest zones, but in fact represent the most frequent fire return interval. Frequent fires in drier plant associations of these series is likely due to higher productivity of fine dead fuels needed to carry another fire compared to the ponderosa pine series.

Fire intensities appear to have been low in the drier Douglas-fir, white fir, and grand fir plant associations where associated dominant understory species were *Carex geyeri* Boott. or *C. pensylvanica* Lam., *Calamagrostis rebescens* Buckl., or *Arctostaphylos uva-ursi* (L.) Spreng.. Longer fire return intervals and higher fire intensities have been found in Douglas-fir forests of the eastern Cascades where the understory dominants were snowberry, ninebark, and *Vaccinium* spp. (Williams and others 1990). This pattern also appears in the cooler grand fir series. In the Elkhorn Range of the Blue Mountains southwest of La Grande, Bork (unpublished data) found individual tree fire return intervals of 50 to 200 years (grand fir/*Vaccinium membranaceum*), and intervals of 66 years and 100 to 200 years (grand fir/*Vaccinium scoparium*) on cool, moist grand fir sites.

Fire Effects

Forests within these three potential climax series have more tree species than the ponderosa pine series, although ponderosa pine was the major seral dominant in many of these forests. In some eastside locations, either one or both of the ponderosa pine or Douglas-fir series is absent, so the transition to forest may be grassland-to-pine or grassland-to-Douglas-fir, or grassland-to-grand fir (Hall 1967).

The white fir series-White fir is the potential climax dominant in these forests, but under pre-1900 conditions in most eastside white fir forests, this species was at most a codominant because of the selective thinning effect of frequent fires. A clumped pattern of stands, often composed of pure species aggregations, was typical of forest structure before European settlers arrived (Bonnicksen and Stone 1981). Thomas and Agee (1986) found clumped distributions of ponderosa pine, sugar pine (*Pinus lambertiana* Dougl.), and white fir in old-growth white fir forests at Crater Lake.

Fire dynamics in white fir forests have been modeled extensively (Kercher and Axelrod 1984, van Wagtendonk 1985). Ponderosa pine is an early dominant in simulations of frequent burning and maintains its dominance through its growth rate, growth form, and thicker bark. More than 50 percent of the basal area is ponderosa or sugar pine.

Descriptions of understory species dynamics in the presence of recurring fire are much like those described for ponderosa pine forests, except that some of the understory species obviously change. The shrubs of these forests are adapted to sprouting after burning or regenerating from fire-scarified seed (Biswell 1973; Kauffman and Martin 1984, 1985, 1991). Although summer and fall burns probably consumed many seeds, such fires also cracked the coats of seeds allowing them to imbibe water and later germinate. Repeated short interval burns favor sprouting shrubs over obligate seeders, as eventually the soil seed supply is exhausted and young plants may not be able to reach sexual maturity before another fire kills them.

The Douglas-fir series-Fire effects in the drier Douglas-fir series with understory dominants like snowberries, pinegrass, and elk sedge are similar to the ponderosa pine and white fir series. Frequent low-intensity fires kept these forests open and parklike, with ponderosa pine present as a seral dominant. Although no information exists on stand pattern, occasional long fire-free intervals allowed some Douglas-fir to grow large enough to resist destruction by fire (Keane and others 1990). The predictability of the fire regime was likely a major determinant of the proportion of Douglas-fir on these sites with occasional longer fire-free intervals associated with more Douglas-fir.

In more mesic Douglas-fir plant associations, a moderate severity fire regime likely mixed low-intensity fires with fires of higher intensity. Understory fires opened larger patches in the forest that were suitable for colonization by species such as western larch. Stand dynamics in a Douglas-fir/ninebark plant association in western Montana were simulated by Keane and others (1990 FIRESUM model). At regular fire return intervals of less than 20 years (fig. 8), Douglas-fir was essentially absent from the landscape because of its low fire tolerance when small. With longer fire return intervals, Douglas-fir became a codominant with ponderosa pine and larch. Most real stands would show a mix of the fire return intervals in figure 8. Some Douglas-fir/ninebark stands of the Okanogan Highlands appear to have had occasional intense fires (Williams and others 1990), suggesting that current models may not incorporate the entire range of fire effects. In the Blue Mountains, it tends to occupy steep, canyon-slope positions that experienced stand-replacing fires (Johnson and Clausnitzer 1991). The Douglas-fir/*Vaccinium membranaceum* plant association likely had a similar fire regime.

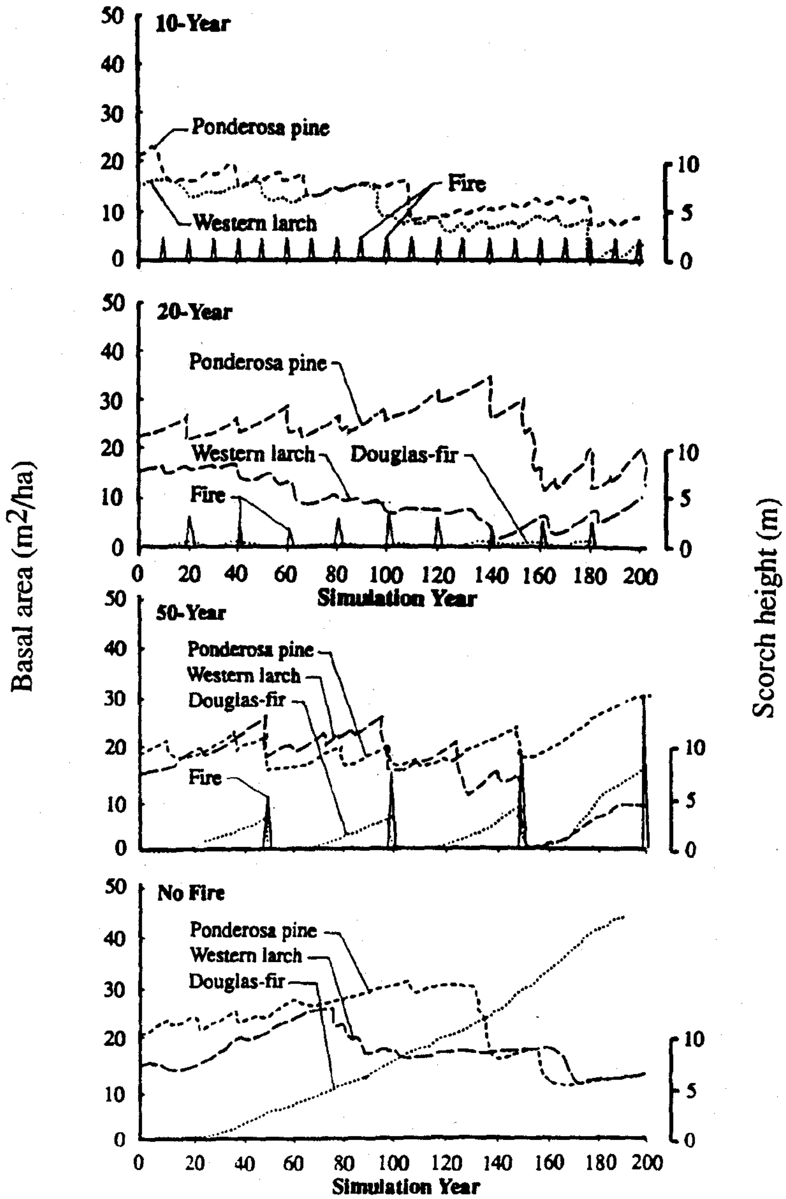


Figure 8. Simulations of relative basal area of species in a Douglas-fir/ninebark plant association under various disturbance regimes (Keane and others 19901. As fire return intervals lengthen, ponderosa pine decreases in importance relative to Douglas-fir. Where grand fir is present, its response would be similar to that shown for Douglas-fir in these figures.

The grand fir series-The grand fir series in the Blue Mountains had a wider range of historical fire regimes than the Douglas-fir series, probably because it appears to replace the Douglas-fir series in places where the latter series is locally absent. The drier grand fir plant associations, with understory dominants such as elk sedge or pinegrass appear to have burned frequently (Hall 1976, Johnson and Clausnitzer 1991). Ponderosa pine and western larch, and to some extent Douglas-fir, were historically more important than grand fir on these drier sites, and rhizomatous shrubs such as snowberry were common shrub associates. No information is available on scale-level pattern that specifically addresses these forest types, but in other mixed-conifer locations, clumping of single species (for example, one group of ponderosa pine, another of true fir) tended to occur (Bonnicksen and Stone 1981, Thomas and Agee 1986).

In the cooler grand fir series, fire return intervals were longer, and the natural fire regimes shifted to moderate severity. Stand-replacement fires were more common. As canopy gaps became larger, early seral regeneration shifted to very shade-intolerant species with special adaptation to fire. Lodgepole pine, with serotinous cones in some varieties, and western larch, with its thick bark, light seeds, and long lifespan, became important associates of grand fir. The fast, early growth rates of these two species enabled them to become canopy dominants after fire but Douglas-fir and grand fir were relegated to subordinate canopy positions (Cobb 1988). Where intense fires occurred at intervals less than 150 years, lodgepole pine generally shared dominance with other seral species at the site (Antos and Habeck 1981, Gabriel 1976). Repeated, intense burning at long-intervals (100 to 200 years) created nearly pure lodgepole pine stands, which become difficult to assign to plant associations because of the absence of the climax species (Johnson and Clausnitzer 1991).

Lodgepole pine is favored by intense fire at intervals less than 200 years (Williams and others 1990). Either low-intensity fire or absence of fire may favor other species. If a second fire occurs within 20 years after a stand-replacement fire, a stand of lodgepole pine, Douglas-fir, and western larch will lose its pine component because the double burn kills the pine and also eliminates the lodgepole pine seed source (Cattelino and others 1979). The absence of fire for a long time favors longer lived or more shade-tolerant species. Stands without lodgepole pine may have arisen from occasional fire return intervals exceeding 200 years, when short-lived lodgepole pine may have been killed by mountain pine beetles (Haig and others 1941). In the Blue Mountains, major mountain pine beetle outbreaks occurred during 1907-12 (Burke 1990), probably in stands 80 years old that regenerated after intense forest fires'in the early 1800s.

Long fire return intervals may be typical of most cool, moist grand fir plant associations with understory dominants such as oak fern (*Gymnocarpium dryopteris* (L.) Newm.), sword fern (*Polystichum munitum* (Kaulf.) Presl.), Rocky Mountain maple (*Acer glabrum* Torr.), and Pacific yew (*Taxus brevifolia* Nutt.). Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) may be present on wetter and colder sites. Most of these locations are lower slope to riparian and may be residual stringers that do not burn as often as the associated uplands. Although Pacific yew is very sensitive to fire, recent observations of seedlings on burned slash units (Ottmar, pers. comm.) suggest it may be more resilient than earlier thought.

Two fires in rapid succession tend to favor shrubs over trees. After the second fire, shrubs will resprout but tree seeds may be lacking. Big huckleberry (*Vaccinium membranaceum* Dougl.) and grouse huckleberry (*V. scoparium* Leiberg.) are two common understory dominants that may share dominance with lodgepole pine after intense fires on moist sites. A second fire in a decade or two will kill the pine before many of the trees have reached sexual maturity, leaving only re-sprouting shrubs and light-seeded herbs on site. Several herbs occurring in mature stands of the grand fir series typically root in the duff and are significantly reduced by fire (Flinn. and Wein 1977). These include twinflower (*Linnaea borealis* L.), prince's pine (*Chimaphila umbellata* (L.) Bart), rattlesnake plaintain (*Goodyera oblongifolia* Raf.), and strawberries (*Fragaria* spp.).

Natural Landscape Patterns

The white fir, Douglas-fir, and grand fir series forests, geographically transitional between lower and higher elevation forest, also appear to have had a transitional landscape pattern. No data support these speculations, so these hypotheses are made with sketchy information and limited field observations. On drier sites, these series had a stable structure, disrupted at the small scale of the patch such as occurred in ponderosa pine forests. In all three climax series, ponderosa pine was the dominant on drier sites. From the environmental midpoint of these series to their margin with the subalpine fir series, patch size was probably larger, and more severe fire events at longer intervals occurred. Under severe fire weather, larger patches (larger than 500-1000 ha) were killed. Intermediate-to low-intensity fires separated the stand-replacing ones, breaking the megapatches into smaller patches, thinning according to species susceptibility to fire and density. A more variable spatial and temporal landscape pattern emerged over time than seen in the woodland or ponderosa pine forests.

THE LODGEPOLE PINE SERIES

Climax lodgepole pine forests are primarily topoedaphic climaxes, occurring where no other tree species has a superior competitive advantage, and all but lodgepole pine have a difficult time becoming established. The most continuous block of lodgepole pine forest is in south-central Oregon, where deep pumice deposits from the eruption of Mount Mazama (now Crater Lake) left infertile, coarse material. A limited lodgepole pine series also occurs in the Blue Mountains, where lodgepole pine is the apparent climax species, but it is restricted to frost pocket sites. Whether it experiences the fire regimes of similar but seral stands on warmer sites (Johnson and Clausnitzer 1991) or the more varied interactive disturbance of fire, insects, and disease typical of the lodgepole pine series in south-central Oregon (Gara and others 1985) is currently unknown. Other climax lodgepole pine forests are found in Colorado (Moir 1969) and the Yellowstone plateau (Despain 1983). This discussion focuses on the south-central Oregon populations where the lodgepole pine series is most widely distributed. The dynamics of other lodgepole pine forests, particularly where they are seral, are discussed in each climax series where lodgepole pine is a seral component.

Fire History

Lodgepole pine forests have a moderate-severity fire regime. Multi-aged stands are the rule rather than the exception. The fire history of lodgepole pine forests is complicated by the presence of scars not just from fire but also from mountain pine beetle strip attacks (Stuart and others 1983). The interactions among beetle activity, disease, and fire are complex, and they create mufti-cohort stands, not all of which are fire-generated. Nevertheless, Stuart (1984) documented a 60-year fire return interval on the Fremont National Forest, Agee (1981) also found a 60-year interval at Crater Lake National Park, and Chappell (1991) found a 40-year fire return interval in a California red fir (*Abies magnifica* A. Murr.) forest directly adjacent to a lodgepole pine flat. The magnitude of natural fires ranges from crown fires to “cigarette burns,” where fires slowly burn along jackstrawed log corridors composed of beetle-killed trees. The stands studied by Stuart (1984) and Gara and others (1985) had an even-aged cohort that appeared to have regenerated after a stand-replacement fire in 1840. Other fires have burned only along logs because these forests have extremely low productivity (1 m^3 ha^{-1} yr^{-1}) and therefore little continuous forest floor.

Fire Effects

In this moderate severity fire regime, a typical disturbance scenario includes selective removal of about a third of the stands every 60 years, either by insects, fire, or a combination of the two. Judging from the mountain pine beetle outbreak in eastern Oregon in the late 1970s and 1980s, the extent of insect attacks tends to remain regional, and is triggered by regional climatic patterns.

Stand development patterns intimately reflect disturbance patterns. When beetles attack a stand, they generally remove large trees with sufficiently thick phloem to support a brood of larvae. The next few years have a greater probability of high-intensity fire than without beetles because of increased fine fuel in the crown or on the forest floor. Fire potential declines until the beetle-killed trees fall, typically in a widely spaced jackstraw pattern. When partially decayed, the logs are capable of sustaining slow-moving, smoldering fires, which often burn corridors only 20 to 30 cm wide across the landscape. Burning logs scar trees (Gara and others 1986) can kill or scar tree roots beneath them. The result is additional stress on adjacent live trees, encouraging another beetle attack, or allowing disease organisms to enter the tree. In subsequent years, beetles prefer to attack trees with fire-related scars or disease (Gara and others 1984). These attacks will create more snags, which provide the vectors for subsequent fires.

Regeneration is commonly found only in areas where competition for soil moisture is low and microclimate is ameliorated, such as the north side of snags or among logs (Stuart 1984). Seedling groups and insect-killed trees are clearly associated (Stuart and others 1989).

Scarce understory vegetation occurs in openings where competition for soil moisture is less severe. Because of the linear or patchy nature of most fires, fires have little effect on understory plants. Grasses, such as western needlegrass (*Stipa occidentalis* Thurb.) sprout back the next year. Bitterbrush, found on only the more productive lodgepole pine sites, is usually killed by fire (see ponderosa pine series).

Natural Landscape Patterns

Although substantial disturbance occurred in lodgepole pine forests, fire and bark beetle interactions created fairly stable landscape patterns. Some stands were burned in crown fire events, but such fires appear to be limited in extent. More common were fires that left some residual forest structure on the landscape and did not burn at a landscape scale because of patchy fuel continuity. Mountain pine beetle outbreaks, like those of the 1980s, must have also occurred in the past. The landscape pattern was probably quite stable over time, however, with occasional (50 to 100 yr) region-wide decreases in basal area (30 to 50 percent) resulting from widespread insect attack.

THE WESTERN HEMLOCK AND WESTERN REDCEDAR SERIES

The western hemlock series is poorly represented in eastern Washington and nearly absent in eastern Oregon. It is a dominant series in the maritime-influenced areas of the northern Rocky Mountains, where the grand fir series is found on excessively drained sites, and the western redcedar series on wetter and drier sites than the grand fir series (Cooper and others 1987).

Fire History

Fire return intervals are not well known for the western hemlock and western redcedar series, although almost every site contains buried charcoal, fire-scarred western redcedar trees or other evidence of fire. A typical fire return interval for low-to-moderate severity fires is 50 to 100 years (table 2); a stand-replacement interval is 150 to 500 years. Western redcedar forests in hydric locations are less susceptible to burning than western hemlock forests (Daubenmire and Daubenmire 1968). Fire intensity is clearly variable. South of Interstate 90 in Idaho and Montana forest types become more riparian, and they typically burn at lower intensity than the adjacent slope forests. In northern Idaho and northwestern Montana, however, they have been the sites of some incredibly intense fires, most notably the 20,000-ha Sundance fire of 1967 (Anderson 1968). Other extensive fires burned in 1934, 1926, 1919, and 1889 (Cooper and others 1987). In 1910, 400,000 ha burned in these plant series in northern Idaho (Barrows 1952).

Fire Effects

The tree species found in these forests, besides western hemlock and western redcedar, include western white pine, lodgepole pine, grand fir, Engelmann spruce, western larch, and Douglas-fir. Many of the stand development patterns of these forests mimic those of the cooler part of the grand fir series, except that the more shade-tolerant western hemlock is present. Western hemlock is also more shade tolerant than western redcedar (Habeck and Mutch 1973), but evidence suggests that western redcedar can maintain itself indefinitely on the wetter sites (Cooper and others 1987).

Moderate-severity fires kill lodgepole pine, Engelmann spruce, grand fir, and western hemlock on these sites. Western larch, and Douglas-fir, ponderosa pine, western white pine, and large western redcedar will resist the fires and are left as residuals. Some western redcedar will often effectively avoid severe fire damage by growing in stream bottoms and other moist microsites where fires burn with difficulty (Arno and Davis 1980).

High-severity fires can kill all the trees on the site, and succession will start with a herb-shrub stage (Davis and others 1980); then the successional dynamics are similar to those of the cooler part of the grand fir series, except that the later successional species include western hemlock and western redcedar.

Natural Landscape Patterns

Where the western hemlock series is widespread east of the Cascade range, fire return intervals are long, and older forests predominate. When disturbance occurs, patch size can be 10,000 to 20,000 ha. Where western hemlock forest is confined to moist benches, ravines, and river valleys, fire stops or burns only patches, leaving residual forest as stringers across a landscape of drier forest that burned.

THE SUBALPINE FIR AND MOUNTAIN HEMLOCK SERIES

The subalpine forests of the eastside are predominantly of the subalpine fir series. The mountain hemlock series occurs only in scattered areas immediately east of the Cascade crest, in the northern Blue Mountains, and in northern Idaho.

Fire History

Fire return intervals tend to lengthen to exceed 100 years, and fire intensities tend to increase in subalpine fir (table 2) and mountain hemlock forests. Bork (unpublished report) noted a 40 to 50 year fire return interval in a location transitional from the grand fir to subalpine fir series. In very high, cold environments with late snowmelt, forests dominated by subalpine larch (*Larix lyallii* Parl.) rarely ever burn (Arno and Habeck 1972), and whitebark pine forests have the shortest fire return intervals of eastside subalpine forests (Arno 1986, Morgan and Bunting 1990).

Fire intensities are high in subalpine forests and substantial mortality is the usual result. In the early USGS forest survey reports, the subalpine fir series was the only eastside forest type in which standreplacing fires were the rule rather than the exception (Gannett 1902, Gorman 1899). Many of the early seral (lodgepole pine) or late seral (Engelmann spruce, subalpine fir, mountain hemlock) tree species are poorly adapted to resist fire. Western larch and mature Douglas-fir, with their thick bark, have been common survivors, (Barrett and others 1991).

No information exists on the size of these prehistoric fires. Fahnestock (1976) found that most fires during the historical period in the Pasayten Wilderness, located in north-central Washington, subalpine fir series were small-only about 15 percent exceeded 15 ha. Two of these larger fires, each burned 10,000 ha, accounting for more than 60 percent of the total area burned. Small crown fires have been observed in the spring while snow is still on the ground (Huff 1988).

Fire Effects

The subalpine fir and mountain hemlock series occupy the coolest eastside forested sites, and the subalpine fir series is, by far, more widespread. Fire tends to kill all the tree species in these forests. Western larch and lodgepole pine were common early seral dominants. The most typical replacement sequence is lodgepole pine, followed by eventual subalpine fir and Engelmann spruce. Because of the cool conditions, growth is generally slow, and replacement may take more than two centuries. Often, another fire occurs before the replacement sequence is complete, and lodgepole pine is again favored.

In the Pasayten Wilderness, Fahnestock (1976) showed lodgepole pine to be the exclusive dominant 50 years after stand-replacing wildfire. Spruce and fir colonized in a relay floristics pattern (for example, Egler 1954), becoming dominant 100 to 200 years after a fire. Some lodgepole pines will persist for up to 400 years on these sites (Fahnestock 1976).

The subalpine fir plant associations with understory dominants such as Queen's cup beadlily (*Clintonia uniflora* (Schvlt.) Kunth.), twinflower, and fool's huckleberry (*Menziesia ferruginea* Smith) tend to be the most maritime plant associations (Johnson and Clausnitzer 1991). They tend to have the longest fire return intervals, and are most likely to be sites on which seral lodgepole pine has disappeared. If lodgepole pine has largely been replaced by the time of the fire, shrubs such as the huckleberries may dominate the site for decades after the fire or may share dominance with western larch. Trees may only slowly recolonize such sites, but the huckleberries will eventually decline with increased shading. After 200 years, the only seral evidence of the previous fire will be the long-lived western larch, sharing dominance with generally younger subalpine fir and spruce. If lodgepole pine is present at the time of the fire, then it is likely to be a postfire dominant.

Little is known of fire effects in the eastside mountain hemlock series because it has such a limited distribution. Stand-replacement fires favor lodgepole pine, yet old-growth stands of mountain hemlock without much fire evidence suggest some stands have not burned for many centuries. In the Oregon Cascades, Dickman and Cook (1989) found that fire fragmented laminated root rot (*Phellinus weirii* (Murr.) Gilbertson) root disease centers in mountain hemlock by favoring *Phellinus*-resistant lodgepole pines after large stand-replacement fires.

In timberline stands, whitebark pine may be a codominant with subalpine fir. Crowns may not close on these sites if disturbances such as fire, snow avalanches, and rockfalls, prevent it. Scattered vegetation is conducive to whitebark pine survival after fire, and trees may escape with only scarring. Clark's nutcrackers are known to cache pine seeds in burned areas from trees in adjacent unburned areas (Tomback and others 1990). In stand-replacement burns, this practice allows whitebark pine to colonize sites first. Subalpine fir is slow to become established because of its large seeds which are not wind-dispersed long distances. Fire often recurs before whitebark pine has disappeared (Morgan and Bunting 1990).

Natural Landscape Patterns

Stand-replacing fires of variable size appear to be the fire pattern of the subalpine fir and mountain hemlock series. The frequency distribution of fires approaches a negative exponential distribution (many small fires, few large ones). In terms of total area, the few large fires will represent the dominant patch age in any river basin, so that the age-class distribution of fires is not represented by a negative exponential distribution. The fire interactions in Pacific Northwest subalpine forests are not equilibrium, steady-state processes, and the patterns will likely vary considerably drainage by drainage. This behavior is much less common for lower elevation forest series.

Subalpine sites are marginal for tree establishment and growth. Depending on the extent of the fire and the weather that follows it, substantial burned areas may remain treeless for decades unless a seed source of lodgepole pine is present at the time of a burn. Where lodgepole pine is present, tree cover is usually rapidly reestablished. Modeling patch sizes created by fire and regeneration that follows is more complicated and variable than modeling for lower-elevation forests.

FIRE EFFECTS IN RIPARIAN AREAS

A riparian area is in contact with or adjacent to open water. In the western States, riparian vegetation is estimated to occupy about 1 percent of the forest (Clary and McArthur 1992), and in the Blue Mountains, riparian areas are estimated to cover about 2 percent of the forest (Kauffman 1988). Though riparian areas occupy a very small proportion of the forest, they have a disproportionate importance in natural resources issues. Besides supporting aquatic organisms, riparian areas are the most productive timber and forage sites and consequently, are constantly used by wildlife.

Fire affects riparian zones both directly and indirectly. Direct effects are those associated with burning within the riparian zone. Indirect effects arise from burning outside the riparian area, which affects transport of sediment, biomass, or water through the riparian zone. Both types of effects can be characterized by knowledge of the fire regime and the type of riparian system. Riparian zones often represent the down-canyon extension of a higher elevation plant series (fig. 9). Cold air drainage at night and less insolation make these areas cooler and moister than associated slopes.

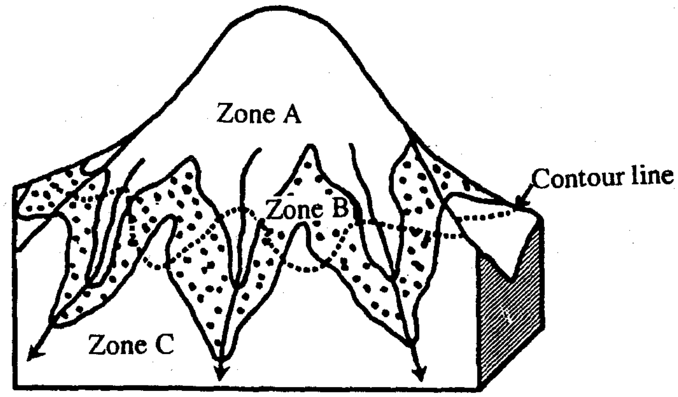


Figure 9. Interfingering of vegetation zones on a mountain slope. Because of less insolation and cool airflow at night, riparian areas tend have vegetation often found at higher elevation on slopes or ridges (from Franklin and Dyrness 1973).

The direct effects of various disturbance types on riparian systems (fig. 10) was conceptually modeled by Agee (1988). Floods have a wider effect where floodplains are better developed and streams are larger. More incised, smaller riparian areas have less wind disturbance because topography breaks up wind patterns. In general, riparian areas do not burn or they burn at reduced intensity, because they are wet sites with more deciduous vegetation and higher dead and live fuel moistures. This generalization appears to hold in coastal spruce forests (Agee and Huff 1980), Douglas-fir forests of the Oregon Cascades (Swanson 1981), spruce-fir forests of the Rocky Mountains (Romme and Knight 1981), and western redcedar forests of Montana (Habeck 1978). Determining fire return intervals is complicated by the presence of riparian tree wounds from ice flows (Filip and others 1989, Rosentreter 1992) and the short life spans of most riparian hardwood species. The riparian zones of drier areas probably did burn more frequently. In the Blue Mountains along the Oregon Trail, early travelers noted riparian groves of aspen and willow that had been recently burned (Evans 1991).

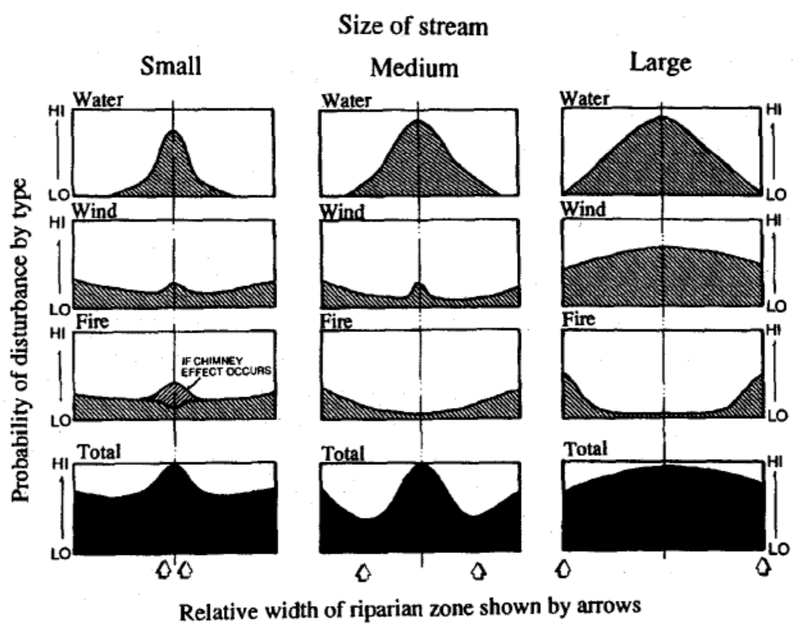


Figure 10. A model of riparian disturbance patterns. Relative probabilities of water, wind, and fire disturbances for small, medium, and large stream systems are shown. The "total disturbance" probability (at bottom, black) suggests that the lowest probabilities of disturbance are at the edges of medium stream channels and perhaps away from the floodplain of large streams (from Agee 1988).

Headwater riparian areas sometimes burn with greater intensity than surrounding slopes because of a channeling effect of wind in an area of generally higher biomass than elsewhere. Some of the hottest burn sites in the 1988 Dinkelman fire near Wenatchee were in riparian areas.

If a small fire creeps through a riparian area, it often topkills most shrubs and deciduous trees, such as willows and cottonwoods, but because most of these species re-sprout, soil stability is not impaired. Regrowth will eventually replace much of the shading effect at the stream edge. Large ponderosa pine, Douglas-fir, or western larch easily survive such fires, and western redcedar often does so by growing in wetter microsites. Engelmann spruce and subalpine fir, present at higher elevations and in cooler riparian zones, are almost always killed. A fire of similar intensity will thus have a more severe effect on tree mortality at high elevation if these conifers are a part of the riparian zone that burns.

In higher severity fires, even riparian stringers are killed. The 1970 Entiat fires left almost no riparian zone along the Entiat River, although scattered western redcedars survived along the bank. Little shading (compared to pre-burn) is present there after 20 years or more. Further upslope in the same watershed, hillslopes were covered with ponderosa pine and Douglas-fir, in the Douglas-fir series. Fire scars on stumps and stump ages of several hundred years indicate the mufti-age-cohort stand had frequent lowintensity burning. Yet the riparian area in the creek bottom directly below was an even-aged stand of lodgepole pine, which regenerated after a stand-replacement event about 1900. This example provides more evidence that riparian zones may burn less frequently but occasionally more intensely than the surrounding slopes.

EFFECTS OF EXTREME WEATHER ON EASTSIDE FOREST ECOSYSTEMS

Weather is considered part of the environment to which native vegetation has become adapted. Most discussions of disturbance (White and Pickett 1985) are vague when it comes to defining when changes in environmental parameters become disturbances. Usually, the definition includes a subjectively defined parameter-for example, “substantial.” In this discussion, I consider unusual weather events that affect growth or survival of eastside vegetation.

Unusual Temperature

Excessive temperatures are important for their absolute value. Additionally, high or low temperatures can occur at critical times, causing damage or mortality to forest vegetation.

Vegetation damage associated with unusual cold weather is often preceded by unseasonably warm temperatures, which have either prevented cold hardening of the foliage of plants or initiated meristematic development of tissue. Recorded temperature extremes have often provided improved competitive advantages for some vegetation, rather than widespread killing of all vegetation. One cold snap in early November 1955, occurred after an unseasonably warm period. Forests on the west side of the Cascades were affected more than eastside forests, with significant damage to western hemlock and western redcedar and widespread killing of red alder (*Alnus rubra* Bong.) and Pacific madrone (*Arbutus menziesii* Pursh.) (Duffield 1956). By contrast, eastside forests escaped relatively unscathed, although truck and orchard crops were significantly damaged (Torbitt 1956).

In January 1989, unseasonably warm weather was followed by extreme cold-temperatures below -30° C were recorded in interior British Columbia. Buds had probably already met their chilling requirements, and cell division had begun in meristematic tissue, with an associated loss of cold hardiness (van der Kamp and Worrall 1990). Douglas-fir, white spruce, Engelmann spruce, and subalpine fir suffered cold damage above snow level, with younger trees being hit harder than older ones. The most significant damage was recorded on western redcedar; surprisingly, little damage was recorded for western hemlock. Lodgepole pine, noted for its cold tolerance, was unaffected.

A poorly documented form of cold damage, which can occur in eastside forests that are heavily fertilized, affects development of cold hardiness in autumn. Trees in experimental plots in the Mud Creek drainage of the Entiat River, fertilized with the equivalent of 400 lbs ac^{-1} urea N, have been almost totally killed within plot boundaries. The delay in cold hardiness is an indirect effect of high N content, which causes changes in translocation of substances affecting cold hardiness. Trees die from the top down over a period of several years. In red spruce (*Picea rubens* Sarg.), increased N has been associated with increased cold tolerance in needles but decreased cold tolerance in buds (L'Hirondelle and others 1993).

Damage from excessively high temperatures on mature trees is rare, although mortality of seedlings from high daytime temperatures is commoq (Daubenmire 1943). High temperature damage in eastside forests is periodically observed during winter months. A phenomenon called "red-belt" (Bega 1978) has been observed every few years in the Entiat River drainage at about 900-m elevation. On cold, snow-covered ground, the ability of trees to absorb water in their roots and to translocate it is low. A temperature inversion associated with either cold air drainage from higher elevation in the mountains or penetration of cold Columbia Basin air, together with a Chinook or foehn warm wind, desiccates a thin band of foliage directly above the inversion layer. The dead foliage is brown in the spring, although buds are usually not affected and produce normal foliage. Such damage must be associated with loss of growth, but leader death is rarely observed. In Colorado, a red-belt effect in lodgepole pine was associated with only 1 percent tree mortality; 97 percent of trees exhibited new growth the following season, and bark beetles did not attack them (Schmid and others 1991).

Unusual Wind

Windstorms of sufficient magnitude to cause widespread blowdown are rare in eastside forests. Some of the strongest winds are from the north, but the mountainous terrain tends to reduce wind intensity. No information was found in the literature on widespread windthrow in eastside forests (Agee and Edmonds 1992). In 1955, a strong wind knocked over a ponderosa pine, killing a camper in a trailer in Yakima County (Phillips 1956). The westside suffered more treefall and powerline failure, however.

Wind-Temperature Interactions

Winter winds are associated with foliage die off on windward sides of subalpine trees, creating asymmetrical crowns or krummholz, where leader growth continually dies back, and trees are shrub-like. Winter desiccation of needles may be less important than abrasion of cuticular wax by wind-blown particles, which reduces the needle's ability to retard moisture loss. When the relative needle moisture declines below 60 percent (MPa moisture stress), needle death is likely (Hadley and Smith 1986). As with most other weather disturbances, this effect is more important for its influence on the trees ability to compete with other vegetation than as a cause of death. Desiccation surely causes some seedling mortality on windblown subalpine ridges, however.

Unusual Moisture Status

Unusual moisture may take the form of above- or below-normal amounts of precipitation, as rainfall or snow. The regional drought of 1920-40 in the Pacific Northwest created substantial insect infestation problems, particularly for pines, and Keen (1937) investigated tree-ring records of eastside forests to attempt to relate the magnitude of this drought to past droughts. He found evidence of drought back to the 1200s, with more recent droughts in 1917-36, 1870-93, 1839-53, 1795-99, 1777-88, 1756-60, and 1739-44. Grauxnlich (1987) showed that the timing of regional droughts differs by subregions in the Pacific Northwest: the Columbia basin is not always synchronous with the western lowlands of the west Cascades or the southern valleys of southwest Oregon and northern California. Synchronous droughts over the last three centuries occurred in 1973, 1929, 1899, 1839, 1739, 1721, and 1717. Evidence suggests that droughts and forest fires occur together, although the predictive ability is not strong.

Rain-on-snow events melt the snow and often cause major flooding. Riparian areas can be detrimentally affected by bank erosion and streamside sloughing. Exceptional snows can cause damage to tree crowns, either by breaking branches of mature trees or deforming leader growth of young trees (Williams 1966). Generally, however, excessive moisture has a positive effect on forest growth because of the amelioration of a regionally limiting environmental factor-lack of water.

An interaction between stand structure and snow falls can lead to increased damage to the “doghair” thickets (very high densities of spindly small trees) of trees that now occur in many eastside forests. Many of these doghair stands are a result of fire suppression and removal of larger canopy dominants through selective harvest. Because of excessive density, stems continue to grow in height but not diameter. After a heavy snowfall, or if the stand is open on one side, these doghair trees will break off because they cannot sustain the additional weight on the crown. This action begins at ratios of crown height to stem diameter of 100:1 (for example, 100-inch tall tree at 1-inch diameter; Oliver-and Larson 1990), but breakage from additional snow can occur at ratios below 100:1. This problem will only exacerbate with time because this kind of stand stagnation rarely corrects itself.

EFFECTS OF FOREST MANAGEMENT ON FIRE REGIMES

Management practices that have had the most significant effects on fire regimes in eastside forests are fire suppression, livestock grazing, and selective tree harvesting. Other activities, such as pest suppression and wilderness fire management, have had a much less noticeable effect. They are covered in a paper by Oliver and others (1993).

Fire Exclusion Policy

Near the beginning of the 20th century, the European tradition of forestry began to be seen as a model for American forestry. In this context, fire was considered a major threat; consequently, new legislation was enacted to protect forests (for example, the Weeks Act of 1911). Additionally, fires from land clearing and other activities had threatened lives and property as well as timber reserves, and those who argued for fire suppression pointed to these problems in making their case. Ironically, as fire was being institutionalized on the west side of the Cascades to reduce slash after logging, it was being eliminated as a management tool in the eastern Cascades where its historical effect had been much more frequent.

The battle over whether fire should be used as a management practice was staged in the pine forests of northern California between 1910 and 1925. The Southern Pacific and Red River timber companies were using low-intensity, prescribed fire to manage their pine timberlands, but this practice was decried as “Piute [sic] forestry,” a tactic meant at the time to imply poor management. The issue was discussed in Sunset Magazine in 1920 (Graves 1920, White 1920). The Forest Service argued that fires killed regeneration under the larger trees, which led to the end of light burning by the end of the 1920s. Few people foresaw that successful regeneration could eventually lead to increased wildfire control and forest health problems.

Two courageous professionals attempted to show the long-term detrimental consequences of such a fire policy beginning in the 1940s: Harold Weaver of the Bureau of Indian Affairs in the eastern Cascades of Oregon and Washington (Weaver 1943), and Harold Biswell of the University of California in the pine forests of that State. Both encountered tremendous resistance within their profession, but near the end of their careers in the 1970s, both saw a change in attitude within the profession (Biswell 1989). Changes in attitudes about fire have not resulted in its being put to use, however. Most use of fire into the 1990s has been associated with activity fuels on harvest units over a small proportion of the landscape (for example, Kilgore and Curtis 1987).

Timber Harvesting

Timber harvesting has historically focused on the more important commercial species, ponderosa pine and western larch. Early harvesting activities concentrated on only the largest trees because merchantability standards did not allow smaller stems to be efficiently processed. Lower elevation stands in the ponderosa pine, Douglas-fir, white fir, and grand fir series were selectively logged for pine and larch. In general, these early seral species were removed, usually leaving smaller climax species to capture the growing space.

In more recent decades, a wider suite of species has been used, including grand fir, lodgepole pine, and subalpine fir. Smaller diameter material has increasingly been harvested for processing into wood chips.

Livestock Grazing Settlement of the Oregon Territory and associated livestock grazing began in the late 1830s. By 1860, 200,000 cattle were settled in Oregon, along with sheep and wild horses (Galbraith and Anderson 1991).. By the winter of 1861-62, the practice of yearlong open range with no shelter or hay storage was being questioned. In the Walla Walla Valley, dead cattle were so numerous at the end of that winter that a person could almost step from one dead animal to another throughout the whole valley (Galbraith and Anderson 1991). Harsh winter weather did not recur for 20 years, so yearlong open-range grazing continued until 1889-90 when another bad winter forced cattJemen to accept the need for food and shelter in winter.

Sheep were increasing on eastern Oregon rangelands in the 1890s, in part because they were cheaper to raise than cattle. Bands of sheep wandering on already overgrazed ranges led to range wars in the early 1900s. The Sheep Shooter's Committee of Crook County claimed to have shot 8,000 to 10,000 sheep a year during that time (Galbraith and Anderson 1991). Grazing laws were developed for National Forests by 1910, but much of the damage had been done (Harris 1991). The rangelands had evolved without substantial grazing pressure, but intensive grazing significantly damaged the perennial bunchgrass ranges.

Effects of Past Management Practices on the Landscape

Because the natural fire regimes of eastside forests were interrupted by the combined effects of these management practices, ecosystems changed-some experienced only minor changes, but in others, the changes were catastrophic and probably irreversible.

Grasslands, shrublands, and woodland ecosystems-The perennial grasslands surrounding the Blue Mountains have been subjected to severe overgrazing; alien species, in particular the annual cheatgrass (*Bromus tectorum* L.), have increased as a result. Cheatgrass germinates in the autumn in this region and maintains a rosette form as it develops a root system during the winter (Young and others 1987). It can use much of the available soil moisture before perennial grasses initiate new growth in the spring. Cheatgrass completes its life cycle in late spring to early summer, and the fine-textured cured foliage is highly flammable. This trait has expanded the historical burning season in some areas, which has only increased the dominance in these sites by annuals (Whisenant 1990). In contrast, intensive grazing elsewhere has removed perennial grass fuels and reduced the ability of fire to spread.

Over the past century, with effective fire exclusion, eastside oak woodland has been invaded by ponderosa pine. Because oaks are more shade-intolerant than pine, they will be killed as the pines grow above them. This pattern is common with Douglas-fir and oaks in westside Cascade valleys (Habeck 1961, Thilenius 1968). Oak woodlands have significantly declined across the Pacific Northwest. This vegetation type might expand significantly with global warming if the gene pool survives, but significant conifer invasion has already occurred. Some oak woodlands will become extinct over the next 25 years unless conifers are selectively removed and light burning is used to control conifer re-invasion.

Most oak woodlands were associated with perennial grassland understories, which were well-adapted to periodic light underburns after the current-year foliage had cured and roots had stored carbohydrates. Oak woodlands have always been favored livestock grazing areas, because both forage from grasses and shade from trees are present. Grazing of perennials, which have not had significant ungulate grazing pressure since early in the Holocene (Daubenmire 1970), together with the introduction of alien grasses such as cheatgrass have irrevocably altered the understories of oak woodlands. Increases in tree cover have favored shrubs and forbs over grasses. Restoration efforts, by tree removal or fire, will inevitably favor alien species, although cessation of grazing in westside oak woodlands has been associated with recovery of some perennial natives (Saenz and Sawyer 1986). All of the northern Oregon white oak woodlands are threatened unless action is implemented in the next decade. At best, only partial recovery can be expected.

Western juniper has doubled its range since 1860 (Burkhardt and Tisdale 1976). Several hypotheses attempt to explain this expansion: climatic change favoring juniper; recovery from logging of juniper during European settlement; overgrazing by domestic livestock, opening up competition-free microsites for juniper; and the .absence of fire, which would otherwise kill fire-sensitive western juniper. Expansion has been slower on big sagebrush sites than on low sagebrush sites (Young and Evans 1981).

The climate-change hypothesis has shown that juniper pollen (and presumably juniper abundance) in eastern Oregon has varied over the past 10 millennia as much as in recent times (Mehringer and Wigand 1987), but the hypothesis provides no climatic evidence for the recent expansion. The logging recovery hypothesis is locally valid in mining districts (Hattori and Thompson 1987) but not regionally important, many areas invaded by junipers show no stumps or other evidence suggesting junipers existed there in the recent past. The overgrazing hypothesis is tenable, but juniper invasion has occurred even in ungrazed areas (Quinsey 1984). Overgrazing might interact with the absence of fire because herbaceous fuel decline would restrict fire spread.

The most plausible single hypothesis is fire exclusion. Historical fires killed junipers by basal or crown scorching. Intense summer wildfires kill nearly all junipers; individual fires under more moderate weather conditions can leave more residuals (Martin 1978). Prolonged herb and shrub stages have been observed on burned western juniper sites (Everett 1987). A return to natural fire regimes would reduce but surely not eliminate juniper across the eastside woodlands. It also would have positive hydrologic effects for this dry landscape (Eddleman and Miller 1992).

The ponderosa pine series-Forests of the ponderosa pine series changed in several significant ways over the last century. The landscape development pattern of clumped groups of even-aged trees was interrupted .by fire protection (Morrow 1985), allowing regeneration to survive not just in openings but under mature clumps. A widespread, fire-protected age class of ponderosa pine has colonized the landscape, creating doghair thickets of pine trees in many areas, with many trees no more than several centimeters in diameter after 60 to 80 years. This dense understory has created additional dry-season moisture stress on the older trees. Where the older trees have been removed, the younger residual stands are too dense and have stagnated, making them susceptible to attack by western pine beetle (Dendroctonus brevicomis LeConte) and mountain pine beetle (*D. ponderosae* Hopkins) (Gast and others 1991). An outbreak of beetles could increase fuel loading and thus fire hazard. Where these younger stands have been mechanically or manually thinned, the risk of beetle outbreaks is reduced (Sartwell and Stevens 1975), although pine engraver beetles (Ips pini (Say)), which breed in the thinning slash, may cause additional mortality if adequate slash is unavailable when the beetles emerge.

Where once-frequent surface fires were carried through pine stands by needle litter and grass, they are now carried by needle and branch fuels. The vertical continuity of fuelbeds has also increased over time which allows surface fires to develop into understory or crown fires under moderate weather conditions. At the same time that average fire intensity is increasing because of fuel buildup, average fire tolerance of stands has dramatically decreased because of overstocking and stagnation.

The increase in tree density, together with intensive grazing, has caused a decline in shrub and herbaceous understory. Understory production in ponderosa pine forests is inversely related to tree crown cover (ease 1958) or other measures of tree competition such as basal area, litter depth, or tree density. With dense tree canopies, forbs are favored over grasses (McConnell and Smith 1970). Once-common grasses such as Idaho fescue, bluebunch wheatgrass, and Sandberg’s bluegrass have declined, also in part because of heavy grazing. In Montana, Idaho fescue was absent in grazed stands (Evanko and Peterson 1955). Average herbaceous production in open, mature stands has probably declined from 1000 to 1500 kg/ha to 100 kg/ha or less (Biswell 1973).

Increased needle litter can replace grass as a fine fuel to carry fire. However, thick tree understories are associated with decreased average windspeed, higher relative humidity, and higher dead needle moisture content in these altered stands, so that fire spreads under fewer weather conditions. On balance, an interaction-in a statistical sense-occurs between fuels and the weather: under moist conditions, fire will not carry through stands it once burned freely across; under dry conditions, it burns much more intensely than in the past. A low-severity fire regime has been converted to a fire regime with a moderate-to-high severity.

The white fir, Douglas-fir and grand fir series-Some of the most visible landscape changes have occurred in these three plant series, particularly on the drier sites. The structural changes noted for the ponderosa pine series have also occurred in these series, but they have been accompanied by a major shift to more shade-tolerant species. These mixed-conifer forests are experiencing the most severe forest health problems (Gast and others 1991).

The duration and intensity of outbreaks of the western spruce budworm (*Christoneura occiilentalis* Freeman) appear to have increased with this shift in species composition to budworm-sensitive species (see Lehmkuhl and others 1993, Hessburg and others 1993). In Montana, stands in the Douglas-fir series once dominated by ponderosa pine are now dominated by Douglas-fir. As fire return intervals have lengthened during this century, budworm outbreak duration has increased from 8 to 13 years to 17 to 29 years, and outbreak severity (on a relative scale of 0-1) from 0.41-0.53 to 0.63-0.70 (Anderson and others 1987). Similarly, Douglas-fir tussock moth (*Orgyia pseudotsugata* (McDunnough)) outbreaks have been most common at the low elevation border of the Douglas-fir and grand fir series (Williams and others 1980), or on ridgetops or south-facing slopes (Mason and Wickman 1988), where, in the past, frequent fires kept Douglas-fir subordinate to ponderosa pine. The first outbreaks known to the Blue Mountains were noted in 1928 (Gast and others 1991), although the tussock has likely been present at low populations for long periods of time. Favoring nonhost species such as ponderosa pine is the most effective control measure for both defoliators (Mason and Wickman 1988).

In most dry mixed-conifer forests, effective fire suppression resulted in filling all of the growing space with trees by about 1960 (McNeil and Zobel 1980), unless larger trees were subsequently harvested. Many trees more than 30 to 40 years old are less than 1 m tall (Agee 1983), and trees in these multilayered forests with a suppressed understory are highly susceptible to budworm attack. This increase in tree density has also had effects on stand pattern and understory production.

The architecture of mixed-conifer stands has changed both horizontally and vertically. The spatial pattern of a mosaic of several species, with each containing a single clump species, has been replaced by the density of a single (Douglas-fir, white fir, or grand fir) shade-tolerant species (fig. 11) (Agee and Edmonds 1992, Thomas and Agee 1986). Where little harvesting activity has occurred, the past pattern of vegetation is sometimes still discernible in the older trees, which suggests that the old pattern may be recoverable with intensive thinning. Where the early seral species such as ponderosa pine or western larch have been removed, thinning will not help to restore a past pattern. Large openings will be required to encourage natural or artificial regeneration of these species (hutch and others 1993). Vertical continuity has increased in these stands similar to that seen in the ponderosa pine series, and most of the added layers are composed of shade-tolerant species. Site-specific evidence of these changes has been documented by Schellhaas and others (pers. comm.). In mixed-conifer locations in the eastern Washington Cascades, they found high tree density, with 70 percent of the trees in cohorts from the fire-exclusion period. Of about 1400 trees ha^{-1}, only 85 ha^{-1} are above 40 cm diameter.

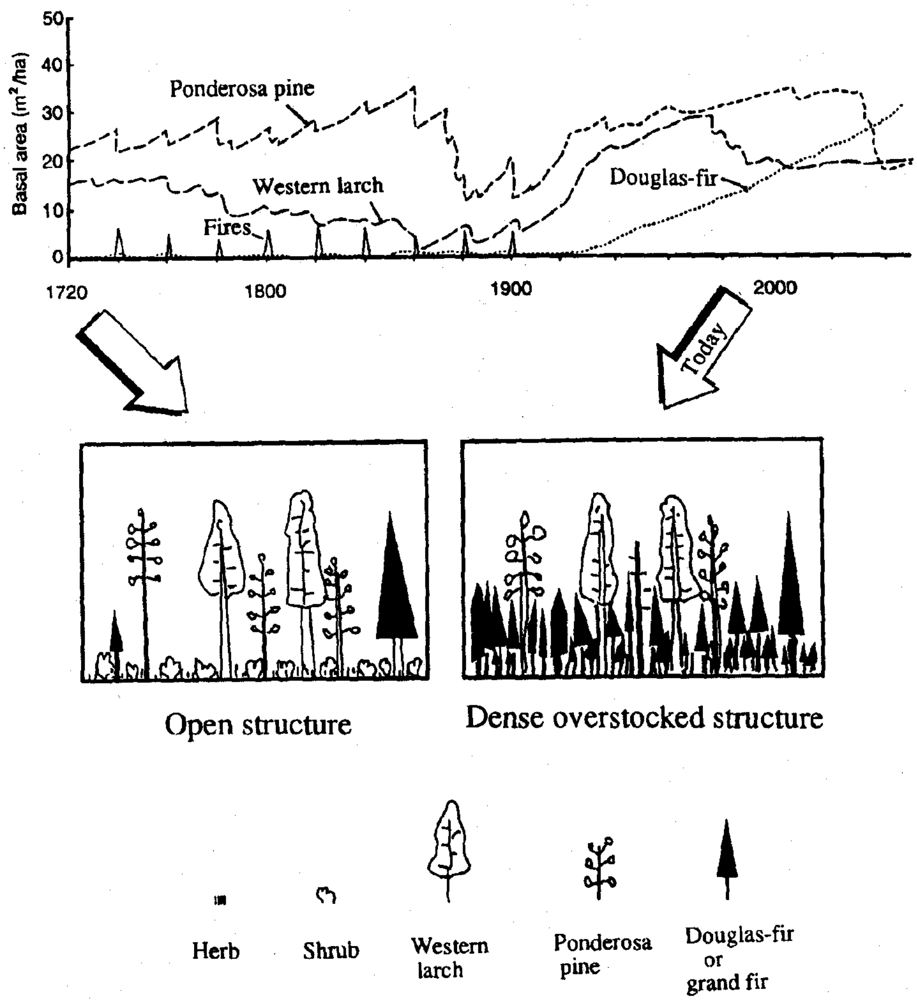


Figure 11. A simulation of ecological change in Eastern Cascade mid-elevation forests resulting from successful fire exclusion. This graph was constructed by combining from figure 8 the results of the 20year fire return interval up to the year 1900, and then the no-fire simulation after 1900. Shade-tolerant trees now choke the understory. Little precedent for current forest structure is apparent (from Agee and Edmonds 1992).

A human-induced shift from low-severity fire towards moderate-to-high severity fire has occurred in the drier portions of the Douglas-fir and grand fir series. An example of increased fire intensity is the Dinkelinan fire near Wenatchee in 1988. Trees that had many fire scars, indicating survival of many past forest fires, were killed in a severe fire. One dead tree had not been scarred by fire since 1889, with about a century of fuel buildup to accentuate fire intensity. Before 1889, the length of fire-free intervals was 19, 10, 10, 20, and 13 years, which would have maintained understories free of tree regeneration. All young trees that had come in since 1889 were also killed. A related example is the Dooley Mountain fire along Highway 245 south of Baker City, Ore. that killed most trees. The large size of ponderosa pine and Douglas-fir trees across the burned area suggests that the trees survived many fires in past centuries. Fire intensity has increased on both of these sites, apparently surpassing the past fire-intensity range, because of fuel buildup and “ladder” fuels enabling surface fires to move into the canopy.

As in the ponderosa pine series, increases in tree density together with grazing have contributed to the decline in understory production. Frequent fires consumed forage to the ground, but rhizomatous pinegrass and elk sedge were able to recover quickly. These two species were preferred by cattle and declined with overgrazing (Hall 1975). In Idaho, pinegrass production in grazed stands was only 28 percent of that in ungrazed stands (Zimmerman and Neuenschwander 1984). Conifer regeneration has been encouraged by the decline in competition from rhizomatous species and disruption of the ground by hooves.

In cool, moist plant associations of the grand fir series, with moderate severity fire regimes, the proportion of low severity fire has declined on areas burned over the last century. Small, low-intensity fires have been effectively controlled. The only fires capable of affecting a landscape are those burning under severe weather where fire suppression efforts have failed. Historically, fire created a complex mosaic of underburns where little regeneration was initiated, low-thinned stands (that is, thinning from below with smallest stems being killed) with large residuals where both shade-tolerant and shade-intolerant species could establish, and stand-replacement patches where shade-intolerant species were best adapted. With each type of fire, seral species were favored and dominance of shade-tolerant species was checked. Highseverity fires now dominate in this portion of the grand fir series. This shift is from a complex, moderate fire-severity regime to one of high fire-severity.

The lodgepole pine series-This ecosystem is one of the few in which modern human management activities appear to have neither detectable nor actual effects. Overharvest of trees has not generally been a problem, because much of this terrain does not have commercial forest potential. Grazing has not been a problem because not enough forage is produced to make the sites grazable. Fuels have not built up to any great deal, again because of low productivity.

The western hemlock and western redcedar series-Significant effects on these two series from management activities are difficult to detect, although they may be present. In northern Idaho, where these series are widely distributed, the typical fire return interval of 50 to 100 years or more is close to the effective fire-exclusion time. Unlike drier forest types that burned frequently in the past, many of these sites would not have burned even in the absence of fire suppression activities.

A different pattern may exist to the south of Interstate 90, where this type is more of a riparian or valley bottom type. Adjacent forest types, which burn frequently, are more likely to have fires of higher intensity because of fuel buildup, which are likely to move through the riparian forests because of the additional potential for energy release. Although no documentation exists, I suspect a higher proportion of fires in these riparian sites are expected to be more severe than if slope forests had been allowed to burn more frequently.

The subalpine fir and mountain hemlock series-During the past century, high-elevation forest types have experienced the least significant changes of any eastside forest type. A fire-exclusion policy has been in effect for almost a century, but this hasn't dramatically altered these forest types because their naturally long fire return intervals produces little noticeable change in these ecosystems at the stand scale. At the landscape scale, the absence of fire has probably resulted in a slight shift towards later seral communities and away from earlier seral communities. In the Eagle Cap Wilderness of the Blue Mountains, Cole (1981) suggested that valley bottom and lower slope plant associations had the most pronounced floristic response to fire suppression and had more subalpine fir in the understory than the overstory. Fires have not been erased from the landscape in these plant series, however, as shown by the upper portion of the 1960 Anthony Lakes burn on the Wallowa-Whitman National Forest, the White Mountain Complex on the Okanogan National Forest, and other subalpine locations. Sheep grazing in summer occurred in middle to high elevation meadows and ridgelines were favored as driveways to herd the sheep to summer range. Ridgelines may show, therefore, the most significant effects of management activities in forested subalpine zone.

Whitebark pine increases in importance in subalpine areas east of the Cascades crest, occupying 10-15 percent of the landscape. Whitebark pine is a seral species on about half of this area (Arno 1986), and subalpine fir is increasing in importance as the pine is killed by beetles, white pine blister rust (*Cronartium ribicola* Fisch.) and shading from the colonizing fir (Morgan and Bunting 1990). Increased use of fire may not help whitebark pine, even though it may kill back the subalpine fir. Young pines are more susceptible to blister rust infection on the bole, which kills the tree.

Riparian areas-Some riparian areas are surrounded by more continuous and greater accumulations of fuels than in the past. Some ecological changes may be defined in terms of increased risk of wildfire rather than an observed change. For example, the relict population of Alaska-cedar in the Cedar Grove Botanical Area on the Malheur National Forest is a very small riparian grove surrounded by a somewhat drier forest type. The grove has historically been a riparian stringer largely unaffected by the fires burning to its margins, although it has survived one fire this century. With fuel buildups in the bordering forests, the next fire regime may be of higher severity. The grove is now at risk from stand-replacement fires that are likely to occur in neighboring forests. A high-intensity fire could eliminate the cedar from this site and eliminate the only glacial relict population of Alaska-cedar in the Blue Mountains.

In inland Northwest areas, seral riparian stands included large western larch that survived several light burns but are now slowly dying because of competitive stress. Eventually, these 300 to 600 year-old-trees will be eliminated from riparian zones as a result of fire exclusion.

In unburned watersheds, hydrologic effects associated with management changes over the last century include more water usage by denser vegetation cover and in burned watersheds, hydrologic effects include greater tree mortality in burned areas and changes in amount and timing of water flow and sediment yields. Increases in juniper cover from overgrazing and fire exclusion, for example, have been associated with 50 percent reductions in winter soil-water recharge in sage-steppe ecosystems (Eddieman and Miller 1992).

The effects of fire depend on how much of the watershed burns. If fires are small, effects on downstream riparian zones are likely undetectable. If fires are large, peak flows may increase (Helvey and others 1976), summer discharge may be higher, and probabilities of sedimentation, storage, and movement are higher (McNabb and Swanson 1990). Direct effects of fire tend to decrease in effect from edge to the center of the riparian zone. By contrast, indirect effects will usually be highest in the center of the riparian zone and decrease outward.

Subalpine and alpine meadows-Tree invasion of high mountain meadows has sometimes been linked to fire protection efforts in these areas (Ratliff 1985). Such an invasion, however, may also occur under natural conditions, so identifying causes is difficult. In the Pacific Northwest, three distinct natural patterns of meadow invasion have been documented. The first is a drought-induced tree invasion into snow-dominated meadows, where the shrub dominants are generally heathers (*Phyllodoce* spp.). During the 1920-40 regional drought, substantial invasion occurred in these meadows (Franklin and others 1971). The second pattern is related to fire-created meadows that are being recolonized by trees (Agee and Smith 1984, Henderson 1973). These sites are usually steep, south-facing slopes, and colonization lags substantially (30 to 50 years), with most regeneration occurring during wetter than normal summers. Little (1992) found that, on burned sites, extended growing seasons created by early snowmelt in spring and wetterthan-normal summers is associated with tree establishment. In the east Cascades, both of these patterns may exist, and they may be accompanied by yet a third pattern.

The third pattern of tree invasion is associated with wet meadows that have been overgrazed. During the grazing, tree establishment may have been minimal while herbaceous species composition shifted. For example, while pristine subalpine meadows in the Eagle Cap Wilderness of the Blue Mountains have an average cover of 40 to 75 percent tufted hairgrass and Holm's Rocky Mountain sedge (*Carex scopulorum* Holm), grazed meadows have a 10 to 12 percent cover in these species (Cole 1981). Fringeleaf cinquefoil (*Potentilla flabellifolia* Hook.) is a common dominant forb in the grazed meadows. In the Sierra Nevada, some wet meadows experienced a slow recovery of herbaceous vegetation that resulted in gully formation which in turn precipitated a drop in the water table. Better drainage encouraged invasion of lodgepole pine into the drier meadow edges. Under drought conditions, these meadows have burned, killing the invading pines (DeBenedetti and Parsons 1979), but the invasion continues because it is largely independent of fire. In eastside subalpine sites, lodgepole pine and Engelmann spruce are the most likely invading tree species, but this invasion has not been identified as a major problem in Pacific Northwest wet meadows.

MANAGEMENT ISSUES FOR THE 1990s

The Management Challenge

Ecological changes in eastside forests associated with fire exclusion and other management activities have expanded fire hazard and led to declining forest health-problems almost everyone views as undesirable. The solutions require a better linkage of biological and sociopolitical systems than in the past. Numerous popular articles recognize fire must be returned to the system, which is easier said than done. The past role of fire in eastside ecosystems is not a model for management: fire is a tool, not the rule. Eastside ecosystems are more complex in almost every respect than they were a century ago, with unprecedented changes in tree species composition and structure, fuel buildups, alien species, and insect and pathogen dynamics. Restoring fire to an altered ecosystem is fraught with uncertainty. Will old trees survive restoration fires with massive fuel buildups at their bases? Will alien species, so adept at colonizing disturbed ground, become even more dominant?

Restoration fires would have to burn in a land crisscrossed with square and rectangular ownership boundaries. Few natural boundaries for fire can be so neatly scribed on the landscape. Fires will release smoke from biomass that has been accumulating for decades. Smoke that once drifted harmlessly across the valleys of the Columbia Basin would drift into local smoke-sensitive communities. Applying ecosystem management principles, including those that recognize people as part of the ecosystems, is essential to achieving workable solutions (Agee and Johnson 1988). Adaptive management, including strong monitoring and feedback, is also essential if successful treatments are to be expanded and failed treatments abandoned (Walters 1986).

Several institutional issues clearly emerge as important to management. The first is scale. If fire is to be addressed at the proper landscape scale, drip torches will not do the job-aerial ignition will be necessary. The technology is here, but the local skill at using it needs to be developed. Cooperation not only with other local institutions but also with international institutions, such as Australian fire managers who regularly use aerial ignitions, will be necessary. Expanded networks of fire weather stations critical to developing site-specific fire prescriptions are cheap insurance against expensive and avoidable fire escapes.

Local and regional statutes dealing with land zoning and fire protection will need review-all landowners are part of the solution. The urban-wildland interface exists in the Blue Mountains, Bend, Spokane, and Wenatchee-everywhere people live.

Fire management is a more complex job than fire prevention and suppression. Smokey Bear can't say it all in one sentence anymore. Posters on fire ecology should become as widely circulated as posters on fire prevention. Smokey could be positively involved in a program to change the public's awareness of fire for eastside forests. Smokey should still be primarily a symbol of fire prevention, but his role has to be more actively integrated with interpreting the use of fire. Otherwise, fire management in eastside ecosystems may not succeed.

The research challenge outlined below assumes that fire is a part of the solution to the eastside forest health crisis. Yet much is still unknown about where and when fire should be used, how often, and for what ends. The research challenge is to periodically redefine the questions and refocus research accordingly.

The Research Challenge

Efforts to create new information through basic and applied research should continue and that information should be transferred to practitioners through education and training programs.

Education and Training-Workshops should be organized soon for three audiences: the interested public, forest managers/resource specialists, and fire managers. Each has a unique need for fire information. The interested public is likely to be most responsive to television, brief lecture series, or one-day symposia.

Topics might include the forest health situation, the natural role of fire, or how fire can be integrated with community goals.

The second group might include members of interdisciplinary planning teams and decision-makers in charge of selecting management alternatives. These professionals will be evaluating the effects of actions on combined resources and need to have specific information-for example, what fire history information is and how it is collected, techniques of managing fuel consumption, effects of fire on soils and wildlife, and so on. The more details, the better for this audience.

The third audience is the fire manager, the people who will be on the ground applying fire. This group will need even more specifics on fire planning and operations-for example, ignition strategies, fire applications of geographic information systems, biomass consumption prescriptions, and fire weather forecasting, fire danger rating.

Waiting to begin until more research is done is not necessary. Training would improve fire planning efforts and community acceptance and trust before the first research project is completed. Continued technology transfer is certainly recommended, but initial efforts could and should begin soon.

Fire Research-There are a number of topical research areas of high priority where increased information will be essential to adequately plan fire management strategies: Natural fire regimes, efforts of global change, vegetation and soil effects, wildlife habitat, visual quality, and effects on cultural resources. More research is needed on variation in regimes and landscape patterns caused by fire. In particular, variation in fire return intervals within a forest series by province or ecoregion is not well documented, and the extent of past fires is largely unknown. Landscape analyses of fire regimes from sagebrush-steppe to alpine zones would be helpful in determining patterns of common scale and severity associated with past fires and the degree to which vegetation-type boundaries acted as fire boundaries. Research projects in oak and juniper woodlands, ponderosa pine, mixed-conifer, and subalpine fir forests, if geographically integrated, could provide a broad understanding of subregional fire histories and yield important information that would have great value in designing restorative fire treatments for landscapes.

Global change scenarios for the Pacific Northwest suggest major shifts in vegetation types. It has been proposed that the shifts will be triggered by landscape disturbances such as timber harvesting or fire (Franklin and others 1991). Climate can be linked to forest fire history and may suggest likely future scenarios with climate change. Some dendroclimatic reconstructions exist for the region (Keen 1937, Graumlich 1987). These reconstructions can be integrated with new information including global change scenarios, to associate climate more closely with what research can tell us about the extent and intensity of past fire.

All ecosystem components will be affected by fire. The focus of research described below is on ecosystem components likely to be significant constraints or opportunities in management plans.

Biomass estimates of major vegetation types are necessary to accurately predict consumption from fire. In particular, relative proportions of live to dead biomass will differ by forest type and by disease and insect hazard category (see Hessburg, appendix A, 1993). Effects of fire on plants is critical if fire prescriptions are to predict the effect of fire on plants based on fireline intensity, season, and ecosystem condition. Examples include effect of intensity and season on bunchgrasses, on aliens such as cheatgrass and knapweeds (*Centaurea* spp.), on juniper (whether or not the juniper is felled before burning), and on ponderosa pines that have substantial fuel buildup at their bases.

Few studies outside of the southeastern United States have looked at the effects of repeated burns. Current studies (for example, Swezy and Agee 1991) measure the effect of only the first restoration fire, which is likely a maximum effect that would not be repeated in subsequent burns. Consequently, more studies are needed on the effects of repeated burns on all vegetation types.

Research in physical, chemical, and biological soil effects will be necessary as restorative fire treatments are proposed. Landscape erosion studies, that tie slope processes to stream inputs are most important. An obvious link to riparian studies is present. Soil chemistry investigations should focus on the fate of nutrients, such as sulfur and nitrogen that are volatilized by fire, with some attention to nutrients that are transformed to a soluble state in which they may be leached from the system. The magnitude of these effects is a function of fire intensity (Klock and Grier 1979). Long-term productivity of ecosystems may depend on the ability of nitrogen fixing plants to replace nitrogen. Sulfur may continue to be a limiting nutrient, particularly in volcanic soils. Effects of fire on biological properties of soils has been undervalued in the past; mycorrhizae and mesofaunal interactions with fire may be very important for tree regeneration, soil development, and nutrient cycling in the moderate and high-severity fire regimes.

Fire indirectly affects food, cover, and water for wildlife, but direct effects are very minor. Increased use of fire will generally increase water availability to wildlife where it has been a limiting factor, particularly in drier envirorunents. This also has a tie to riparian issues because of potentially increased water flows. Both thermal and hiding cover are likely to be reduced for deer and elk regardless of what is done with managed fire because excessive live-tree density appears to be nonsustainable with or without fire. One important facet will be the value over time of dead understory, created by fire or insect mortality, as hiding or thermal cover. In general, fire will increase summer range forage, but winter range food, such as shrubs, may decrease in cover but have improved palatability. Analyses of the needs of critical wildlife species will be important.

Air quality is the major constraint to fire use in eastside forests, as it is elsewhere. Particulate emissions generated by fire is the most important air-quality parameter when assessing visibility reduction and potential health risks to the public and to fire managers (Sandberg and Dost 1990). Biomass inventories should be tied to emission factors by vegetation type and fuel moisture.

Fire effects on the landscape have been documented in Douglas-fir and ponderosa pine ecosystems in other regions. Effects of fire and other manipulations on the visual quality of specific landscapes can be simulated by computer modelling. This effort needs to be extended to the grand fir series in moderate severity fire regimes and perhaps to western juniper and oak woodlands.

Eastside forests contain tens of thousands of cultural sites and artifacts, some of which may be at risk with increased fire use. More cultural resource inventories will be needed to project potential effects of fire on cultural sites and artifacts.

CONCLUSIONS

Fire was a frequent visitor to eastside ecosystems in the past, and will be in the future. Unlike the inability to predict or mimic some disturbances, people can choose, to some extent, the kind of relation they will have with fire. Past management decisions did not consider or foresee future effects of fire exclusion or fire use. Future choices should be based on the idea that natural resources management is a grand experiment, but not one that has to be unpredictable.

This experiment is grand not only in concept but in scale. If fire were to be reintroduced on a 15-year rotation to ponderosa pine ecosystems and on a 30-year rotation to mixed-conifer ecosystems, about 335,000 ha per year would need to be burned! This area is more than the eastside area now burned with managed fire, much of which is pile burned.

We will have to make better use of available prediction tools, and develop new ones. Considering ecological relations by climax series and plant association groups is useful in understanding major processes and effects, and how each of them varies. Ultimately, insect, disease, and fire hazards should be understood by climax series and these same or similar plant association groups. Landscape ecology organizes and integrates information about fire and other ecosystem components and processes. Increasing and applying this information to eastside ecosystems will ensure their sustained productivity in the broadest biological and social context.

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GLOSSARY

Aspect-The direction a slope faces.

Avoider-A life-history strategy of plants with little adaptation to fire.

Climax-Species or communities representing the final (or an indefinitely prolonged) stage of a sere.

Crown fire-A fire burning into the crowns of the vegetation, generally associated with an intense understory fire.

Endurer-A life-history strategy of plants to fire where the plant resprouts or endures the effects of fire.

Evader-A life-history strategy of plants to fire where long-lived propagules are stored in soil or canopy and evade elimination from the site after fire.

Fire frequency-The return interval of fire.

Fire predictability-A measure of variation in fire frequency.

Fire regime-The combination of fire frequency, predictability, intensity, seasonality, and extent characteristic of fire in an ecosystem.

Fire severity-The effect of fire on plants; for trees, often measured as percentage of basal area removed.

Fireline intensity-The rate of heat release along a unit length of fireline, measured in kW m^{-1}

Foehn wind-A dry wind associated with windflow down the lee side of a plateau or mountain range.

Habitat type-The land area capable of supporting a single plant association.

Initial floristics-A process of succession in which seeds or plants of later successional stages are present from the outset but are subordinate to other species (see relay floristics).

Invader-A life-history strategy of plants to fire where the plant, through highly dispersive propagules, invades the site after fire.

Natural fire rotation-A fire return interval calculated as the quotient of a time period and the proportion of a study area burned in that period.

Plant association-The basic abstract unit in the classification of potential vegetation, described by overstory/understory indicator species.

Plant community-An assemblage of plant species that occur widely enough across the landscape to be recognized as a unit; this assemblage can be a pioneer group of species, a late successional group, or a combination of both.

Plant series-Aggregations of plant associations with the same overstory dominant.

Prescribed fire-A fire ignited under known conditions of fuel, weather, and topography to achieve specified objectives.

Prescribed natural fire-A fire ignited by natural processes (usually lightning) and allowed to burn within specified parameters of fuels, weather, and topography to achieve specified objectives.

Rate of spread-The rate at which a fire moves across the landscape, usually measured in m sec^{-1}.

Relay floristics-A process of succession where one set of species prepares the site and is replaced by a new set (like passing the baton in a relay race); (see initial floristics).

Resister-A life-history strategy of plants to fire where the plant, through an adaptation such as thick bark, survives low-intensity fire relatively unscathed.

Seral-A plant species or community that will be replaced by another plant community if protected from disturbance.

Sere-The product of succession: a sequence of plant communities that successively occupy and replace one another in a particular environment overtime.

Spotting-Mass transfer of firebrands ahead of a fire front.

Succession-The process of change in plant communities.

Surface fire-A fire burning along the surface without significant movement into the understory or overstory, usually below 1-m flame length.

Timelag class-A method of categorizing fuels by the rate at which they can lose or gain moisture, indexed by size class of fuel.

Understory fire-A fire burning in the understory, more intense than a surface fire with flame lengths of 1 to 3 m

Vegetation zone-A land area with a single overstory dominant as the primary climax dominant. (Occasionally, zones are named after major seral species; other climax types may exist in the zone.)

Water repellency-The resistance to soil wettability, which can be increased by intense fires.

Wildfire-A human-caused or natural fire that is not meeting land management objectives.

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Agee, James K. 1994. Fire and weather disturbances in terrestrial ecosystems of the eastern Cascades. Gen. Tech. Rep. PNW-GTR-320. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 52 p. (Everett, Richard L., assessment team leader; Eastside forest ecosystem health assessment; Hessburg, Paul F., science team leader and tech ed., Volume III: assessment.)

Fire history and effects are described for grassland and shrubland ecosystems and the range of forested communities by plant series: ponderosa pine, Douglas-fir/ white fir/grand fir, lodgepole pine, western hemlock/western redcedar, and subalpine fir/mountain hemlock. The effects of extreme weather events, including unusual temperature, wind, or moisture have generally had less significant impact than fire. The management issues for the 1990s include both management and research issues, at a grand scale with which we have little experience. Ecosystem and adaptive management principles will have to be applied.

Keywords: Forest fire, fire history, *Juniperus occidentalis*, *Pinus ponderosa*, *Pseudotsuga menziesii*, *Abies grandis*, *Abies lasiocarpa*, *Pinus contorta*.

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