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Active Management of Dry Forests in the Blue Mountains: Silvicultural Considerations

David C. Powell; Forest Silviculturist
Supervisor's Office; Pendleton, OR

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> **Change is the only constant in nature**
> *Heraclitus, Greek naturalist, 6th century BC*

CONTENTS

1. INTRODUCTION

The Blue Mountains ecoregion extends from the Ochoco Mountains in central Oregon to Hells Canyon of the Snake River in extreme northeastern Oregon, and then north to the deeply carved basalt canyons and rimrock of southeastern Washington. This large area contains more than 5½ million acres of National Forest System lands (fig. 1).

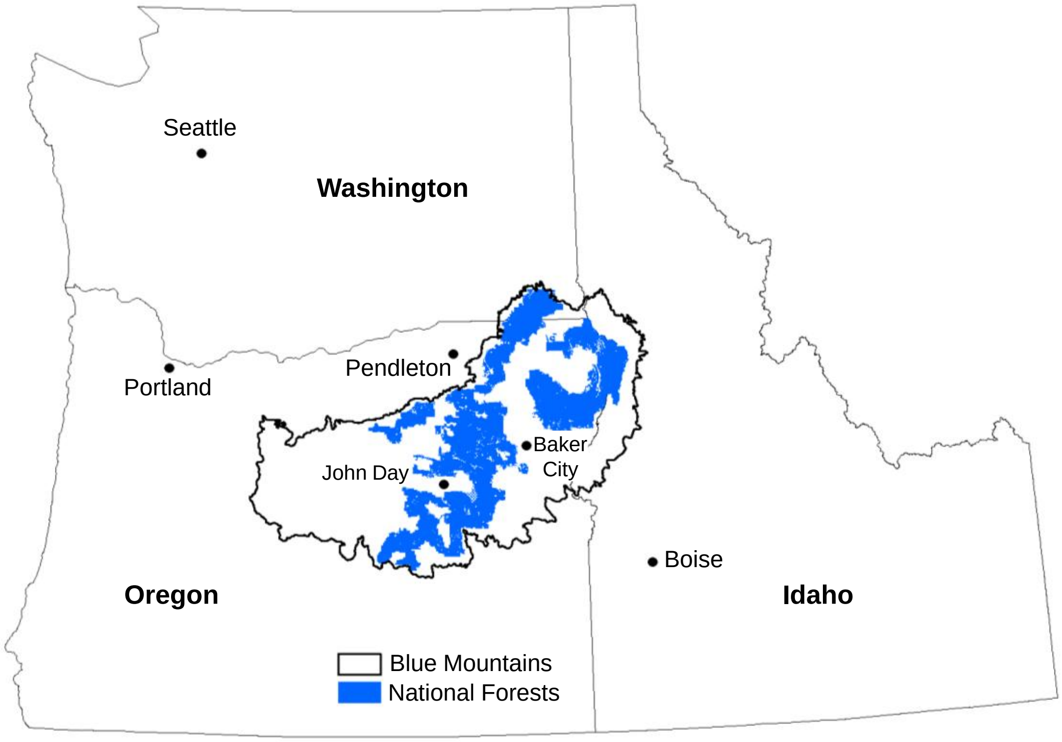


Figure 1 – The Blue Mountains ecoregion of northeastern Oregon, southeastern Washington, and west-central Idaho. The Blue Mountains ecoregion consists of a series of mountain ranges occurring in a southwest to northeast orientation, extending from the Ochoco Mountains in central Oregon, the southwestern portion of the ecoregion, to the western edge of the Seven Devils Mountains in west-central Idaho, the northeastern portion of the ecoregion. Blue shading depicts the spatial extent of three national forests in the Blue Mountains ecoregion: Malheur, Umatilla, and Wallowa-Whitman.

Beginning in the mid 1960s, the Blue Mountains experienced a series of insect outbreaks, disease epidemics, and wildfires. These disturbance events were viewed as unusually severe because they caused great amounts of damage or affected more area than was typical. The Blue Mountains eventually gained the dubious distinction of having perhaps the worst forest health in the western United States (Durbin 1992; East Oregonian 1992; Gray and Clark 1992; Kenworthy 1992; Lucas 1992, 1993; McLean 1992; Peterson 1992; Phillips 1995; Richards 1992).

Articles in magazines and newspapers contributed to a public perception that the Blue Mountains were experiencing a forest health crisis of unprecedented magnitude. This perception led to a series of broad-scale scientific assessments examining forest health effects and their underlying causes (Caraher et al. 1992, Gast et al. 1991, Henjum et al. 1994, Hessburg et al. 1999a, Johnson 1994, Lehmkuhl et al. 1994, Mutch et al. 1993, Quigley 1992, Quigley et al. 1996, Schmidt et al. 1993, Tanaka et al. 1995, Wickman 1992).

Among other things, the 1990s assessments concluded that:

- In the 1980s, an unusually severe outbreak of western spruce budworm, a defoliating insect whose habitat is mixed-conifer forest, functioned as a symptom of impaired forest health for the Blue Mountains, particularly for dry forest environments (Caraher et al. 1992, Gast et al. 1991, Johnson 1994, Mutch 1994, Powell 1994, Quigley 1992, Schmidt et al. 1993, Tanaka et al. 1995, Wickman 1992).

- It soon became apparent that budworm defoliation and other conditions contributing to a Blue Mountains forest health crisis were occurring throughout the interior Pacific Northwest and elsewhere in the western United States, particularly for dry forest environments (Everett et al. 1994, Hessburg et al. 1994, Hessburg et al. 1999, Lehmkuhl et al. 1994, Oliver et al. 1994c, Quigley et al. 1996, Sampson and Adams 1994).
- Fine-scale project planning corroborated findings from the broad-scale assessments by suggesting that certain symptoms of impaired forest health (such as uncharacteristic wildfire and insect impact) were largely related to species composition, forest structure, and tree density being outside their historical ranges of variability. Once again, this finding pertained mostly to the dry-forest portion of the Blue Mountains ecoregion.

We know that many of our fire-dependent, dry-forest ecosystems are degraded, with wildfire and other disturbance processes behaving much differently now than they did in the past. This report examines causes, effects, and possible responses to dry-forest deterioration, and it does so by using the following analytical framework (Egan and Howell 2001):

1. Define the ecological setting and historical context for dry-forest ecosystem components (species composition, forest structure, and tree density).
2. Identify some factors that may have contributed to dry-forest ecosystem changes through time.
3. Define what needs to be done to restore dry-forest ecosystem components.
4. Develop criteria for measuring the success of restoration activities.

The first sections characterize the ecological setting and provide an historical narrative for dry forests. The middle sections examine how fire exclusion, plant succession in the absence of recurrent fire, domestic and native ungulate herbivory, and selective timber harvest allowed the historically high resilience of dry-forest ecosystems to decline to low levels. Final sections describe restoration options for dry-forest ecosystems, including how active management treatments could be applied in such a way as to help recover some of their lost resilience.

The scope of this white paper is dry upland forests, a biophysical environment found predominantly on the southern half of the Umatilla National Forest and elsewhere in the central and southern Blue Mountains, and to a lesser extent on the northern half of the Umatilla National Forest and elsewhere in the northern Blue Mountains (fig. 2, table 1). The appendix provides a list of the potential vegetation types (plant associations, plant community types, plant communities) occurring in the dry upland forest potential vegetation group.

Active management. Human intervention into the nature, extent, and timing of disturbance to forest ecosystems for the purpose of obtaining desired goods and services (Haeussler and Kneeshaw 2003).
Resilience. The intrinsic properties allowing the fundamental functions of an ecosystem to persist in the face of extremes of disturbance. Resilience recognizes that systems have a capacity to absorb disturbances, but this capacity has limits and bounds, and when they are exceeded, the system may rapidly transform to a different state or developmental trajectory (Gunderson et al. 2010).

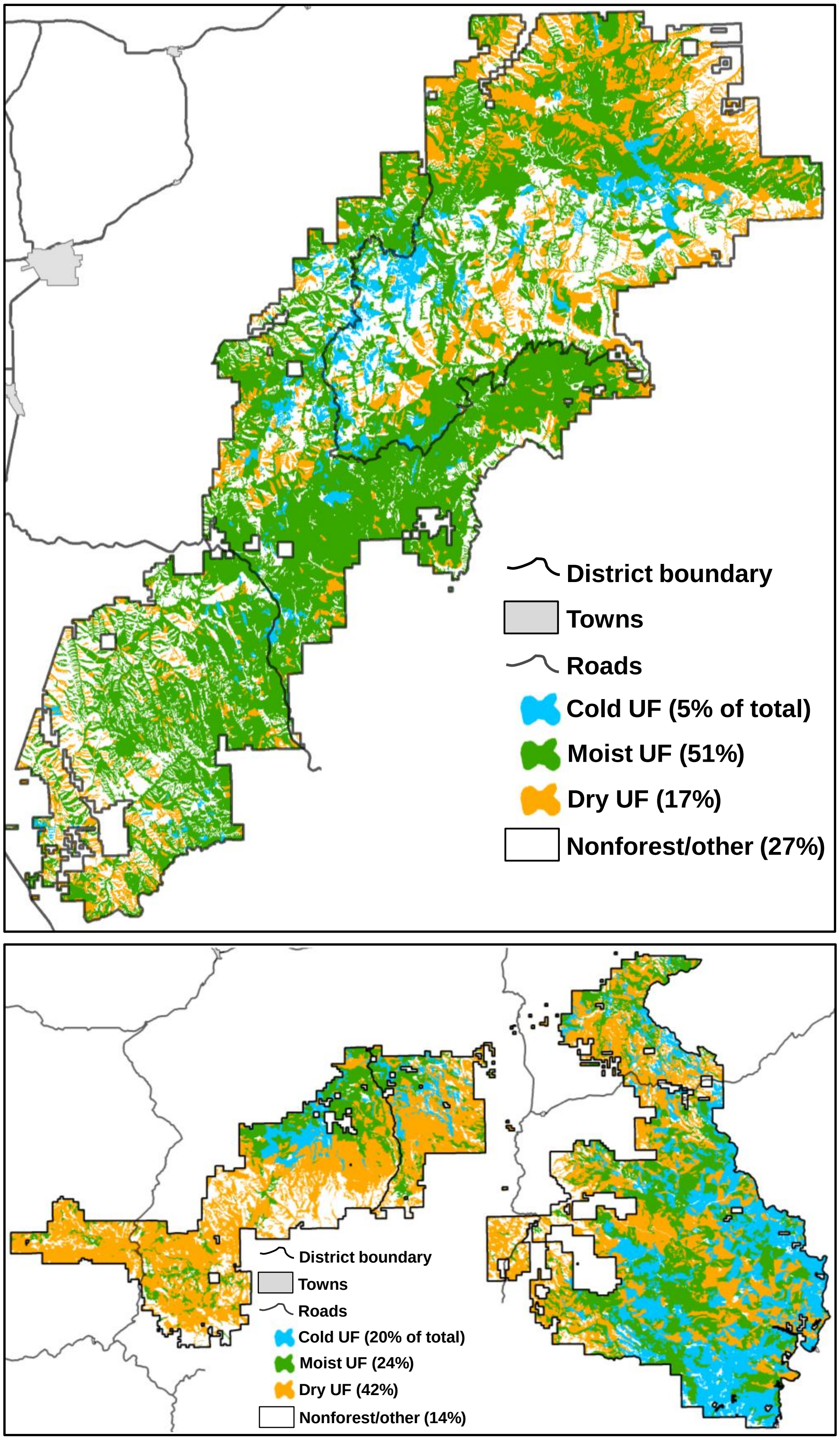


Figure 2 – Distribution of upland forest (UF) potential vegetation groups on the Umatilla National Forest (north-end districts above; south-end districts below).

Table 1: Acreage summary for upland forest potential vegetation groups of the Umatilla National Forest.

Potential Vegetation Group	North Half	South Half	Total
Cold Upland Forest	5,975 ac (4.5%)	125,491 ac (95.5%)	131,466
Pct. Of Forested	1%	21%	12%
Pct. Of Total	1%	19%	9%
Moist Upland Forest	420,923 ac (68%)	199,507 ac (32%)	620,430
Pct. Of Forested	79%	34%	55%
Pct. Of Total	58%	30%	44%
Dry Upland Forest	107,940 ac (29%)	263,530 ac (71%)	371,470
Pct. Of Forested	20%	45%	33%
Pct. Of Total	15%	39%	27%
Nonforest	193,340 ac (69%)	86,998 ac (31%)	280,338
Pct. Of Total	27%	13%	20%

Sources/Notes: Derived from spatial data available in the Umatilla National Forest geographical information system.

2. Ecological Setting

A distant summer view of the Blue Mountains shows a dark band of coniferous forest occurring above a lighter-colored grassland zone. Each of the two contrasting areas seems to be homogeneous, and the border between them appears sharp. A closer view reveals great diversity within each zone (fig. 3) and borders that are poorly defined: herbaceous communities and stands of deciduous trees are scattered throughout the coniferous forest, and the species of dominant conifer changes from one site to another (Powell 2000).

At the foot of the Blue Mountains, fingers of forest and ribbon-like shrub stands invade the grassland zone for varying distances before becoming progressively less common and eventually disappearing altogether. This vegetation pattern indicates that the Blue Mountains are actually broken up into a myriad of small units, many of which repeat in an intricate, changing pattern. Making sense of this landscape mosaic is possible using a concept called potential vegetation (Powell 2000).

Potential vegetation is defined as the community of plants that would become established if all successional sequences were completed, without interference by humans, under existing environmental conditions (Hall et al. 1995). It also implies that over the course of time and in the absence of disturbance, similar types of plant communities will develop on similar sites (Pfister and Arno 1980). For the Blue, Ochoco, and Wallowa mountains of northeastern Oregon and southeastern Washington, potential vegetation has been organized into two closely related hierarchies – a fine-scale hierarchy useful for project planning (Hall 1989), and a mid-scale hierarchy ideally suited for strategic assessments (Johnson et al. 1999, REO 1995).

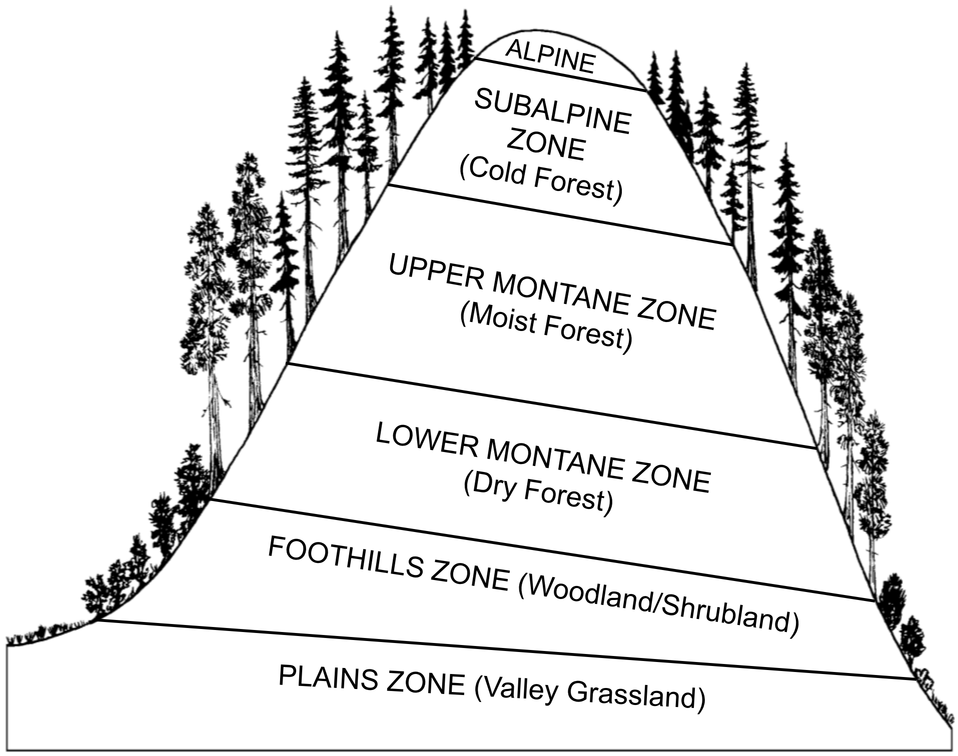


Figure 3 – Vegetation zones of the Blue Mountains. In the northern hemisphere, a south-facing slope receives more solar radiation than a flat surface, and a north-facing slope receives less (south slope is to the left, and north is to the right). These solar radiation patterns result in the vegetation zones or bands shown here – they are arranged vertically in response to elevation (moisture), and sloping downward from south to north (left to right) in response to slope direction or aspect (temperature).

The plains zone contains grasslands and shrublands because moisture is too low to support forests except along waterways. The foothills zone is usually dominated by western juniper, often with a mixture of mountain-mahogany shrublands. Located just above the western juniper woodlands is the lower montane zone, which contains dry mixed-conifer forests in the ponderosa pine, Douglas-fir, and grand fir potential vegetation series (dry mixed-conifer forests are the subject of this white paper). This dry mixed-conifer zone consists of 3 dry grand fir types, 21 ponderosa pine types, and 11 dry Douglas-fir types (plus a few other miscellaneous types; see appendix). The upper montane zone includes moist forests in the Douglas-fir, grand fir, and subalpine fir series. High elevations support a subalpine zone with Engelmann spruce and subalpine fir, or an alpine zone near mountain summits where trees are absent. Neither subalpine nor alpine environments are common in the relatively low-elevation Blue Mountains.

Potential vegetation group (PVG). An aggregation of plant association groups (PAGs) with similar environmental regimes and dominant plant species. Each group (PVG) typically includes PAGs representing a predominant temperature or moisture influence (Powell et al. 2007). The scope of this white paper is the Dry Upland Forest PVG.

Plant association group (PAG). Groupings of plant associations (and related potential vegetation types such as plant communities and plant community types) representing similar ecological environments as characterized by using temperature and moisture regimes. The most common PAG in the Dry Upland Forest PVG is the Warm Dry Upland Forest PAG. [Both definitions derived from Powell et al. (2007).]

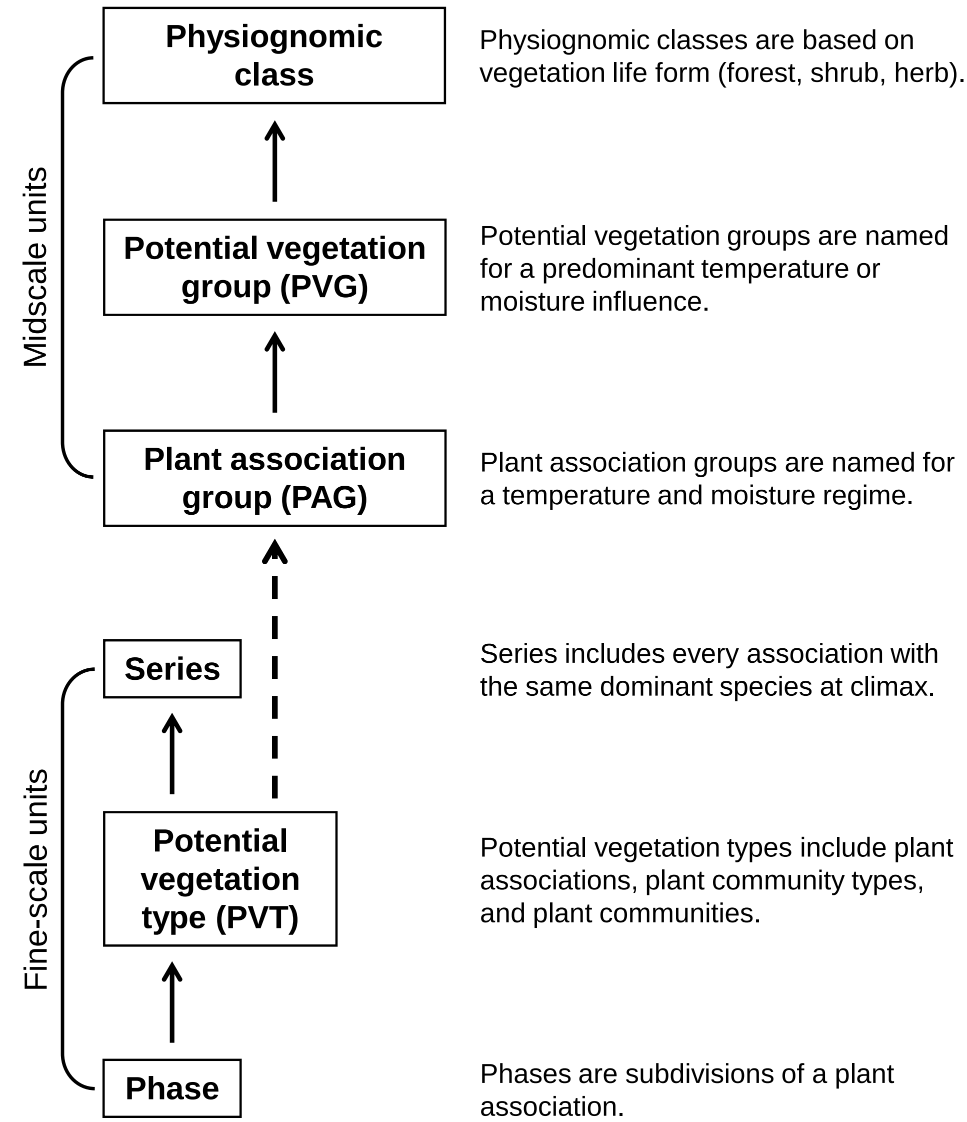


Figure 4 – Hierarchy of potential vegetation (PV) for the Blue Mountains (from Powell et al. 2007). PV taxonomic units have been organized as two integrated portions of a hierarchy. The fine-scale hierarchical units are described in PV classification reports and their associated keys (Crowe and Clausnitzer 1997, Johnson 2004, Johnson and Clausnitzer 1992, Johnson and Simon 1987, Johnson and Swanson 2005, and Wells 2006). Potential vegetation types (PVTs) provide a link between the fine- and mid-scale portions of the hierarchy because PVTs are aggregated to form plant association groups.

The mid-scale potential vegetation hierarchy has three levels: physiognomic classes, potential vegetation groups, and plant association groups (Powell et al. 2007). Since plant associations (potential vegetation types) are aggregated to form plant association groups, the plant association provides a link between the fine- and mid-scale hierarchies (fig. 4).

Potential vegetation (PV) is used to classify biophysical environments because it has an important influence on ecosystem processes. It is the ecological engine that powers vegetation change – it controls the speed at which shade-tolerant species get established beneath shade-intolerant trees, the rate at which forests produce biomass, and the impact that fire, insects, pathogens, and other disturbance agents have on ecosystem composition and structure. The implications of these processes are predictable (within limits) because they can be related to PV, and sites with similar PV behave in a similar way (Cook 1996, Daubenmire 1961).

Because of its predictive power, PV is useful for estimating the impact of disturbance processes and management activities on differing ecological environments. For example, a prescribed fire with a flame length of 2 feet and a fireline intensity of 25 BTU/ft/sec has relatively benign, nonlethal results when used on dry sites where overstory trees have thick bark (ponderosa pine, Douglas-fir, western larch). The same activity has dramatically different results (near-complete tree mortality) on cold sites dominated by thin-barked firs and lodgepole pines.

2.1 Dry Upland Forest Potential Vegetation Group

Dry upland forests occur at low to moderate elevations of the montane vegetation zone. Late-seral stands are dominated by ponderosa pine, grand fir, or Douglas-fir as the climax species; ponderosa pine or Douglas-fir function as early- or mid-seral species depending on plant association. Western juniper is expanding into this PVG as a result of fire exclusion and climate change (Gedney et al. 1999, Quigley et al. 1996), moving upward from a woodland zone located below the montane zone. Dry forests are adjoined by moist upland forests at their upper edge, and by woodlands and shrublands of the foothills vegetation zone at their lower edge (fig. 3).

For the Blue Mountains, the Dry Upland Forest PVG consists of three plant association groups (PAG) – one from the warm temperature regime (Warm Dry PAG), and two from the hot temperature regime (Hot Moist and Hot Dry PAGs). Of the three PAGs, Warm Dry is by far and away the most common member of the Dry UF potential vegetation group. The warm dry PAG supports dry mixed-conifer forests, with 3 dry grand fir potential vegetation types, 21 ponderosa pine types, and 11 dry Douglas-fir types (plus other miscellaneous types; see appendix).

Warm, dry forests tend to be the most common forest zone in the Blue Mountains, and because they occur at the lowest forested elevations, they have a long history of human use – both for commodity purposes (such as domestic livestock grazing), and as an area where effective fire exclusion occurred early on and eventually led to obvious changes in species composition, forest structure, and stand density. Dry-forest sites were historically dominated by ponderosa pine because it is well adapted to survive in a fire regime featuring low-severity fires occurring every 5 to 20 years (Agee 1996b; Hall 1976, 1980).

Common dry-forest undergrowth species feature graminoids and mid-height shrubs. Elk sedge and pinegrass are ubiquitous graminoids, while birchleaf spiraea, snowberry, ninebark, and bitterbrush are common shrubs. On the very driest sites, the Dry UF PVG has mountain-mahogany, big sagebrush, bluebunch wheatgrass, and western juniper (Hot Dry PAG).

Insect and disease agents of notable importance for dry-forest sites include defoliating insects such as western spruce budworm and Douglas-fir tussock moth (but only in those situations where Douglas-fir and grand fir invaded stands historically dominated by ponderosa pine), Douglas-fir dwarf mistletoe, and western dwarf mistletoe and bark beetles in ponderosa pine.

3. HISTORICAL CONTEXT

When Euro-Americans pushed up the Oregon Trail into the Blue Mountains of northeastern Oregon in the mid-1800s, they encountered a strikingly beautiful forest unlike any they had seen on the way west (fig. 5). Widely spaced ponderosa pines formed a towering canopy over an understory so free of brush and small trees that settlers could often drive their wagons through the forest as if it was a carefully manicured park (Evans 1991, Kenworthy 1992, Murphy 1994).

Oregon Trail diarist Rebecca Ketcham described this open condition well in her journal entry for Tuesday, September 6, 1853, written just after her party left the Grande Ronde River valley near La Grande, Oregon as they continued their journey to the northwest (Evans 1991):

> "Our road has been nearly the whole day through the woods – that is, if beautiful groves of pine trees can be called woods. I can almost say I never saw anything more beautiful, the river winding about through the ravines, the forests so different from anything I have seen before. The country all through is burnt over, so often there is not the least underbrush, but the grass grows thick and beautiful. It is now ripe and yellow and looks like fields of grain ripened, ready for the harvest."

Rebecca Ketcham was not the only emigrant who noticed fire's influence on vegetation conditions along the Oregon Trail. When 66 accounts from a book synthesizing journals by 19th century travelers on the Blue Mountains portion of the Oregon Trail (Evans 1991) were analyzed, 89% of them referred to open ponderosa pine stands, and 54% noted burned underbrush or grassy glades, much smoke in late summer and fall, or a lack of underbrush or dense tree thickets (Wickman et al. 1994).

Selected passages from Evans' book describing fire and vegetation conditions are provided below; misspellings from the original journals are retained in the excerpts (Evans 1991):

> "…the grass has been lately consumed, and many of the trees blasted by the ravaging fire of the Indians. These fires are yet smouldering, and the smoke from them effectually prevents our viewing the surrounding country, and completely obscures the beams of the sun."
> *Journal of John Kirk Townsend, August 31, 1834*

> "Came to trees, at first quite thin & without underbrush having fine grass. But as we arose we came to a densly timbered country, mostly pine & fir. The most beautiful tall straight trees. Our traviling through the timber was quite difficult as the path wound back and forth and many logs lay across it." *Journal of Medorem Crawford, September 12, 1842*

> "They [mountains] are mostly covered with high bunch grass, which at this season is quite dry. This often gets on fire, burning for miles and days together. One of these burnings is in sight of us today. It is on the opposite side of the river from us, or I should feel alarmed.
>
> The fire in the mountains last night was truly grand. It went to the tops of them spreading far down their sides. We were obliged to go over after our cattle at dark and bring them across the stream. The fire extended for several miles, burning all night, throwing out great streamers of red against the night sky. This morning there is none visible." *Journal of Esther Hanna, August 15-16, 1852*

Figure 5 – Open ponderosa pine forest with herbaceous undergrowth (stand of old-growth *P. ponderosa* near Whitney, Oregon, ca. 1900 [J. W. Cowden]; courtesy Gary Dielman, Baker City library). Pioneer journals (Evans 1991), early forestry surveys (Gannett 1902, Munger 1917), and fire history studies (Heyerdahl 1997, Maruoka 1994) suggest that many dry-forest sites in the Blue Mountains had presettlement conditions resembling those depicted in this image, particularly for the Douglas-fir/pinegrass and grand fir/pinegrass plant associations (Weaver 1967a, b). These biophysical settings featuring a warm and dry temperature-moisture regime, in combination with a disturbance regime dominated by frequent surface fires, created and maintained the distinctive composition and structure shown here.

According to these journal accounts, forest conditions at low and middle elevations consisted mainly of ponderosa pine, the pine forests were open and park-like with grass and herbs as the predominant undergrowth vegetation, and fire was a common occurrence in late summer and autumn (Evans 1991, Wickman et al. 1994). We can surmise that the typical landscape pattern was a fine-scale mosaic of stands of varying ages and stages of development, with young stands a result of infrequent, stand-replacing fires or bark-beetle outbreaks.

H.D. Foster (1908) described the open, park-like condition well when he observed, “the forest floor is open, free from underbrush in any quantity, so much so that it is possible to ride in almost any direction through the forest without following trails.”

It is widely reported that the Blue Mountains were named to commemorate a bluish haze enveloping them during late summer and fall when fires were burning (Mutch et al. 1993). The two journal entries below (Beckham 1991, Evans 1991), however, speculate that their name commemorates the blue-green color imparted by extensive pine stands. In either case, fire was an important reason for the name because it was not only responsible for smoke, but also for maintaining the ponderosa pine forests that would have been rare without underburning.

> "It is probable that they have received their name of the Blue mountains from the dark-blue appearance given to them by the pines."
> *Journal of Captain John Charles Fremont, October 17, 1843*.

> "I presume these mountains take their name from their dark blue appearance being densely timbered with pine timber, which being ever green gives the forest a sombre appearance, besides the limbs of the trees are all draped with long festoons of dark colored moss or mistletoe." *Journal of John or David Dinwiddie, August 30, 1853*.

Almost fifty years after Rebecca Ketcham's observations, scientist and geographer Henry Gannett examined Oregon's forests during a survey of federal forest reserves. Fire's effect on vegetation was clearly recognized during his survey, as described below (Gannett 1902):

> "The burns are greatest and most frequent in the most moist and most heavily timbered parts of the state, and are smaller and fewer where the rainfall is less and where the timber is lighter. This is owing to the density and abundance of the undergrowth in the heavily forested regions, which feeds the fire and vastly increases its heat.
>
> In the comparatively sparsely timbered southern portions of the Coast Range and the Cascades and in the Blue Mountains, where the forests are largely or mainly of yellow pine in open growth, with very little litter or underbrush, destructive fires have been few and small, although throughout these regions there are few trees which are not marked by fire, without, however, doing them any serious damage."

3.1 Ponderosa Pine in Eastern Oregon

Thornton T. Munger, first director of the U.S. Forest Service's Pacific Northwest Research Station, examined eastern Oregon's ponderosa pine forests more than a decade after Gannett's survey; he made these observations about the forest's structure and composition, including frequent comments about the obvious influence of fire (Munger 1917):

> "In most of the pure yellow-pine forests of the State the trees are spaced rather widely, the ground is fairly free from underbrush and debris, and travel through them on foot or horseback is interrupted only by occasional patches of saplings and fallen trees. The forests are usually not solid and continuous for great distances, except along the eastern base of the Cascades, but are broken by treeless 'scab-rock ridges,' or natural meadows.
>
> In the Blue Mountains the herbage is rather more luxuriant and varied than on the eastern slopes of the Cascades and their outstanding ranges. In the early summer the open yellow-pine forests are as green with fresh herbage as a lawn, except here and there where the green is tinged with patches of yellow or purple flowers. Some of this luxuriant herbage is pine grass (*Calamagrostis* sp.), a plant which is not eaten by stock except very early in the season; but much of the ground cover makes excellent range for cattle and sheep.
>
> In the Blue Mountains western larch (*Larix occidentalis*) is its [western yellow pine] usual companion and grows with it in an intimate and harmonious mixture. In the moister situations white fir (*Abies concolor*) is a common associate, as is also Douglas fir (*Pseudotsuga taxifolia*) in most parts of the State. In the Blue Mountains it is common for the south slopes to be covered

with a fine stand of yellow pine, while the north slopes are covered almost entirely with larch, white fir, and Douglas fir.

In the Blue Mountains the reproduction of yellow pine is very abundant, both in the virgin forest and after cuttings. Perhaps it is more prolific here than anywhere else. In this region where an area has not been burned over by a surface fire for a number of years, there is quite commonly a veritable thicket of little trees from a few inches to several feet high. Actual counts have shown that there are sometimes 14,000 seedlings on a single acre, the ages ranging from 13 to 21 years.

Yellow pine grows commonly in many-aged stands; i.e., trees of all ages from seedlings to 500-year-old veterans, with every age gradation between, are found in intimate mixture. Usually two or three or more trees of a certain age are found in a small group by themselves, the reason being that a group of many young trees usually starts in the gap which a large one makes when it dies.

Light, slowly spreading fires that form a blaze not more than 2 or 3 feet high and that burn chiefly the dry grass, needles, and underbrush start freely in yellow-pine forests, because for several months each summer the surface litter is dry enough to burn readily. Practically every acre of virgin yellow-pine timberland in central and eastern Oregon has been run over by fire during the lifetime of the present forest, and much of it has been repeatedly scourged.

It is sometimes supposed that these light surface fires, which have in the past run through the yellow-pine forests periodically, do no damage to the timber, but that they ‘protect’ it from possible severe conflagrations by burning up the surface debris before it accumulates. This is a mistake. These repeated fires, no matter how light, do in the aggregate an enormous amount of damage to yellow-pine forests, not alone to the young trees, but to the present mature merchantable timber.

A careful cruise of every tree on 154½ sample acres in typical yellow-pine stands in several localities in the Blue Mountains showed that 42 out of every 100 trees were fire-scarred.

Ordinarily, a fire in yellow-pine woods is comparatively easy to check. Its advance under usual conditions may be stopped by patrolmen on a fire line a foot or so wide, either with or without backfiring. The open character of the woods makes the construction of fire lines relatively easy, and in many places horses may be used to plow them.”

4. Influence Of Fire Exclusion

On dry-forest sites, fire’s influence was perhaps as important as sunlight and rain. The historical fire regime of frequent, low-severity surface fires maintained a pattern of large, widely spaced, fire-tolerant trees (fig. 5). These savanna forests supported trees with low flammability traits, and this contributed to ecosystem persistence (Bond and Midgley 1995).

For dry sites, dramatic reductions in fire frequency allowed tree seedlings and saplings, particularly of fire-sensitive species, to persist on the landscape in biophysical settings where most of them would have been eliminated by the historical fire regime (Agee 1996b, Cooper 1960, Munger 1917, Sloan 1998, White 1985, Wright and Agee 2004).

Fires in California's presettlement ponderosa pine type, for example, occurred on a frequency of about every 8 years between 1685 and 1889 (Show and Kotok 1924). In eastern Oregon, Keen (1937) sampled a 670-year-old ponderosa pine tree with 25 fire scars dating from 1481 to the mid 1930s, and it might very well have experienced more fires than that because not every fire creates a scar (Agee 1993).

Fire-dependent ponderosa pine forest ('park-like pine') was not unique to the Blue Mountains or to eastern Oregon; it was present in almost every forested region of the western United States, including northeastern California (Laudenslayer et al. 1989), western Montana (Gruell et al. 1982, Habeck 1990), central Idaho (Brock and Brock 1993), Colorado's Front Range (Marr 1967, Veblen and Lorenz 1991), and Arizona and New Mexico (Woolsey 1911).

Perhaps the most important reason for alteration and loss of park-like ponderosa pine forest has been exclusion of frequent wildfire, whose historical influence was so pervasive that Rebecca Ketcham, Henry Gannett, and Thornton Munger could hardly fail to notice its impact on forest composition and structure.

If Ketcham, Gannett, or Munger could return to the interior Pacific Northwest today, they would not recognize existing forest conditions, particularly for dry-forest sites. Gone are many of the ponderosa pines, some of which were harvested to make moldings, window sashes, and doors, in addition to crates for apples and other fruit crops (Bolsinger and Berger 1975, Gedney 1963). Other old-growth ponderosa pines succumbed to a widespread outbreak of western pine beetle in the early to mid 1930s (Cowlin et al. 1942, Weidman and Silcox 1936).

As ecologically benign fires crept through dry-site forests every 5 to 20 years, they eliminated brush and small trees in their wake (Everett et al. 2000, Franklin and Dyrness 1973, Hall 1976, Wright and Agee 2004). Historical fire ignitions probably came from a combination of lightning and human sources (Boyd 1999, Morris 1934). Fire intervals of less than 5 years are uncommon for the Blue Mountains (Heyerdahl 1997, Maruoka 1994, Hall 1976), suggesting that once a fire occurred, several years of fuel accumulation were required before the same area could burn again (Wright and Agee 2004).

Archaeological evidence indicates that humans have inhabited interior Columbia River basin ecosystems for at least 15,000 years (Knudson 1980). It is generally assumed that when Europeans arrived in the New World, American Indians sparsely occupied the land, the impacts of native peoples were relatively minor, and the landscape was pristine (Cronon 1996, Kay and Simmons 2002). Subsequent work has shown this assumption to be incorrect, as described here by ecologist Daniel Botkin:

> "It often seems that the common impression about the American West is that, before the arrival of people of European descent, Native Americans had essentially no effect on the land, the wildlife, or the ecosystems, except that they harvested trivial amounts that did not affect the 'natural' abundances of plants and animals. But Native Americans had three powerful technologies: fire, the ability to work wood into useful objects, and the bow and arrow.
>
> To claim that people with these technologies did not or could not create major changes in natural ecosystems can be taken as Western civilization's ignorance, chauvinism, and old prejudice

against primitivism – the noble but dumb savage. There is ample evidence that Native Americans greatly changed the character of the landscape with fire, and that they had major effects on the abundances of some wildlife species through their hunting" (Botkin 1995).

It is entirely possible that Blue Mountain forests were more primeval at the time of Euro-American settlement than before that era. When Columbus landed in 1492, it is estimated that North America (exclusive of Mexico and central America) supported at least 3.8 million Native Americans. By 1800, their numbers had been reduced to a million or less by measles, smallpox, cholera, influenza, and other European diseases (Denevan 1992, Mann 2006).

Even though their populations were already declining dramatically due to diseases introduced after European contact (Cook 1955), Native Americans of the interior Pacific Northwest may have expanded their use of fire in the early 1700s, perhaps to promote forage for the horses they had just acquired for the first time (Habeck 1987, Haines 1938, Humphrey 1943, Mosgrove 1980, Stewart 1951).

Recent investigations indicate that American Indians were far from the passive hunters and gatherers often depicted in western movies and novels. Their actions had a profound influence on the structure and composition of western ecosystems, a not unexpected result when considering they used hundreds of plants and animals for food, fiber, shelter, forage, and medicine. Fire was often their main tool for creating and maintaining the habitats required by these plants and animals (Boyd 1999, Denevan 1992, Kay 1994, Robbins 1997, Shinn 1980, Williams 2000).

Because ecosystems with native peoples differ markedly from those lacking an aboriginal influence, a hands-off approach by today's managers will not duplicate the conditions under which presettlement ecosystems developed (Botkin 1995, Boyd 1999, Christensen et al. 1996, MacCleery 1992, Stevens 1990, Vale 2002). Conversely, it is important to recognize that the technologies used by Native Americans to manage landscapes for thousands of years were far different than those employed by Euro-Americans (Aplet and Keeton 1999, Cronon 1996).

4.1 Plant Succession on Dry Sites

Suppressing underburns had the unintended consequence of allowing open stands of park-like ponderosa pine to be transformed into dense forests of grand fir and Douglas-fir (Harrod et al. 1999, Mast et al. 1999, Sloan 1998, Turner and Krannitz 2001) (fig. 6). Fire exclusion also transformed the structure of dry forests by shifting most of the canopy leaf area from the overstory layer to one or more understory layers. Ironically, many of these thick, multi-layered forests may be more attractive than the park-like pine stands they replaced because there seems to be an intuitive sense that when it comes to forests, lush is better (Gruell 2001, Scott 1998).

The tree species that invaded park-like pine forest – grand fir and Douglas-fir – have thin bark, low-hanging branches, highly flammable foliage, and other characteristics rendering them vulnerable to fire damage, particularly when they are small (table 2). With thick bark and few branches close to the ground, ponderosa pine and western larch easily resist the surface fires that eliminated firs and other invading tree species (Agee 1994, Cooper 1960, Dickman 1978, Weaver 1967b, White 1985).

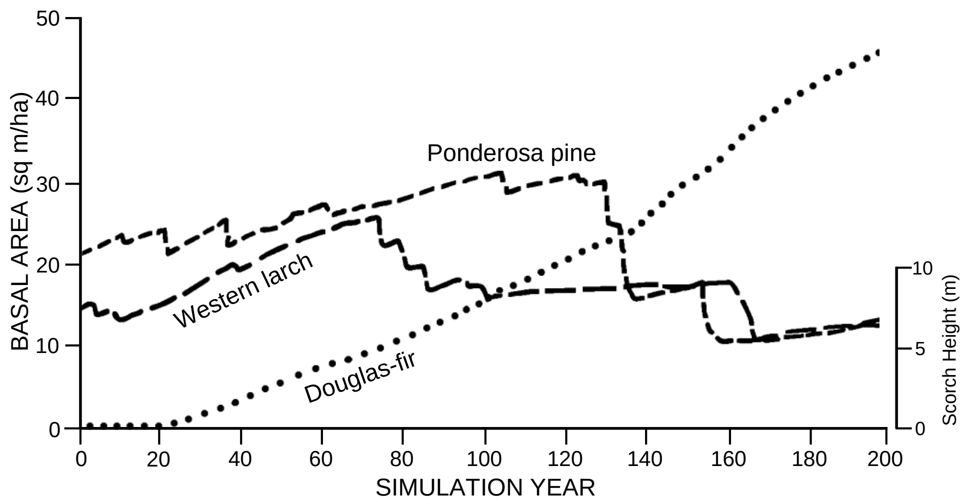


Figure 6 – Forest succession on the Douglas-fir/mallow ninebark (PSME/PHMA) plant association in the absence of recurring wildfire (adapted from Keane et al. 1990). In this simulation study, "compositional shifts from ponderosa pine and larch to Douglas-fir occurred in simulations of 50-yr fire intervals and with fire suppression. The simulated scenario of fire suppression (shown above) resulted in development of dense stands of relatively small trees. Such stands are susceptible to insect and disease infestations. They are also vulnerable to severe damage by wildfires because of heavy accumulations of dead fuels, and continuity of ladder and overstory fuels" (Keane et al. 1990). This study also demonstrates that regular fire intervals of 20 years or less result in Douglas-fir being essentially absent from dry-forest landscapes due to its high fire vulnerability as a seedling or sapling (Agee 1996). Since dry-site surface fires occurred in the Blue Mountains on a cycle of 5-20 years (Hall 1976, 1980), this study helps explain why species composition has changed dramatically on these areas.

When considering climate only (precipitation and temperature), Douglas-fir or grand fir are assuredly the climax species on dry, mixed-conifer sites of the Blue Mountains (the dry-forest PVG includes dry mixed-conifer forests in the ponderosa pine, grand fir, and Douglas-fir potential vegetation series; see appendix). But when surface fire is superimposed on climatic effects, the result is a marked change in vegetation composition because ponderosa pine, western larch, and other fire-adapted species are then put at a distinct advantage (Habeck 1976, Hall 1976).

In a plant succession context, dry-forest sites where surface fire favored dominance by ponderosa pine are generally early seral; areas where fire exclusion promoted establishment of grand fir and Douglas-fir are late seral (table 3). The pine-dominated early-seral condition is now rare, whereas fir-dominated late-seral stands are currently abundant (Arno and Allison-Bunnell 2002, Caraher et al. 1992, Habeck 1976, Hessburg et al. 1999b, Lehmkuhl et al. 1994).

Although the late-seral grand firs and Douglas-firs can establish under ponderosa pine in the absence of underburning, they may not have enough resilience to make it over the long run, let alone survive the next drought. This means that many of the late-seral stands replacing ponderosa pine are destined to become weak, and weak forests are susceptible to insect, disease, and fire outbreaks (fig. 7; Agee 1996, Covington et al. 1994, Filip et al. 1996, Filip and Schmitt 1990, Hessburg et al. 1994, Mutch et al. 1993, Oliver et al. 1994a, Powell 1994, Wickman 1992).

The successional roles of ponderosa pine and white (grand) fir were recognized by early silvicultural researchers, as demonstrated by the following comments about forest succession and development in the Sierra Nevada Mountains of central California (Dunning 1923):

> "Where natural conditions of site favor white fir, this species is destined to succeed yellow pine unless the normal succession is disturbed by fire or other accidents. Fir seeds germinate more abundantly than pine under stands of yellow pine, whose litter and shade exclude their own seedlings, and the young [fir] trees endure suppression longer. Moreover, height growth of fir is more rapid, and the total height attained is greater than for yellow pine. In the past occasional fires have been primarily responsible for sustaining yellow pine on fir sites.
>
> Fir seedlings and young trees are far more susceptible to fire damage than the pine because of their thinner bark with balsam cysts, more inflammable foliage, and small resinous terminal buds which are far less resistant than those of yellow pine. The fir is more often eliminated by fungi entering through fire scars than is pine. Exposure of mineral soil and openings created by fire favor yellow pine. Striking examples of the succession of white fir with fire exclusion may be seen in many places, where the mature stand is composed of practically pure yellow pine, while the reproduction beneath it is over 90 per cent white fir."

Plant succession. The process by which a series of different plant communities, and their associated animals and microbes, successively occupy and replace each other over time in a particular ecosystem or landscape location following a disturbance event (Kimmins 1997). The process of development (or redevelopment) of an ecosystem over time (Botkin 1990).
Surface fire. A fire burning primarily along the ground, consuming leaf litter (needles), grass, forbs, shrubs, short trees, fallen branches, and other fuels located on, or directly adjacent to, the forest floor (Scott and Reinhardt 2001). Surface fires tend to cause minimal damage to larger trees; historically, this was the prevailing fire type for ponderosa pine ecosystems throughout the western United States.

Table 2: Fire resistance characteristics for common conifers of dry-forest sites.

Tree Species	Bark Thickness	Rooting Habit	Bark Resin (Old Bark)	Branching Habit	Stand Density	Foliage Flammability	Overall Resistance
Ponderosa pine	Very thick	Deep	Abundant	Moderately high & open	Open	Medium	High
Douglas-fir	Very thick	Deep	Moderate	Moderately low & dense	Moderate to dense	High	High
Western larch	Very thick	Deep	Very little	High and very open	Open	Low	Very high
Grand fir	Thick	Shallow	Very little	Low and dense	Dense	High	Medium

Sources/Notes: Adapted from Flint (1925) and Starker (1934). Species rankings reflect the predominant situation for each trait. Tree species generally achieve fire tolerance by developing thick bark to protect their cambium, and by self-pruning to raise their lower crown above average flame height in the event of a fire. Species traits vary during the lifespan of an individual tree, and from one individual to another in a population. For example, grand fir's bark is thin when young, but relatively thick when mature. Fire occurred often on dry-forest sites, and on many moist-forest sites as well (see fire return interval data in table 3).

Table 3: Comparison of fire return interval and tree longevity, in years.

PVG	Fire Return Interval	Seral Stage	Predominant Tree Species	Tree Longevity (Years) Typical	Maximum
Dry Forest	15 Years	Early	Ponderosa pine	300	725
		Mid	Douglas-fir	200	500
		Late	Grand fir	200	400
Moist Forest	30-50 Years	Early	Western larch	300	915
		Mid	Western white pine	400	615
		Late	Grand fir	200	400
Cold Forest	80-110 Years	Early	Lodgepole pine	100	300
		Mid	Engelmann spruce	250	550
		Late	Subalpine fir	150	250

Sources/Notes: PVG (potential vegetation group) is described in Powell et al. (2007). Fire Return Interval is from Agee (1993; table 1.2, page 13). Seral Stage refers to a particular phase in the sequence of plant communities occurring after a disturbance event; seral communities are classified as early-, mid-, or late-seral depending on the successional role of their species composition (Hall et al. 1995). Predominant Tree Species shows the predominant species associated with each seral stage by PVG. Tree Longevity age values are from Powell (2000).

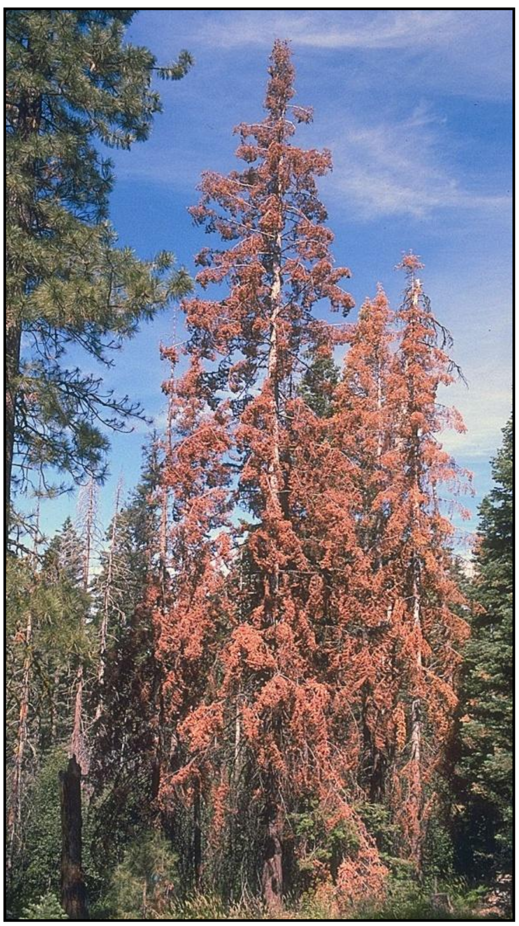


Figure 7 – Grand fir trees killed by fir engraver beetles (from Powell 1994). Defoliation, drought, root disease, dwarf mistletoe, overstocking, and other stressors increase a tree's susceptibility to bark beetle attack (Filip 1994, Filip and Schmitt 1990). Fir engraver and Douglas-fir beetles caused widespread damage in the Blue Mountains during the late 1980s and early 1990s. On dry-forest sites, bark beetles and other insects focus their attention on water-stressed and low-vigor trees (Schowalter and Withgott 2001). High-vigor trees are better able to ward off insect and disease attacks by producing phenols, terpenes, resins, and other defensive chemicals (Christiansen et al. 1987, Waring 1987). Thinning, a silvicultural practice, is used to release overcrowded trees from the effects of competition and improve their physiological condition and vigor (Oliver and Larson 1996). In the Blue Mountains, high stand density is known to favor at least eight forest insects and seven forest diseases or parasites, primarily because overstocking contributes to low tree vigor, and low vigor translates into reduced insect and disease resistance (Kolb et al. 1998, Langenheim 1990, Mitchell et al. 1983, Nebeker et al. 1995, Phillips and Croteau 1999, Pitman et al. 1982, Safranyik et al. 1998).

4.2 Fire's Influence on Site Nutrition

After frequent fires were suppressed following Euro-American settlement, microbial decomposition has been unable to process the rapidly accumulating organic debris (needles, twigs, and branches) on dry sites. Impaired decomposition and nutrient cycling rates can be the first signs of stress in dry-forest ecosystems (Bormann and Likens 1979). High organic matter levels on dry sites, with nutrients held in forms unavailable for plant growth, indicate that the decomposition and nutrient cycling processes are not functioning properly (Yazvenko and Rapport 1997).

Numerous studies have documented the slow decomposition rates associated with woody biomass in the western United States. This means that forests of the interior Pacific Northwest may have depended more on nitrogen-fixing plants and surface fire to cycle nutrients than on microbial decomposition of woody debris (Harvey 1994, Harvey et al. 1994) (fig. 8). And these two processes are obviously related because recurrent fire functioned to periodically rejuvenate snowbrush ceanothus, lupines, peavines, American vetch, russet buffaloberry, and other nitrogen-fixing plants (Hendrickson and Burgess 1989, Newland and DeLuca 2000).

Having nutrients tied up in pine litter, which decomposes much more slowly than grass litter (Hart et al. 1992), means that nutrient cycling has probably deteriorated for contemporary forests when compared with historical conditions (Cooper 1960; Covington and Moore 1994a, 1994b; Weaver 1943). Dry pine and mixed-conifer forests are accumulating biomass faster than it is being removed by timber harvest or wildfire, leaving millions of acres vulnerable to drought stress, insect and disease outbreaks, and uncharacteristic wildfire (Sampson and Adams 1994).

Providing adequate levels of site nutrition is important for maintaining tree resistance to insects and diseases (Mandzak and Moore 1994). In central Oregon, for example, Reaves and others (1984, 1990) found that ash leachates (e.g., chemical compounds produced when water percolates through the ash produced by a fire) from prescribed burns in ponderosa pine forests had a negative effect on the growth of *Armillaria ostoyae*, cause of Armillaria root disease. These studies found that much of the Armillaria suppression was related to a fungus called *Trichoderma* – a strongly antagonistic competitor of Armillaria root disease – and *Trichoderma* apparently benefited from ash leachates (Filip and Yang-Erve 1997; Reaves et al. 1984, 1990).

On low-productivity sites (generally dry areas with coarse or shallow soils, and thin forest floors), broadcast burning can be detrimental from a nutritional standpoint. The short-term benefits of prescribed fire may be offset by high soil pH, nitrogen and sulfur deficiencies, and other nutritional problems later in a forest's life (Brockley et al. 1992, DeBell and Ralston 1970, Mandzak and Moore 1994, Tiedemann 1987).

In central Oregon, prescribed fire was observed to cause a net decline in nitrogen mineralization rates and long-term productivity (Cochran and Hopkins 1991, Monleon et al. 1997). But a reduction in site productivity following prescribed fire might not be entirely due to nutrient cycling issues – up to 40 percent of a tree's annual net production on low-productivity sites is used to produce fine roots (Keyes and Grier 1981), and because these roots are located near the soil surface, they can be damaged or killed by prescribed fire, particularly when fire is applied in spring. In a study involving ponderosa pine on the Wenatchee National Forest in eastern Washington, wood increment was suppressed on spring-burned areas for at least 8 years after treatment, and much of this growth reduction was attributed to fine-root damage (Grier 1989).

The forest floor can also play an important role in an ecological process called allelopathy (Rose et al. 1983, Tinnin and Kirkpatrick 1985, Wardle et al. 1998). Allelopathy refers to a competitive strategy in which some plant species produce chemical compounds interfering with the germination, growth, or development of competing plants. The chemicals produced during allelopathy are often referred to as phytotoxins (Kelsey and Harrington 1979, Rietveld 1975).

If phytotoxins are produced by a climax species, such as ponderosa pine on dry-forest sites where moisture is too limiting and growing-season temperatures are too extreme to allow establishment of Douglas-fir or grand fir, then the phytotoxins would obviously affect its own offspring. In situations where a dominant plant species produces chemicals limiting its own abundance, the phytotoxin is referred to as an autotoxin (i.e., a 'self-toxin').

If ponderosa pine produces an autotoxic chemical on sites where it is the climax tree species, and this hypothesis has not been definitively proven to my knowledge, then it could confer survival value to the species. When moisture is limiting, as it so frequently is for dry-forest sites, and when these growth-inhibiting conditions occur in an ecosystem where short-interval surface fire was the prevailing disturbance process, then adequate tree survival and growth can only be maintained at relatively low tree densities. Therefore, chemicals from the mature trees could function as 'density regulators' by reducing the germination and growth of its own progeny (Kelsey and Harrington 1979).

Using allelopathy to regulate seedling density could be an important evolutionary adaptation because it would limit or prevent overcrowding, stagnation, and competition between individuals of the same species. This would ensure that some seedlings could grow fast enough to reach a size allowing reasonable resistance to the frequent surface fire regime operating on dry-forest sites (Biswell 1973, Cooper 1960, White 1985).

Fred Hall, a Forest Service ecologist, speculated that a selective inhibitory substance is present in ponderosa pine litter, and that it is destroyed by periodic underburning (Hall 1991). Without fire, this substance could accumulate in the upper mineral soil (or in the organic horizons?) and reduce ponderosa pine establishment and growth. And we already know that leachate from pine litter and pinegrass leaves has been shown to retard root growth of germinating ponderosa pine seeds (Eckert 1975, Jameson 1968, Kelsey and Harrington 1979, McConnell and Smith 1971, Rietveld 1975), perhaps corroborating Hall's suspicion. But when considering the impact of pathogenic fungi located in the forest floor's organic horizons (Daniel and Schmidt 1972), I wonder if Fred's 'selective inhibitory substance' might have involved pathogenic fungi, allelopathic phytotoxins, or perhaps some combination of both?

Although not all of the causal mechanisms are understood, it is clear that when plant succession occurs on dry-forest sites in the absence of recurring wildfire, it will eventually result in reduced availability of mineral nitrogen and cause increased accumulation of polyphenolic compounds in the mineral soil (MacKenzie et al. 2006, Souto et al. 2000, Wardle et al. 2000).

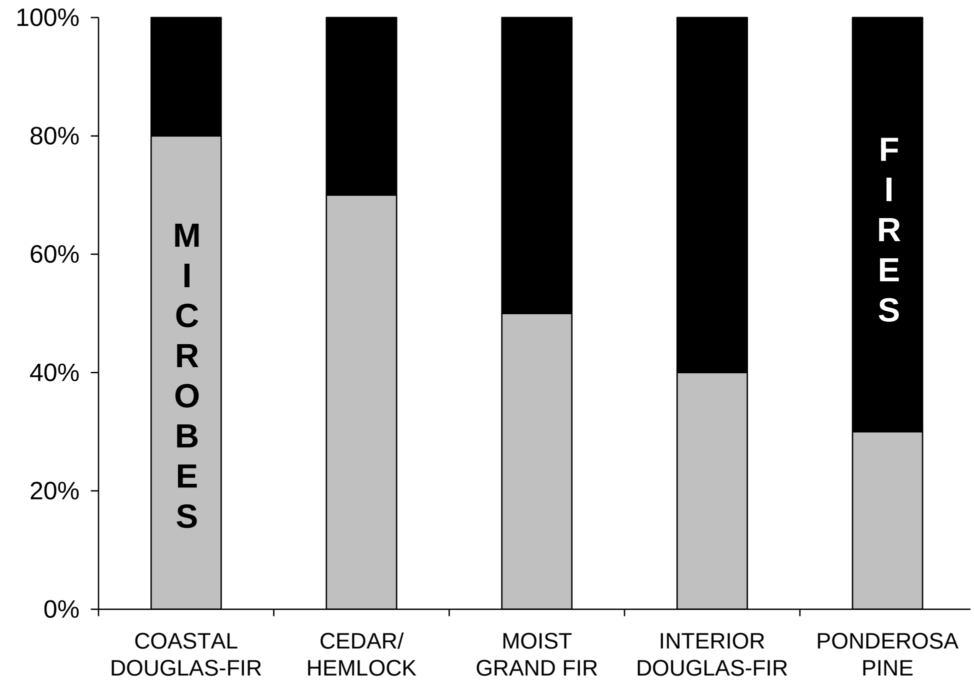


Figure 8 – Microbes and fire as agents of decomposition (adapted from Harvey et al. 1994). Fire (black portion of bars) and microbes (gray portion) are important decomposition and nutrient cycling agents. For the dry-forest climatic zone of the interior Pacific Northwest (the interior Douglas-fir and ponderosa pine forest types above), the short-interval fire regime (surface fires) was the primary cycling process because microbial decomposition is too slow to keep pace with biomass accumulation on these sites. Microbial decay is limited for cold or dry environments, allowing biomass to accumulate.

4.3 Awareness of Fire-Caused Changes

If fire exclusion caused major changes in ecosystem components (e.g., species composition, forest structure, and tree density) on dry-forest sites, then why weren't they recognized sooner? Actually, many of the changes were recognized early, but weren't acted on because of the prevailing attitudes of the time.

Two of the studies described earlier illustrate differing attitudes about fire's role in ponderosa pine forests. Gannett (1902) surveyed the federal forests before they were viewed as a source of commodities; he found many trees with fire scars (fig. 9) but fire had not done "them any serious damage." Munger (1917) found few stands without some sign of fire's influence and yet fire was a scourge causing an "enormous amount of damage to yellow-pine forests."

Munger's (1917) comments about fire-caused damage reflect the commodity paradigm of his era; ponderosa pine forests were to be managed as a sustainable source of wood products, and fire was perceived as little more than an obstacle to reaching that goal. William Greeley, an early Chief of the U.S. Forest Service, expressed the commodity philosophy this way (Greeley 1912):

> "To the extent to which the over-ripe timber on the National Forests cannot be cut and used while still merchantable, public property is wasted. This is the very antithesis of conservation."

Figure 9 – Many ponderosa pine trees have basal scars caused by recurrent surface fire, a pervasive disturbance process before wildfire exclusion efforts began around 1900 (image acquired by D.C. Powell on the North Fork John Day Ranger District, Umatilla National Forest, in October 2009). Species like ponderosa pine achieve fire tolerance by developing thick bark to protect their cambium, and by self-pruning their lower crown to raise the crown height base above average flame height in the event of a fire. "Both of these characteristics are size dependent; thick bark is a relative characteristic with individuals of larger diameter having thicker bark, and crown height is dependent on the height of individuals" (Roberts and Betz 1999). This means that fire tolerance is primarily a species-specific life history trait (see table 2), but it also varies with the size of individuals in a population.

Munger's commodity orientation was shared by other Forest Service researchers working in the western United States, as demonstrated by the following passage from *The Role of Fire in the California Pine Forests* (Show and Kotok 1924).

> "Physical conditions in the pine forests of California have led to the frequent recurrence of fires for centuries, but the fact that magnificent forests still cover large areas and give the appearance of well-stocked, vigorous stands has blinded the public to the harm that fires have done and are steadily working throughout the whole region.
>
> Were it possible for the observer to visualize the entire area on which pine has grown, and to behold it truly fully stocked, he would then see by comparison that the present California pine forests represent broken, patchy, understocked stands, worn down by the attrition of repeated light fires."

Land managers working for the early U.S. Forest Service also recognized that fire-caused changes were occurring on the landscape, as described in these two accounts:

> "There are patches of 'scabland,' characterized by very shallow soil, many rock fragments and a total absence of vegetation except in the spring months. It is interesting to note that some of these areas are being occupied by sagebrush where a few years ago, there was none. A possible explanation is that the annual fires of the Indians kept it killed out and now it has a chance to develop.

Yellow pine is slowly encroaching upon the sagebrush; the chief factor in its rate of advance being moisture, provided fire is kept out. The same statement will hold true in regard to the other open areas as well. As fast as the reproduction has pushed out from under the protection of the parent trees, the periodical fires have killed it back, thus keeping the timberline practically stationary.

In recent years, conditions have improved, and it is noticeable that the pine is reaching out, although slowly. The north slopes [are] being occupied by a thick stand of fir reproduction. Even pine is gaining a foothold here, and is gradually creeping across the ridge to the south slopes" (Evans 1912).

"Throughout the conifer type there is ample reproduction to more than replace the present stand of timber. The major part of the reproduction has come in since the forest has been protected against fires. Several areas were noticed where the yellow pine seedlings were so thick that it was almost impossible to ride through them. Practically all of the stockmen were complaining that the reproduction is coming in so thick on their allotments that it is greatly decreasing the carrying capacity of the range" (Aldous 1914).

When considered in the context of resulting changes to ecosystem composition and structure, fire exclusion was probably not an appropriate policy. The problem was not necessarily fire exclusion per se, but the fact that surrogates providing similar ecosystem functions, such as nutrient cycling and tree thinning, were not substituted for fire.

More than 60 years ago, an early fire ecologist (Harold Weaver) made some insightful observations about fire exclusion and its impact (Weaver 1943). Many of his comments have obvious relevance to our contemporary situation featuring uncharacteristic fire behavior in dry mixed-conifer forests, caused primarily by unusually high fuel accumulations (Arno and Allison-Bunnell 2002, Carle 2002, GAO 1999, Hessburg et al. 2005, Kenworthy 1992, Pyne 1997).

"It is obvious that the present policy of attempting complete protection of ponderosa pine stands from fire raises several very important problems. How, for instance, will the composition of the reproduction be controlled? If ponderosa pine is desired on vast areas how, unless fire is employed, can other species such as white fir be prevented from monopolizing the ground? On the other hand, if it is decided to permit such species as white fir to come in under mature ponderosa pine, how much of the public's money are foresters justified in spending in trying to keep fire out? Even with unlimited funds, personnel, and equipment, can they give reasonable assurance that they can continue to keep such extremely hazardous stands from burning up? If they feel reasonably sure of this, can they then give assurance that the timber products of such stands will be more valuable than those that might otherwise be derived from ponderosa pine and will in addition justify the high protection costs?"

Seral communities developing under the influence of recurrent disturbance can be ecologically resilient. Disturbance frequency determines the length of successional cycles for a particular ecological system. Ecosystems with frequent disturbances have continually interrupted successions and exhibit a relatively narrow range of plant communities and vegetation structure (Steele and Geier-Hayes 1995). A good example of a forest ecosystem maintained by frequent disturbance is the park-like ponderosa pine community (see fig. 5).

When Thornton Munger examined eastern Oregon's ponderosa pine forests in 1910-1911, an open park-like structure was clearly evident (Munger 1917):

> "In pure, fully stocked stands in the Blue Mountains region there are commonly from 20 to 30 yellow pines per acre over 12 inches in diameter, of which but few are over 30 inches. Over large areas the average number per acre is ordinarily less than 20." [Note that 20 trees per acre results in an equilateral (triangular) spacing of 50 feet between trees, which is most assuredly an open stand condition.]

These presettlement forests typically consisted of large trees with an open to moderately dense canopy, an understory featuring vigorous shrubs and herbs, and small patches of young trees. "Light and water could penetrate the forest canopy to nurture and maintain a healthy understory. The observation that more wildlife species are adapted to large-tree, open canopy forest than to any other combination of tree size and canopy closure suggests that open conditions were common" historically (Gruell 2001).

4.4 Summary: Changes Caused by Fire Exclusion

The dry-forest landscapes of today may look attractively lush, but their thickening canopy reflects a host of changes related to the long-term influence of fire exclusion:

1. Without fire to retard plant succession, grand fir and Douglas-fir invaded sites where ponderosa pine had been maintained as a fire disclimax (Lunan and Habeck 1973).
2. Deep layers of organic matter accumulated under thickening conifer forests, tying up nitrogen and other nutrients that are cycled slowly without fire (Harvey 1994).
3. Fire exclusion removed an important tree thinning agent, causing tree density to accumulate and eventually contributing to a wide variety of density-related changes:
   a. Bark beetle outbreaks occurred frequently in overstocked, second growth ponderosa pine forests (Sartwell 1971).
   b. Small trees killed by suppression (density-dependent mortality) were usually the shade-intolerant species succumbing quickly to intertree competition (fig. 10).
   c. Dense forests produce less water for streams and springs than open forests (Bosch and Hewlett 1982, Covington and Moore 1994b, Troendle 1983).
4. Light surface fires facilitated ponderosa pine regeneration by exposing some mineral soil, and by temporarily reducing competition from grasses and sedges (Hall 1976).
5. Surface fires raised the 'height to live crown base' by pruning the lower branches of overstory trees, reducing the potential for crown-fire initiation (Agee 1996, Keyes 1996).
6. By maintaining open stands and allowing perennial herbs to persist, fire provided forage for both livestock and wildlife (Hedrick et al. 1968, Irwin et al. 1994).
7. Fire maintained nutrient availability by rejuvenating snowbrush ceanothus, lupines, peavines, vetch, buffaloberry, and other nitrogen-fixing plants (Newland and DeLuca 2000).
8. Frequent fires maintained low fuel accumulations and low crown-fire susceptibility in areas with dry summers, high winds, and abundant lightning (Dodge 1972, Hall 1976).
9. Fire smoke limits germination of dwarf-mistletoe seeds (Zimmerman and Laven 1987), so fire exclusion probably contributed to worsening dwarf-mistletoe problems.

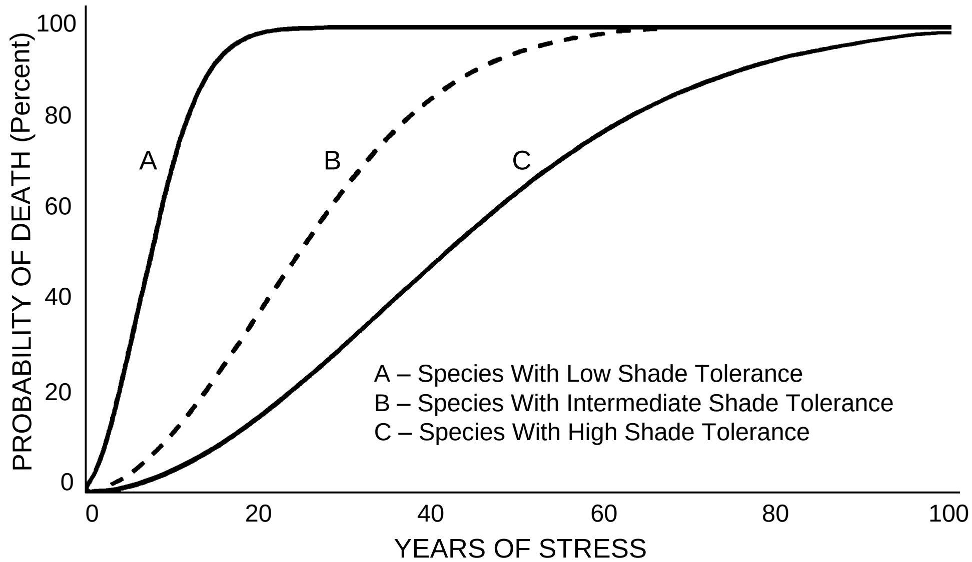


Figure 10 – Tree resistance to stress varies with shade tolerance (adapted from Keane et al. 1996). Intolerant tree species (lodgepole pine, ponderosa pine, western larch) die relatively quickly when exposed to chronic stress such as high stand density. Trees with intermediate tolerance (Douglas-fir and western white pine) can withstand a longer period of stress without dying. Shade tolerant species (Engelmann spruce, grand fir, subalpine fir) can endure relatively long periods of stress before experiencing mortality.

10. Fire exclusion allowed certain fire-sensitive shrubs (bitterbrush, sagebrush) to invade grasslands in eastern Oregon (Burkhardt and Tisdale 1976, Gedney et al. 1999).
11. Western juniper increased with fire exclusion, reducing water yields because juniper uses more water than the grasses and shrubs it replaced (Angell and Miller 1994).
12. Loss of an open park-like ponderosa pine structure apparently had a detrimental impact on nesting success for blue grouse (Pelren and Crawford 1999).
13. Tree mortality caused by density-responsive insects and diseases increased, particularly from bark beetles and defoliators (Anderson et al. 1987, Hadley and Veblen 1993).
14. Fire-intolerant conifers displaced fruit-bearing shrubs, deciduous trees, and herbaceous plants, all important food sources for wildlife (Bartos and Campbell 1998, Gruell 2001).
15. Fire exclusion created landscapes that are more homogeneous, with fewer vegetation types and lower patch densities (Lehmkuhl et al. 1994, Miller and Urban 2000).
16. Landscape diversity declined after fire was prevented from periodically creating early-seral plant communities (Hessburg et al. 1999b, Taylor and Skinner 1998).

4.5 Active Management Implications of Fire Exclusion

Over the last four decades, Blue Mountain forests experienced increasing impacts from wildfire, insects, and diseases. According to results from the many scientific assessments completed during this period, impacts associated with these disturbance agents are primarily related to changes in species composition, forest structure, and tree density, all of which were affected to a large degree by fire exclusion. For dry forests, low-severity surface fire is the keystone disturbance process, and its exclusion by human society has many consequences – some of which were intended, but many of which were not.

Fire exclusion allowed fire-resistant species (ponderosa pine primarily) to be replaced with fire-sensitive species (Douglas-fir when small, grand fir, and western juniper when small). This change affected both ecosystem resistance and resilience because dry forests cannot resist fire when their composition is dominated by fire-sensitive species, and they cannot sustain their resilience if a high proportion of the trees are killed by fire (see fig. 15 later in this paper).

The list in section 4.4 enumerates 16 ecosystem changes related to excluding fire from dry-forest sites. Although extensive, it still may not be a comprehensive accounting of all fire-exclusion influences – but it does provide an inkling of the vast scope of fire as an ecosystem process, including its effect on dwarf mistletoe seed germination and other life-history functions.

Active management treatments, particularly thinning and prescribed fire, can be implemented as restoration practices, in proper places and at appropriate times, to help recover and then sustain the resilience of these crucially important dry-forest ecosystems (section 7 – Restoration of Dry-Forest Ecosystems – provides a detailed restoration discussion).

5. Influence Of Ungulate Herbivory

Fire exclusion obviously influenced forest structure and composition, particularly for dry sites, but it is not the only factor to have done so. Many studies from western North America indicate that herbivory by wild and domestic ungulates has been as influential as fire exclusion in shaping wildland ecosystems (Belsky and Blumenthal 1997, Fleischner 1994, Hatton 1920, Madany and West 1983, Oliver et al. 1994b, Parks et al. 1998, Riggs et al. 2000, Rummell 1951, Steele et al. 1986, Zimmerman and Neuenschwander 1984).

Livestock, primarily cattle and sheep, were initially brought into eastern Oregon and eastern Washington during the 1840s via the Oregon Trail (Irwin et al. 1994, Oliver et al. 1994c). But Native American horse herds were already large and well established by then, having arrived in the Blue Mountains around 1730 after progressively migrating northward from the Santa Fe, New Mexico area (Haines 1938).

At the time of Euro-American settlement, much of the interior Pacific Northwest was covered with lush grass and other herbaceous vegetation (Galbraith and Anderson 1970, Humphrey 1943, Munger 1917). Forest inspector Harold Langille described rangeland conditions prior to extensive changes caused by heavy livestock grazing this way (Langille 1906):

> "A few years ago Eastern Oregon was one of the best range sections of the West. The rich bunch grass waved knee deep on hill and plain in such close growth that it was mowed with machines for hay."

During the summer and fall of 1861, large numbers of sheep and cattle were driven into eastern Oregon and eastern Washington from the Willamette River valley of western Oregon. The winter of 1861-1862, however, was one of the most severe ever recorded for the Pacific

Northwest and it almost wiped out this fledgling livestock industry (Galbraith and Anderson 1970, Humphrey 1943).

During the late 19th and early 20th centuries, immense bands of sheep grazed in the Blue Mountains (figs. 11 and 12), causing persistent changes in vegetation composition (Bright and Powell 2008, Coville 1898, Galbraith and Anderson 1970, Griffiths 1903, Humphrey 1943, Tucker 1940). Sheepherders made an annual migration with their flocks, following the snow from low elevations in the spring to high elevations in the summer, and then back to low elevations in the autumn (Darlington 1915, Oliver et al. 1994c).

Sheep grazing caused conflict and enmity between cattle ranchers, homesteaders, and sheepherders because the sheepherders were often nomadic (in contrast to cattle ranchers and homesteaders who tended to be year-long residents), and because conventional wisdom held that sheep caused rangeland deterioration to a greater extent than cattle (Lomax 1928, Minto 1902, Oliver et al. 1994c). Forest inspector Harold Langille described the sheep grazing situation well in this account from his 1906 report (Langille 1906):

> "Sheep from Wasco, Crook, Sherman, Gilliam, Umatilla and Morrow Counties are driven to the mountains early each season and ranged up to the very doors of the actual settlers and cattle owners. There has been some trouble in the past resulting in bloodshed, but nothing as serious as that which threatens to come about in the near future."

An early survey of sheep ranges found moist mountain meadows that were entirely devoid of vegetation and experiencing severe soil erosion. A complete collection of the herbaceous plants growing in a heavily grazed meadow found not a single perennial species, and no annuals exceeding two inches in height. Sheep browsing had damaged all shrubs other than snowbrush ceanothus; even the small ponderosa pines were fed upon (Griffiths 1903, Langille 1903).

When the Blue Mountains were surveyed in the early twentieth century, overgrazing was deemed to have been severe enough to influence whether forest cover was present or not, as described here by Forest Inspector Harold Langille during his examination of the Heppner Forest Reserve (Langille 1903):

> "It was everywhere observed that upon tracts upon which there is no forest cover there is no soil. At one time these areas were covered with soil to a depth of from one to two feet, and sufficient soil binding vegetation grew upon it to resist the destructive elements – wind and water – but persistent overgrazing destroyed this cover, and, there being no tree growth to protect the soil, it rapidly disappeared, leaving nothing but a bed of exposed rocks."

Figure 11 summarizes historical grazing trends for three classes of livestock (cattle and calves, sheep and lambs, horses and ponies) for nine counties in northeastern Oregon and southeastern Washington.

A dense sod of perennial graminoids provided nutritious forage for ungulates, but it also influenced tree regeneration patterns. Competition for soil moisture and nutrients, as well as allelopathic inhibition by grass and other herbs (Fisher 1980, Larson and Schubert 1969, McDonald 1986, Randall and Rejmanek 1993, Rietveld 1975), were critical factors limiting the establishment of tree seedlings (Cooper 1960, Kolb and Robberecht 1996, Pearson 1942, Rummell 1951).

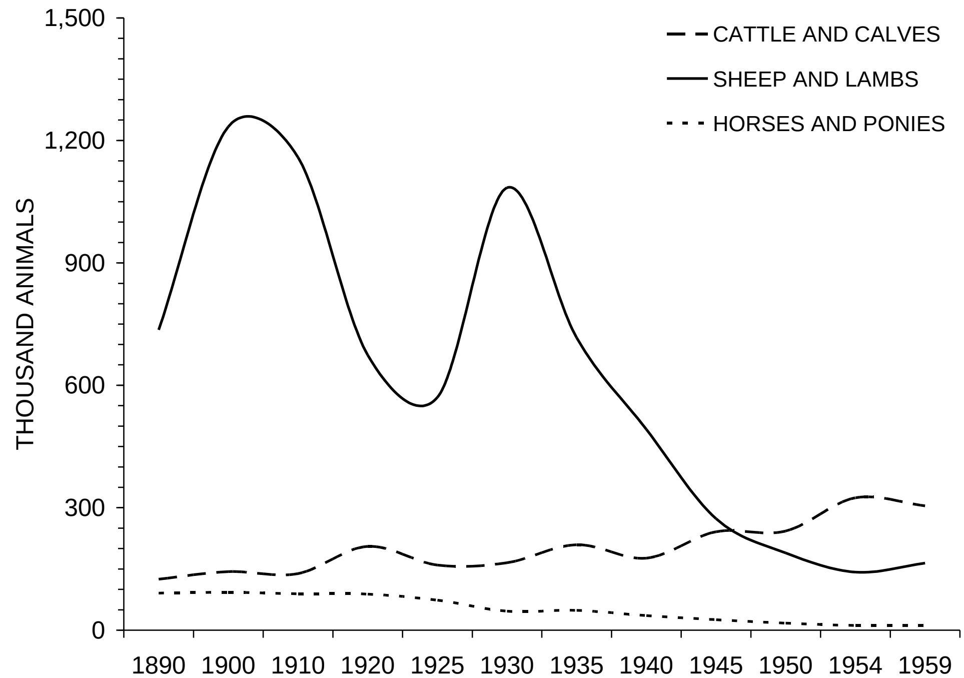


Figure 11 – Number of domestic grazing animals, summarized for three livestock categories, for nine counties in northeastern Oregon and southeastern Washington (Asotin, Columbia and Garfield in Washington; Grant, Morrow, Umatilla, Union, Wallowa and Wheeler in Oregon). Data derived from Bureau of Census agricultural summaries (Bureau of Census 1895, 1902, 1913, 1922, 1927, 1932, 1942, 1946, 1952, 1956, 1961).

Figure 12 – A band of sheep on the Wenaha National Forest (taken from Bright and Powell 2008; photograph taken by M.N. Unser in 1913). Sheepherders made an annual migration with their flocks, following the snow as it retreated from the lowlands in spring to the high country in summer, and then back down to the valleys in autumn.

Livestock herbivory removes plant foliage (forage); plants respond to this defoliation by reducing growth, particularly underground (root) growth (Schuster 1964). This means that livestock grazing may have made it easier for tree seedlings to germinate and survive. This was especially true for open stands of ponderosa pine because competition from graminoids and other herbaceous vegetation was an important factor regulating seedling establishment (fig. 13; Covington and Moore 1994a, Sloan and Ryker 1986, Yazvenko and Rapport 1997).

Domestic livestock grazing in the early 1900s was not the only factor that may have affected forest regeneration; in many areas, the impact of native ungulates (deer, elk) was more pervasive and, unlike domestic animals, continues at moderate or high levels today (Case and Kauffman 1997, Humphrey 1943, Parks et al. 1998, Riggs et al. 2000).

Elk are indigenous to the Columbia River basin but were not common before 1850. Market and subsistence hunting by Euro-Americans nearly exterminated elk by 1900 (Oliver et al. 1994c). Elk were reestablished by importing animals from Yellowstone National Park and Jackson Hole, Wyoming in 1911-1913, 1918, and 1930 (Bright and Powell 2008, Cliff 1939, Tucker 1940). Elk populations expanded quickly after being reintroduced (fig. 14).

As often happens, grazing impact was likely influenced by interactions with other factors. Heavy grazing by domestic and native ungulates apparently created ideal conditions for establishment of western juniper, other upland conifers, and shrubs. Fire exclusion then allowed them to persist on sites where they otherwise would have perished had the native disturbance regime (surface fire) been allowed to function properly (Young and Evans 1981).

5.1 Active Management Implications of Ungulate Herbivory

There are obvious interactions between fire and ungulate herbivory as ecosystem processes. Fire relied on herbaceous plant cover as an important spread component, while herbaceous plant communities relied on fire to suppress shrubs, juniper, and other woody vegetation whose shade and plant-suppressing chemicals (produced by allelopathy) would weaken and eventually kill the herbs. The effect of herbaceous cover on fire spread was recognized early on – high levels of domestic livestock were promoted as a fire protection measure because grazing would remove the fine fuels and thereby limit fire spread (Hatton 1920).

The restoration section (section 7) describes how it will not be possible to allow prescribed fire to substitute for free-ranging surface (wild) fire without careful and deliberate livestock grazing management to ensure fine-fuel continuity across dry-forest sites. And since prescribed fire occupies a primary position in the hierarchy of active management treatments considered for dry-forest restoration, grazing management must play a critical role in any effort to craft suitable habitat for restoring wildfire functioning with characteristic behavior and effects.

Herbs functioned as more than just a fine fuel component to help carry surface fire across dry sites – they also served to suppress tree regeneration (fig. 13). And since high tree density is a common problem throughout the interior Pacific Northwest (Powell et al. 2001), the importance of this inhibitory effect on tree establishment should not be overlooked.

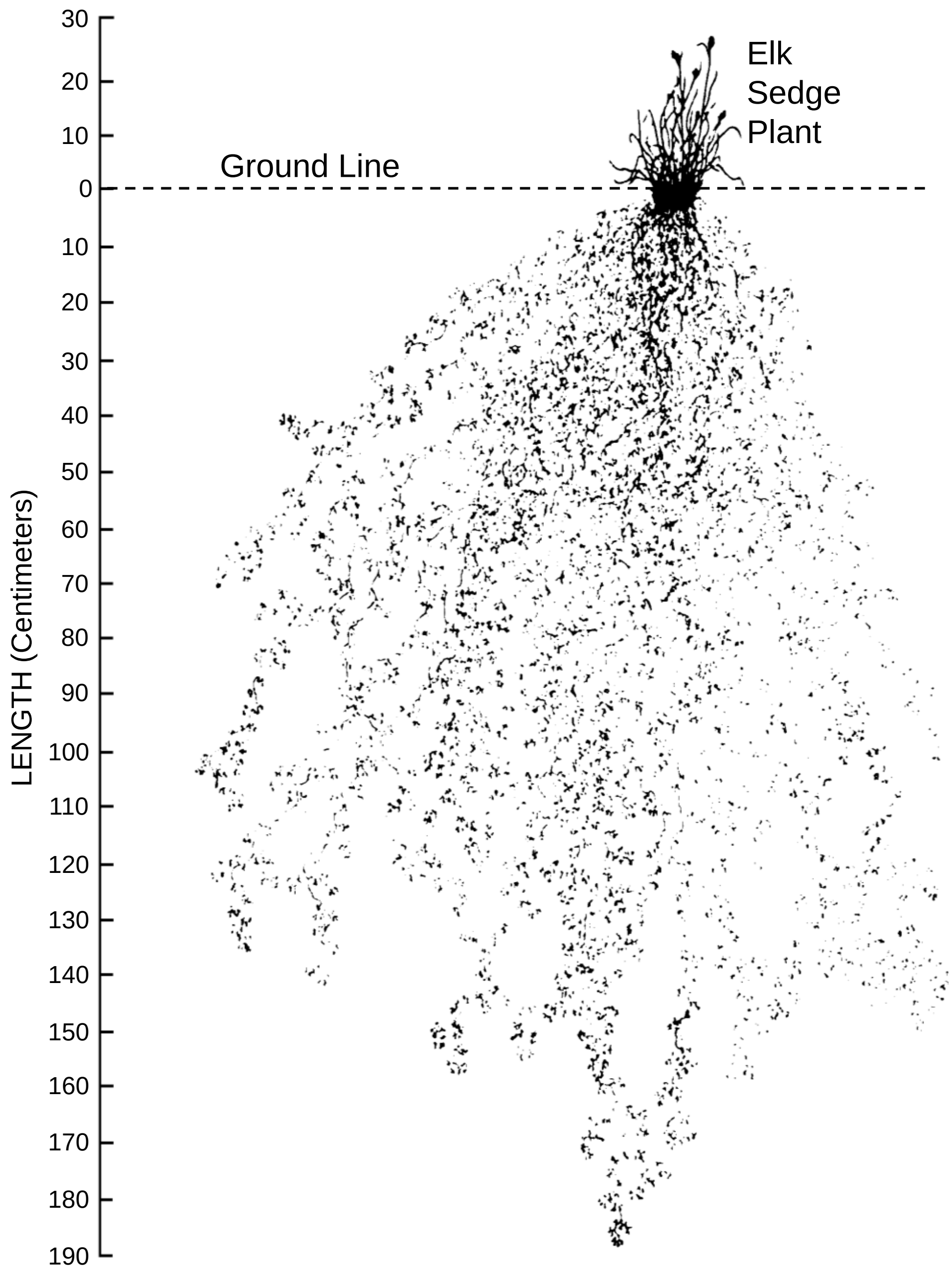


Figure 13 – Elk sedge has a fibrous root system occupying an impressive volume of soil (adapted from Sloan and Ryker 1986). The plant in this diagram is 12 inches tall and 10 inches wide, but its roots spread 56 inches wide and 75 inches deep; the dashed line shows ground level. Competition from the extensive root systems of bluebunch wheatgrass, Idaho fescue, elk sedge, and other perennial graminoids limits establishment of tree seedlings on dry sites (Cooper 1960, Munger 1917, Weaver 1967b). In some contexts, the inhibitory effect of rhizomatous herbs is viewed as a management problem because the herbs function as 'competing vegetation' by limiting the survival of planted tree seedlings. But in an ecological context, competition from graminoids and other herbaceous vegetation could easily be perceived as beneficial because it regulates seedling establishment on a biophysical environment (dry-forest sites) where large numbers of seedlings (and eventually mature trees) would easily exceed the area's capacity to support sustainable levels of tree stocking (Cochran et al. 1994, Powell 1999).

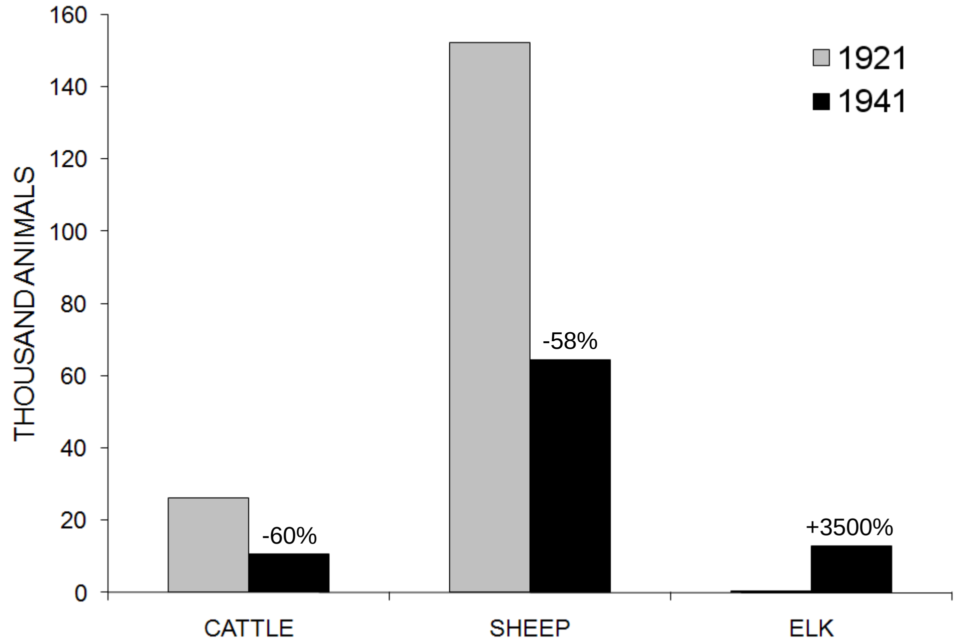


Figure 14 – Ungulate trends for the Whitman National Forest in the Blue Mountains of northeastern Oregon (data from Pickford and Reid 1943). This chart shows that cattle and sheep numbers declined dramatically between 1921 and 1941, and that elk numbers increased from only 360 animals in 1921 to more than 13,000 by 1941.

6. Influence Of Selective Cutting

Fire exclusion allowed a multi-layered structure to develop on a majority of dry-forest sites. After 40 or 50 years of fire exclusion, these areas often had an overstory of old-growth ponderosa pine and western larch, and an understory of Douglas-fir, grand fir, and occasionally, lodgepole pine. When wood products were harvested from these stands beginning in the late 1940s and the 1950s, many of the overstory trees were removed for these reasons (Powell 1994):

- The pine was usually old (often 200 years or more) and was adding little or no timber volume because of its slow growth. Since old pines often have low vigor and little resistance to insect attack, they were harvested before being attacked and killed by western pine beetle or mountain pine beetle (Cowlin et al. 1942, Keen 1936, Weidman and Silcox 1936).
- One reason for low vigor in old-growth pine trees was competition from a dense tree understory, and this understory would not have been present if the frequent surface fire regime had been allowed to continue its historical role.
- Old-growth ponderosa pine has a much higher selling value than associated species. Because of this economic advantage, harvesting ponderosa pine provided abundant Knutson-Vandenberg (K-V) receipts, which could then be used for noncommercial thinning, wildlife and range improvements, and other land management activities in the timber sale areas.
- As forestry intensified in the 1950s to meet increasing lumber demands after World War II (Fedkiw 1999, MacCleery 1992), dry mixed-conifer stands began to be managed. Mature

pines and larches were removed from the overstory, followed by a thinning in the immature understory of Douglas-fir and grand fir.

- An overstory removal strategy seemed to make good sense: it avoided the cost of tree planting, an expensive practice; it avoided the undesirable appearance associated with clearcutting; it maintained the pleasing aesthetics of a green, forested setting; and it capitalized on the previous growth of understory trees that had been there for 60 years or more.
- The understory trees (primarily Douglas-fir and grand fir) were viewed as a fast-growing gift of nature (i.e., not a result of intentional management), so why shouldn't they provide the next crop of timber products (Dezellem 1983)?

Some level of selective cutting has been occurring ever since Euro-American emigrants settled in the Blue Mountains (selective cutting is defined on page 52). The first commercial timber harvest in the northwestern pine region (eastern Oregon and eastern Washington) began about 1890 (Weidman and Silcox 1936), although some previous harvesting occurred in conjunction with mineral development (Lindgren 1901).

Mining activity in the lower Columbia River basin can trace its origins to the discovery of gold on Canal Gulch of Orofino Creek, a tributary of the Clearwater River, by Captain E.D. Pierce in 1860. In early spring of 1861, a miner from Pierce's party sold $800 worth of gold dust at Walla Walla and a stampede to the gold fields soon followed. By May of 1861, there were over a thousand miners in the Pierce City/Orofino area. Lewiston was founded in June 1861 and it quickly became an important center for resupplying the mines (Tucker 1940).

For the Blue Mountains, gold was discovered in Griffiths Gulch, located a few miles southwest of Baker City, Oregon, in the fall of 1861 (Lindgren 1901, Mosgrove 1980). Other discoveries soon followed, leading to a large influx of prospectors and miners in 1862. They established Auburn, Canyon City, Granite, Sumpter, Susanville, and other mining towns; by 1890, Baker, Union, and Grant counties already had a combined population of 23,900 (Lindgren 1901).

Within a year after gold was discovered in the John Day River valley (in June 1862 near Canyon City), a sawmill was operating to provide lumber for miners building flumes and sluiceways (Robbins 1997). Early cutting to supply mines and their adjacent settlements was substantial in localized areas; a turn-of-the-century map of Oregon's forests showed that much timber harvest had already occurred near Sumpter by 1900 (Gannett 1902, Thompson and Johnson 1900).

Since an extensive road network was not present in the Blue Mountains during the mining era, widespread timber harvests did not occur. A far ranging road system evolved in the Blue Mountains as wagon roads were developed for hauling wood and rails out to farms and ranches (Tucker 1940, Mosgrove 1980).

Early Euro-American settlements were often located in river bottoms containing forests of black cottonwood. Since cottonwood was not suitable for house logs or fence rails, settlers needed access to mountain timber. The favorite timber of early pioneers was tamarack or western larch (they called it 'tam-brack') because it was durable (decay resistant), the young trees furnished long, straight poles, and the large trees split easily into the finest rails that ever enclosed a pig pen or garden patch (Tucker 1940).

As emigrants settled in the fertile river valleys, they were accompanied by large herds of cattle and horses that freely roamed the adjoining foothills of bunchgrass. Once the settlers began growing grain and needed more timber to fence their fields to exclude free-range livestock, the road system was extended to access additional larch forest. Several roads in the northern Blue Mountains (Scoggins Ridge and Iron Spring-Clearwater, for example) were developed by 1870-1875 during this early Euro-American settlement era (Tucker 1940).

Later, some of these same roads were used to harvest timber for production of railroad ties. Although other species were also used, the resinous, durable woods of ponderosa pine and western larch were ideal for manufacturing railroad ties (Robbins and Wolf 1994, Tucker 1940).

Beginning in the early 1940s, national forest tree harvests increased to meet a heightened demand during World War II, and for new housing after the war (Fedkiw 1999). After World War II, ponderosa pine and other species were intensively harvested to feed a rapidly growing market for clear lumber for home construction, railroad ties, and to fabricate shipping crates for apples and other fruit crops (Bolsinger and Berger 1975, Gedney 1963).

Due to market conditions, early selective cuttings were typically a 'diameter-limit' harvest with the largest trees being removed (O'Hara et al. 2010). Diameter-limit cutting gradually altered the forest composition by removing the economically valuable trees (large-diameter ponderosa pines, western larches, and Douglas-firs), leaving behind a high proportion of small grand firs and Douglas-firs.

The following passage describes how partial cutting was applied in the early ponderosa pine forests of Oregon (Munger 1917).

> "The system of cutting which seems to be ideal for this type of forest is a form of selection cutting. Periodic cuttings are made, in each of which all the overmature and thoroughly ripe trees in the stand and all the defective ones are removed; and the saplings, poles, and young, thrifty trees are left standing to form the basis for the next crop.
>
> No tree is removed until it has reached its majority, so to speak, and no old, slow-growing tree is allowed to stand and occupy space which should be devoted to young and rapid-growing trees. It is customary to set an appropriate diameter limit of from 16 to 22 inches, the majority of the trees above which limit are cut, and those below left."

Why was diameter-limit cutting used if it favored low-value species (true firs) instead of the valuable ponderosa pine and western larch? Under the market (economic) conditions of that era, selective cutting was viewed as a wise use of natural resources because it captured the economic value of mature trees before they died, thereby initiating a rudimentary level of forest management (O'Hara et al. 2010).

With diameter-limit cutting, low-value trees were harvested to whatever extent allowed by prevailing market conditions. Many low-value species were left in the hope that some of them would become merchantable by the next silvicultural entry in 40-60 years. The following passage describes this situation for western white pine (Haig et al. 1941), but it was also true for ponderosa pine forests (Starker 1915).

> "The low values are due to high susceptibility to heart rot of western hemlock, grand fir, and some other species, and to the fact that the selling price of lumber manufactured from these species is often insufficient to meet production costs even if nothing were paid for the standing timber. Where trees of such species are not defective, the Forest Service policy has been to leave them uncut in the hope that at some future time they can be sold at a profit.
>
> But leaving these low-value species on areas that are cut over encourages their reproduction and tends to decrease the proportion of western white pine in the reproduction – an undesirable result both silviculturally and economically."

In many respects, selective cutting was the opposite of how native disturbance processes operated in dry mixed-conifer forests. Surface fire was the most influential disturbance process (Agee 1993, Cooper 1960, Munger 1917, Sloan 1998, White 1985, Wright and Agee 2004) and it discriminated against the fire-intolerant invaders (grand fir and Douglas-fir), while favoring the fire-tolerant trees with short, open crowns (ponderosa pine and western larch).

In contrast to surface fire, selective cutting removed the fire-resistant ponderosa pines and western larches, while allowing grand firs and other fire-susceptible species to remain on these dry-forest sites and then flourish (Filip 1994, Filip and Schmitt 1990).

Dry forests of the interior Pacific Northwest have had a history of high-grading (much of the early selective cutting was basically implemented as high-grading). High-grading did not seek to regulate stand structure; instead, harvesting simply removed timber. High-grading entries can be dysgenic and leave an inappropriate stand structure comprised of low-vigor trees susceptible to insect and disease attack (Carlson and Lotan 1988, Cochran 1998, Laudenslayer et al. 1989).

The late-seral species being favored by selective cutting had less value for timber products than ponderosa pine. Early Blue Mountains foresters recognized that partial cutting could have an undesirable impact on species composition and timber values, as described below.

> "White fir, though of slower height growth, is far more tolerant than bull pine, reproduces fairly freely, and under normal conditions would naturally supplant the pine in time. This condition has been greatly aggravated in the portions that have been lumbered by cutting the pine and leaving the white fir. The fir, often already on the ground under the pine, springs up, and pine reproduction is thus impossible" (Kent 1904).
>
> "In all sales on this Forest, care should be exercised in marking the timber not to leave the cutting area in such condition that a valuable stand be supplanted by inferior species. White fir, though occasionally used for fuel when no better species are available, makes poor fuel wood, while for saw timber it is all but valueless owing to the fact that nearly all mature trees are badly rotted by a prevalent polyporus, and the wood season-checks badly.
>
> Unless care is taken this species is prone to supplant such species as yellow pine and tamarack since it is much more tolerant of shade in early life" (Foster 1907).

6.1 Active Management Implications of Selective Cutting

Ponderosa pine, a keystone species for dry-forest ecosystems, was preferentially removed during historical timber harvest programs, particularly for the central and southern portions of the Blue Mountains where selective harvests were especially common (O'Hara et al. 2010). Not only did harvest of ponderosa pine result in removal of the tree species with highest resistance to disturbance processes, but the harvests were often conducted in such a way as to inadvertently favor other species with lower resilience to disturbance (e.g., Douglas-fir and grand fir).

Selective harvests also tended to remove the largest-diameter trees, so they functioned as an overstory removal by releasing the small seedlings and saplings in the understory. This means that selective harvests typically caused a pronounced change in vertical forest structure.

Active management treatments can be implemented as one component of a restoration program to help recover and then sustain ponderosa pine as a keystone tree species of dry-forest ecosystems (section 7 describes restoration options in more detail). In some situations, it may be necessary to first remove some of the ecologically inappropriate composition (grand fir and Douglas-fir) in order to liberate growing space for occupancy by ponderosa pine (including planting ponderosa pine, if need be, to help recover its historical abundance).

7. Restoration Of Dry-Forest Ecosystems

As a result of substantial reductions in park-like ponderosa pine forests throughout the interior Pacific Northwest, they are now considered to be a threatened ecosystem of the United States. Reed Noss and others described the loss of park-like ponderosa pine forest in their endangered ecosystems report: “conifer forests that depend on frequent fire, notably longleaf pine in the southeast and ponderosa pine in the west, have declined not only from logging but also from increases in tree density and from invasion by fire-sensitive species after fire suppression. These kinds of changes can cause the loss of a distinct ecosystem as surely as if the forest were clear-cut” (Noss et al. 1995).

Recurrent underburns are now extinct following a long-standing policy of fire exclusion (Stephens and Ruth 2005). Land managers responded to wildfire with Smokey Bear fire prevention campaigns, an arsenal of slurry bomber airplanes, mountaintop fire lookouts, aerial reconnaissance flights, radar-assisted lightning detectors, and crews of elite smokejumpers and specially trained, hotshot firefighters. In many respects, fire exclusion has been effective enough to be considered the most successful program in the U.S. Forest Service’s history (Fedkiw 1999).

Replacement of park-like ponderosa pine with mixed-conifer forest was caused by human alteration of a disturbance regime. Following at least 75 years of fire exclusion in the West, we now have millions of acres where the fire-resistant ponderosa pines are surrounded by shorter trees that grew to 40, 50 or even 75 feet tall, but only because they escaped fire when just three or four feet high (Arno and Allison-Bunnell 2003, Mutch et al. 1993, Powell 1994).

If man had not altered the disturbance regime on dry-forest sites by suppressing frequent surface fire, many of these younger trees would have perished while still small (Barrett 1988, Powell 1994, Sloan 1998, Steele et al. 1986). Now that smaller trees are present to function as ‘ladder fuel’, easily lifting surface fire up into the forest canopy, crown fires are more common than historically, leading to our contemporary perspective that crown fire, not timber harvest, is currently the greatest threat to old growth on dry sites (fig. 15).

Figure 15 – Crown fire in the Blue Mountains of northeastern Oregon (top photo from Powell 2010; bottom photo shows aftermath of crown fire at the 1996 Wheeler Point fire site on the Heppner Ranger District). In dense forests with large amounts of fuel, fires are very intense and travel rapidly from one tree crown to another. Crown fires were an important process for perpetuating lodgepole pine, grand fir, and subalpine fir forests, although any particular area seldom experienced a stand-initiating crown fire more often than once every 80 to 110 years (see table 3). Historically, crown fire was rare on dry-forest sites; that is no longer true following major changes in species composition, structure, and density over the last 100 years (Arno and Allison-Bunnell 2002).

7.1 Characterization of Reference Conditions

A restoration program benefits from a characterization of reference conditions. Reference conditions describe how vegetation has changed over time as a result of human influences and disturbance agents; they help us understand what an ecosystem is capable of, how disturbances function, and how ecosystems recover after disturbance (Falk 1990, REO 1995). They can also help us understand how we got where we are now, and to decide where we want to be in the future (Gruell 2001).

Compiling collaborative historic evidence from photographs, aerial photography, maps, reports, and other historical sources is used to derive reference conditions (Bright and Powell 2008, Egan and Howell 2001, Evans 1991). As Don Falk (1990) described it: "restoration uses the past not as a goal but as a reference point for the future. If we seek to recreate the temperate forests, tallgrass savannas or desert communities of centuries past, it is not to turn back the evolutionary clock but to set it ticking again."

Seven decades ago, for example, 74% of the commercial forest in eastern Oregon and eastern Washington was classified as ponderosa pine, much of which was old-growth (Cowlin et al. 1942). By the late 1970s, it was believed that at least 25% of the ponderosa pine type in the Pacific Northwest had been replaced with dry mixed-conifer forest (Barrett 1979); the reduction was apparently greater for northeastern Oregon where ponderosa pine declined by more than 50% between 1936 and 1980 (fig. 16; Powell 1994).

These forest inventory trends conclusively demonstrate that dry mixed-conifer forest, frequently overstocked with Douglas-firs and true firs tolerant of overcrowding, have replaced ponderosa pine and now cover much of the eastside landscape (in a potential vegetation context, 'dry mixed-conifer' and 'dry forest' are synonymous terms) (Mason and Wickman 1994).

The following comments, taken from a report summarizing results of the 1950s forest inventories for eastern Oregon counties (Gedney 1963), suggest that a trend of ponderosa pine being replaced by other species was recognized long ago.

> "If present trends continue, the proportion of ponderosa pine will be less in the future than at present. In 29 percent of all the pine sawtimber types, there is no understory of pine, only other species – Douglas-fir, white fir, and lodgepole pine. In another 27 percent of the pine sawtimber stands, the understory is a mixture of young ponderosa pine and other species. On more than half of this area, species other than pine predominate. Unless something happens to change this relationship, or unless more intensive forest management is undertaken, about 40 percent of the pine sawtimber type is likely to shift to some other type."

7.2 Forest Health Considerations

Forest health. The perceived condition of a forest based on concerns about such factors as its age, structure, composition, function, vigor, presence of unusual levels of insects or disease, and resilience to disturbance. Note that perception and interpretation of forest health is influenced by individual and cultural viewpoints, land management objectives, spatial and temporal scales, the relative health of stands comprising the forest, and the appearance of a forest at any particular point in time (Helms 1998).

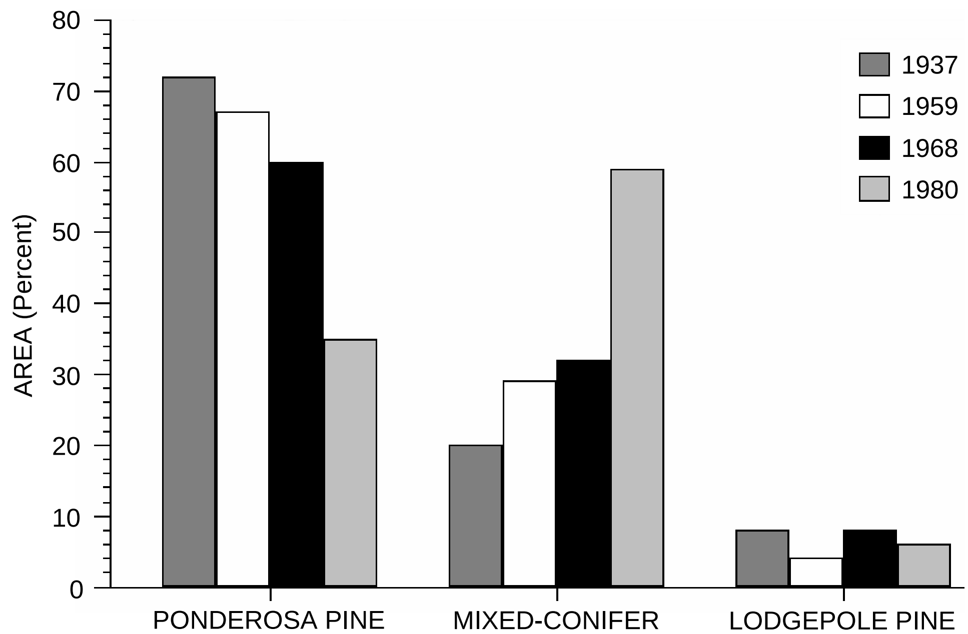


Figure 16 – Change in forest cover types for the Malheur National Forest, 1937-1980 (from Powell 1994). Ponderosa pine forest declined by more than half between 1937 and 1980, the mixed-conifer type increased by an equivalent amount during this period, and the lodgepole pine type remained relatively constant. This figure shows that mixed-conifer forest – prime habitat for defoliating insects – increased by 195% between 1937 and 1980. The increase in mixed-conifer forest was an important reason for the unprecedented magnitude of a Blue Mountains budworm outbreak between 1980 and 1992.

Altered disturbance regimes often result in forest health problems such as stand-initiating fires or insect outbreaks (figs. 7 and 15), but the conditions causing these problems take decades or centuries to develop. An example of altered disturbance regimes is provided by a recent U.S. Fish and Wildlife Service analysis of 146 threatened, endangered, or rare plant species for which credible fire effects information was available. It found that 135 of these plants (92%) either benefit from fire or are found in fire-adapted ecosystems, suggesting that a decline in their abundance or persistence may be attributable to fire exclusion (Hessl and Spackman 1995).

Plant succession, in combination with human influence and weather extremes or climate change, is a recipe for forest health issues; insect outbreaks and disease epidemics may be little more than symptoms of an underlying problem (Shlisky 1994, Sloan 1998, Steele 1994). Forest ecosystems must adjust to altered disturbance regimes with the only tools available – insects, diseases, wildfire, and to a limited extent, microbial decomposition (Harvey 1994; fig. 8). In this respect, forest health functions as a unifying concept because it integrates the effects of forest succession, tree physiology, and insect and disease susceptibility (Clark et al. 1998).

Once a forest's vigor falls to low levels, insects and diseases can quickly become catalysts of change (Gast et al. 1991, Wickman 1992, and many other citations in the References section). Without application of needed restoration treatments in the near future (15-30 years), there is a high probability that the Blue Mountains' legacy into the next century will be large, homogenous landscapes recovering from conflagration wildfires and other ecosystem setbacks on a scale unprecedented in recent evolutionary history (Sampson et al. 1994).

Landscape-scale changes have occurred to such an extent that simply reintroducing native disturbance processes (wide-ranging surface fire, for example) may produce effects outside of any historical precedent. These effects are undesirable because they would move the ecosystem farther away from, rather than closer to, the desired future condition (Landres et al. 1999). In situations where current conditions deviate significantly from reference (historical) conditions, some type of restoration treatment (such as reducing tree biomass or herbivore populations) may be needed before a disturbance process can be successfully reintroduced (Aplet and Keeton 1999, Case and Kauffman 1997, Pickett and Parker 1994).

One example of this concept is that standing and surface fuels have often accumulated to an extent where prescribed fire cannot be applied safely unless preceded by a mechanical treatment such as thinning (Arno et al. 1995, Feeney et al. 1998, Fiedler et al. 1996, Fiedler et al. 1999, Graham et al. 1999). This caution about reintroducing fire is appropriate because fire exclusion alone did not create our current problem, and fire's reinstatement will not cure it. Fire is an ecological catalyst that takes its character from whatever surrounds it. Ecosystems with uncharacteristic ('unhealthy') conditions will yield uncharacteristic fires (see fig. 15).

To successfully reinstate fire, we first need to restore suitable habitat for desirable fire regimes. The woods need to be thinned before reintroducing wildland fire, but it's not just the trees that matter, it's also the grass. Without careful and deliberate grazing management to ensure fine-fuel continuity, it may be difficult to reestablish the short-interval fire regime on dry sites (Madany and West 1983, Pyne 1997, Rummell 1951).

Exclusion of low-severity fires and selective harvesting of large, old trees have generally homogenized the eastside landscape, especially for the montane, mixed-conifer zone at middle elevations (Hessburg et al. 1994, 1999; Lehmkuhl et al. 1994). In drier forests of eastern Oregon and eastern Washington, alteration of the disturbance regime by suppressing fire has been defragmenting the inherent patterns of fuel distribution and accumulation, thereby increasing the potential for large wildfires (Hessburg et al. 2005, Rochelle et al. 1999) (see fig. 15).

Unnaturally large, contiguous areas of densely stocked and highly stressed trees provide an increased food base for defoliating insects (Mason and Wickman 1988, Gast et al. 1991, Hessburg et al. 1994), and they are also more favorable for the occurrence of parasitic plants (Zimmerman and Laven 1984, Gast et al. 1991) and fungal pathogens (Filip and Schmitt 1990). Historically, defoliating insects and bark beetles tended to affect only small patches of forest, but such insects now occupy large, landscape-scale areas at episodic outbreak levels (Hessburg et al. 1994, Powell 1994, Wickman 1994).

Reducing stand density to minimize moisture and nutrient stress for individual trees, and then reintroducing fire – the natural thinning agent – are primary objectives of restoration management, but these activities are controversial to some publics (Agee 1994, Arno and Ottmar 1994). Scientists emphasize that restoration efforts must be focused at the landscape scale to reestablish a mosaic of forest types and structural stages that will, in turn, reduce the continuity of food sources for defoliating insects (Mason and Wickman 1994, Torgersen 2001), while also crafting habitat for free-ranging wildfire (Arno and Ottmar 1994).

7.3 Ecosystems Out of Balance

How did fire exclusion, in combination with selective tree harvest and ungulate herbivory, contribute to dry-forest ecosystems that are now out of balance? These ecosystem alterations had a detrimental impact on ecological integrity by modifying vegetation diversity and complexity, particularly at a landscape scale, resulting in forests at risk.

The forests most at risk are those under the most stress because they contain too many trees, or too many of the wrong kind of trees, to continue to thrive. As these forests get older and denser, the competition between trees intensifies, stress increases, and the probability of uncharacteristic (catastrophic) change goes up dramatically (Sampson et al. 1994, Sloan 1998).

Over-protection from fire can render a forest susceptible to serious soil damage when a fire eventually occurs (Grier 1975). When historical wildfire regimes have been altered because society is not prepared to accept fire-related risks to life and property, then land managers should attempt to design thinnings and other silvicultural treatments emulating the desirable characteristics of presettlement fire regimes (Kimmins 1997).

Historically, spatial variation in fire intensity was important for providing diversity in landscape patterns (fig. 17). [Munger (1917) provides excellent observations about the spatial pattern associated with pine forests; see pages 11-12.] Under the recent fire management paradigm (fire exclusion), the influence of fire as an ecological process has been dramatically reduced, resulting in more homogeneous landscape patterns than would have existed historically (Hessburg et al. 1999b, 2005; Lehmkuhl et al. 1994; del Moral 1972; Taylor and Skinner 1998).

The fire exclusion strategy "may lead to tree population explosions and dead fuel accumulation to such an extent that catastrophic adjustments become inevitable. Eventually, catastrophic disturbances such as insect and disease attack and crown fire may cause extensive mortality at a scale never before experienced by the community of organisms" (Covington et al. 1994a).

7.4 Emulating Disturbance Processes

A primary focus of dry-forest restoration is to use silvicultural treatments to emulate the intensity, scale, and pattern of historical disturbance regimes. The objective of an active restoration approach is to address fire hazard and insect and disease problems; production of timber, water, and other commodities (if any) is nothing but a by-product of meeting the restoration objectives (DeGraaf and Healy 1993).

The choice of silvicultural treatment can be important in both an ecological and an economic context. For example, a general trend over the last decade has seen a transition from forest treatments producing relatively large, high-quality timber to those generating small, low-value material at a high production cost (Fiedler et al. 1999, Larson and Mirth 1998, LeVan-Green and Livingston 2003). This trend has obvious implications on the economic viability of using commodity revenues to offset the costs of dry-forest restoration treatments (Rainville et al. 2008).

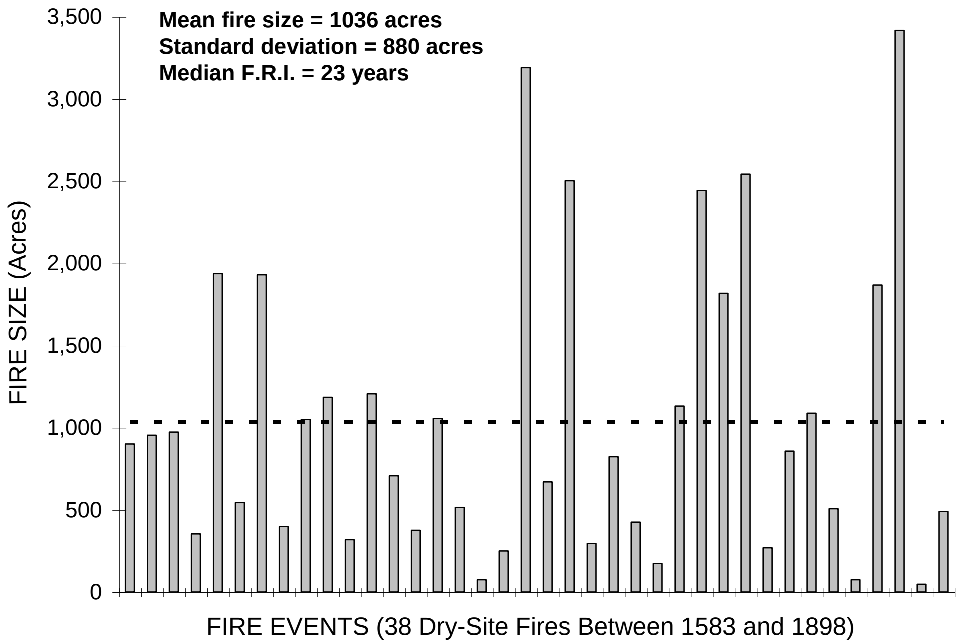


Figure 17 – Spatial variability in fire extent for dry-forest sites in the Tucannon watershed, northern Blue Mountains, southeastern Washington (based on data from Heyerdahl 1997). Forty-two individual fire events were interpreted for the watershed, and 38 of them occurred on dry-forest sites. The smallest fire extent on dry-forest sites was 47 acres and the largest was 3,417 acres. Average fire extent for the 38 dry-site fires was 1,036 acres (the dashed line shows the average). Note that the last recorded fire for this watershed occurred in 1898 (Heyerdahl 1997).

Current ecological conditions in dry forests of the interior Pacific Northwest suggest that immediate management action is warranted (Bonnicksen 2000b). This management intervention needs to be intensive and to cover wide areas of the landscape, but to be effective it must be substantially different in both impact and appearance from what was done historically (Sampson et al. 1994). This means that management intervention should use an adaptive approach that considers the forest as a fully functioning ecosystem (Hunter 1999, Rowe 1992).

An eminent group of fire ecologists cautioned that a status quo solution for the Blue Mountains "will leave us with seriously degraded ecosystems offering little value in an ecological, aesthetic or economic sense. This option goes counter to the values and concerns of society today, such as biological diversity, beautiful and 'natural' landscapes, healthy plant and animal communities, and long-term productivity" (Mutch et al. 1993). "Restoration efforts will require that we discard the misconception that nature is unchanging and accept the reality that people need to be actively involved in managing forests and woodlands for sustained values" (Gruell 2001).

If the scale of tree harvest does not emulate the scale of native disturbance processes, then we can expect ecosystem changes such as reduced biological diversity and impaired nutrient cycling (Baydack et al. 1999, Eng 1998). Using a variety of cutting patterns, for example, is important to avoid uniform landscapes; grouping cut blocks reduces the total amount of edge, minimizes fragmentation, and maintains larger patches of interior forest habitat.

Society's response to deteriorated dry-forest conditions in the interior Pacific Northwest has lacked consensus. Some stakeholders advocate a passive approach, believing that active intervention would make an unfortunate situation even worse (Beschta et al. 2004). Many of the proponents of passive restoration contend that our knowledge of historical (reference) conditions will never be complete, so we should rely on wildfire, insect outbreaks, and other disturbance processes to transform forest composition and structure (Frank 2003, Stephenson 1999).

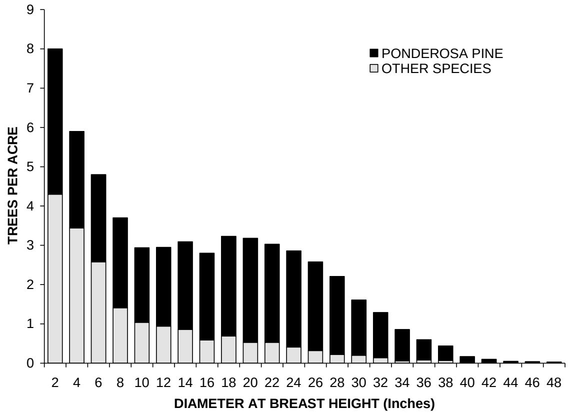
"The present vulnerability of these forest ecosystems requires that we temper our need for more complete information with an urgency created by the current risk of crown fires" (Allen et al. 2002). As an example, all of the causal mechanisms are not understood, but it is clear that when plant succession occurs on dry-forest sites in the absence of frequent wildfires, it will result in reduced availability of mineral nitrogen and cause increased accumulation of allelopathic compounds in the mineral soil (MacKenzie et al. 2006, Souto et al. 2000, Wardle et al. 1998).

7.5 Desired Conditions for Dry-Forest Sites

Desired conditions contributing to a sustainable composition and structure for dry-forest sites include these attributes (Fiedler 2000b):

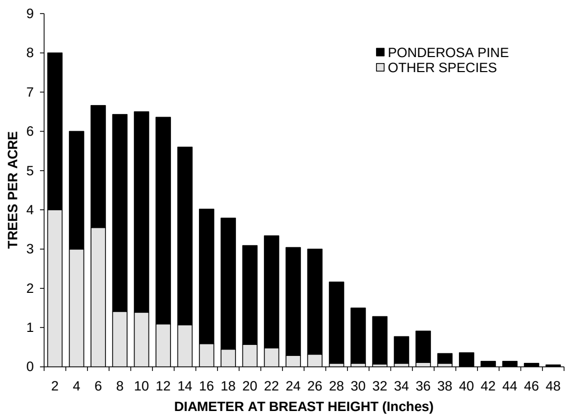
- A moderately open stand density (40 to 70 square feet per acre of basal area). Stands with a predominance of big trees (> 21" dbh) could be toward the upper end of this stocking range.
- A multi-cohort or uneven-aged structure at the stand level, although discrete groups in a stand generally consist of a single cohort (even-aged groups in an uneven-aged stand). Up to 70 percent of the even-aged groups in an uneven-aged stand structure would have a single-layer structure (fig. 33 later in this paper illustrates the groupy or clumpy structure).
- A predominance of large trees – up to 60 percent of the basal area per acre would occur in trees whose diameter at breast height was 21 inches or greater.
- A species composition dominated by ponderosa pine – up to 70% would consist of ponderosa pine. At least ⅔ of the species composition should consist of early-seral, shade-intolerant species to minimize spruce budworm susceptibility (Carlson and Wulf 1989).
- Coarse woody debris (CWD) levels ranging between 5 and 20 tons per acre (Brown et al. 2003). Note that coarse woody debris is typically defined as dead standing and downed pieces larger than 3 inches in diameter (Harmon et al. 1986). Between 4 and 7 tons per acre of the 5-20 ton per acre CWD range would exist as standing snags at a total rate of 6 to 14 stems per acre (2 to 4 of the snags per acre would be at least 15" dbh) (Harrod et al. 1998).

These desired conditions acknowledge that to bring tree density (basal area) back into the historical range of variation (RV), management activities should emphasize producing fewer but larger trees (Allen et al. 2002, Wright and Agee 2004). [Section 7.13 provides detailed RV information for composition, structure, and density.] Numerical goals relating to a desired future condition depend on how the metric is quantified. Inventory data collected in 1910-1911 for three forest tracts in the Blue Mountains (fig. 18), for example, showed that when tree density was expressed as basal area, 66% of it occurred in trees whose diameter was 21 inches or more. When forest density was expressed as trees rather than basal area, stems with a diameter of 21 inches or more comprised only 23% of total stocking (Bright 1912; Munger 1912, 1917).



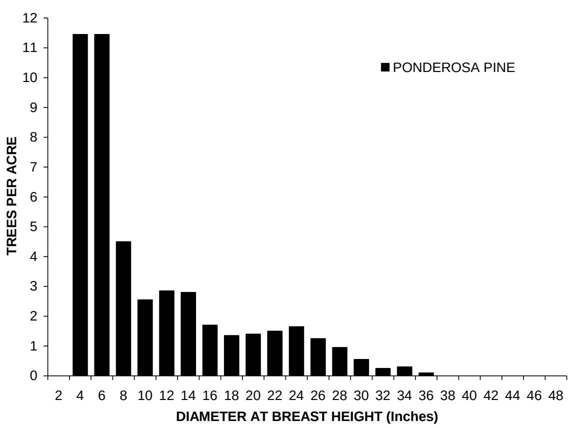
AUSTIN-WHITNEY TRACT

Live trees per acre (1″+ DBH):	56.5
Ponderosa Pine Proportion	67%
Live basal area; square feet/acre (1″+)	94.7
Ponderosa Pine Proportion	84%
Quadratic mean diameter (Inches)	17.5
Ponderosa Pine QMD (Inches)	19.5
Stand density index	139.5
Ponderosa Pine Proportion	85%
Live trees per acre > 21″ in diameter	15.9
Ponderosa Pine Proportion	87%
Live basal area per acre > 21″ DBH	68.0
Ponderosa Pine Proportion	87%



LOOKINGGLASS CREEK TRACT

Live trees per acre (1″+ DBH):	73.6
Ponderosa Pine Proportion	75%
Live basal area; square feet/acre (1″+)	112.6
Ponderosa Pine Proportion	88%
Quadratic mean diameter (Inches)	16.8
Ponderosa Pine QMD (Inches)	18.2
Stand density index	169.1
Ponderosa Pine Proportion	88%
Live trees per acre > 21″ in diameter	17.1
Ponderosa Pine Proportion	90%
Live basal area per acre > 21″ DBH	74.8
Ponderosa Pine Proportion	91%



WINLOCK'S MILL TRACT

Live trees per acre (1″+ DBH):	46.6
Ponderosa Pine Proportion	100%
Live basal area; square feet/acre (1″+)	43.8
Ponderosa Pine Proportion	100%
Quadratic mean diameter (Inches)	13.1
Ponderosa Pine QMD (Inches)	13.1
Stand density index	69.9
Ponderosa Pine Proportion	100%
Live trees per acre > 21″ in diameter	6.6
Ponderosa Pine Proportion	100%
Live basal area per acre > 21″ DBH	24.5
Ponderosa Pine Proportion	100%

Figure 18 – Selected stand attributes for three forested tracts in the Blue Mountains of northeastern Oregon (adapted from Powell 1999). This data was derived from relatively large sample areas measured in 1910 or 1911 (sampled areas were 258½ acres for Austin-Whitney, 44 acres for Lookingglass Creek, and 20 acres for Winlock's Mill). Data sources were Bright (1912) and Munger (1912, 1917).

It is also important that any characterization of desired conditions account for a range of disturbance processes and the function of biological legacies, rather than attempting to directly replicate one particular disturbance agent (Foster et al. 1998, Hansen et al. 1991, Urban et al. 1987). Moreover, land managers should focus their attention "on the rates at which changes occur, understanding that certain rates of change are characteristic, desirable and acceptable, whereas others are not" (Botkin 1990).

Alan White (1985) suggested that ponderosa pine regeneration requires a safe site such as the ash bed of a fire-consumed log, where at least a few tree seedlings could get established before herbaceous and woody fuels recovered enough to support another fire. Although frequent surface fire caused overall seedling survival to be low, long-term survival of saplings that successfully made it through this initial ‘fire filter’ was high. This fire regime produced a low density of small-diameter ponderosa pine trees, and the resulting diameter distribution was relatively flat. This differs from the classical, inverse-J distribution expected for uneven-aged forests on moist sites (Mast et al. 1999, White 1985).

Munger (1917) also observed that “yellow pine grows commonly in many-aged stands” (see page 12). Historical inventory data for the Blue Mountains (fig. 18) exhibits the flat diameter distribution expected for uneven-aged stands maintained by frequent surface fires on dry sites.

Adopting a very conservative approach to restoration of dry forests is not a choice of “no action” because such a strategy accepts the risk of high-severity wildfire and other uncharacteristic disturbance events. Upon recognizing that the risks of no action are probably unacceptable for most scenarios, managers must design flexible, adaptive treatments to restore more natural conditions (e.g., more historically appropriate conditions), including high levels of spatial heterogeneity for dry-forest sites (Allen et al. 2002, Wright and Agee 2004, and others).

The solution to forest health problems could begin with thinnings to reduce tree density in overcrowded forests, particularly for dry-forest sites where over-crowding was a rare phenomenon before the onset of fire exclusion, selective cutting, and intense livestock grazing (these three activities contributed to creating the condition class 2 and 3 conditions described and illustrated in table 4) (Belsky and Blumenthal 1997; Covington and Moore 1994a, 1994b; Madany and West 1983; Oliver et al. 1994c; Rummell 1951).

A simulation study examined changes in fire risk associated with active restoration treatments. It found that fire risk at a landscape scale decreased steadily as management intensity increased. After five decades, the no-treatment scenario had nearly 30 percent of the landscape in a high-risk category, whereas active management (thinning and prescribed fire) had 100 percent of the landscape in a low-risk category (Wilson and Baker 1998).

No single restoration solution, however, can hope to precisely reproduce the inherent variability of a dry-forest landscape because ecosystems are shaped by a wide variety of disturbance types, frequencies, and intensities (Voller and Harrison 1998). Deciding to take immediate remedial action can result in a philosophical shift toward proactive management to curtail excessive fire and insect impacts, and a shift away from reactive management in response to landscape-scale disturbance events (Covington 2003).

The challenge is to integrate a suite of active management treatments that effectively and appropriately emulate the natural disturbance regime of dry-forest landscapes (fig. 19). Successfully meeting this challenge will produce a semblance of historical forest structure and species composition – a desirable outcome not because the resulting condition is historical, but because it is sustainable (e.g., vigorous, self-perpetuating, pine-dominated, and at low risk to stand-replacing fire and defoliating insects) (Fiedler 2000).

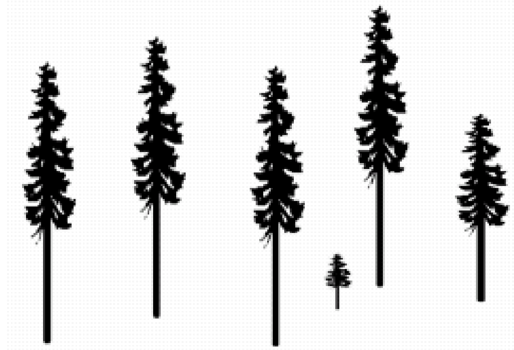
Using burlap to beat out a surface fire in ponderosa pine, Wallowa National Forest, about 1910. As Thornton Munger noted, "Light, slowly spreading fires that form a blaze not more than 2 or 3 feet high and that burn chiefly the dry grass, needles, and underbrush start freely in yellow pine forests. Practically every acre of virgin yellow pine timberland in central and eastern Oregon has been run over by fire during the lifetime of the present forest" (Munger 1917).

Dense ponderosa pine forests developed after the influence of fire was suppressed during the past 100 years. On many dry sites, fire exclusion had the unintended consequence of allowing late-seral tree species (grand fir, white fir, and interior Douglas-fir), none of which are adapted to a recurrent fire regime as small or mid-sized trees, to replace the ponderosa pines.

Thinning and prescribed fire can be used in tandem to restore sustainable and resilient forests on dry sites. Changing a dense forest condition (middle frame) to one that more closely approximates the historical composition and structure will go a long way toward allowing us to restore the native disturbance regime – frequent surface fires.

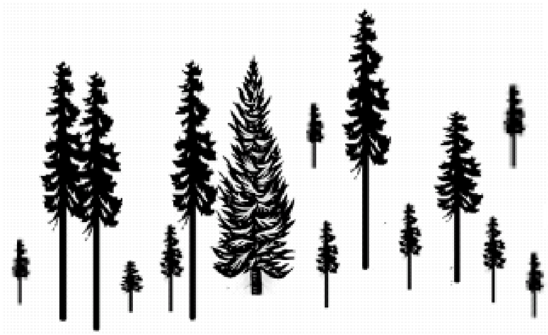
Figure 19 – Restoration of ponderosa pine ecosystems (from Powell et al. 2001; top photograph taken from Boerker 1920, bottom two photographs from USDA Forest Service 2001). Almost immediately after its inception in 1905, the Forest Service began suppressing wildfire on national forest system lands (Fedkiw 1999) (top). By removing surface fire as a thinning agent, fire exclusion caused tree density to increase substantially on dry sites (middle). Restoration of dry forests features thinning or another mechanical treatment to reduce tree density, followed by prescribed fire for nutrient cycling and to reestablish fire as an important ecosystem process (bottom) (Arno and Allison-Bunnell 2002, Arno et al. 1995, Fiedler et al. 1996, Fiedler et al. 1999). Note that dry-forest landscapes (top) are said to have strong 'ecological memory' due to the strength of interactions between ecological processes (surface fire) and landscape pattern. "When ecological memory is strong, landscape pattern is persistent; pattern tends to be maintained rather than destroyed by fire" (Peterson 2002).

Table 4: Fire regime condition classes for dry forests (Brown et al. 2003, GAO 2004, Schmidt et al. 2002, and Zimmerman 2003).



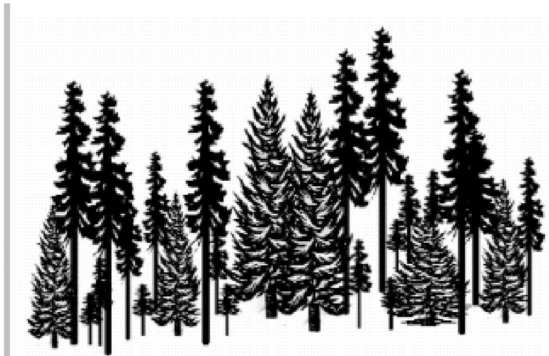
CONDITION CLASS 1
(ecosystem maintenance stage)

Composition and structure: open, park-like, mature ponderosa pine stands; even-aged clumps occurring as an uneven-aged structure; single-layer canopy structure.
Tree density: stocking levels are within the historical range; density remains consistently below the lower limit of the self-thinning zone.
Vigor: high seasonal energy activity; high capacity to repel or resist disturbance agents such as insects and pathogens.
Fire regime: maintained within or near the historical range; no departure from historical frequency or severity (nonlethal regime).
Fuel dynamics: surface and total fuel loads maintained at historical levels (between 5 and 10 tons per acre).
Resilience and risk: high capacity to remain fully functional following fire; low risk of losing key ecosystem components after fire.



CONDITION CLASS 2
(ecosystem alteration stage)

Composition and structure: beginning to depart from the reference conditions; lack of fire allows establishment of fire-sensitive species and a multi-layer canopy structure.
Tree density: stocking levels in upper half of the historical range; density may exceed the lower limit of the self-thinning zone.
Vigor: moderate to high seasonal energy activity; somewhat diminished capacity to repel or resist insect or pathogen attack.
Fire regime: frequency reduced and departing from historical range; severity increased with some mortality of overstory trees.
Fuel dynamics: surface and total fuel loads in the upper half of the historical range (10 to 20 tons per acre).
Resilience and risk: fairly high potential to return to condition class 1 using prescribed fire; moderate risk of losing key ecosystem components following wildfire.



CONDITION CLASS 3
(ecosystem degradation stage)

Composition and structure: highly altered from the reference conditions; fire-sensitive species common; open, park-like appearance completely lacking; multi-layer canopy structure.
Tree density: stocking levels exceed the historical range; total tree density may be 3-4 times greater than for condition class 1.
Vigor: little fluctuation in seasonal energy activity; greatly increased susceptibility to insect or pathogen attack.
Fire regime: dramatic departure from historical frequency and severity; many fire return intervals missed; larger average fire (patch) size.
Fuel dynamics: surface and total fuel loads outside historical range (> 20 tons per acre); increased fuel continuity at landscape scale.
Resilience and risk: low potential to return to condition class 1 using prescribed fire; mechanical treatments needed before reintroducing fire; high risk of losing key ecosystem components to stand-replacing wildfire.

7.6 Thinning and Prescribed Fire as Restoration Treatments

Restoration. Restoration refers to holistic actions taken to modify an ecosystem to achieve desired conditions and functions, including the process of returning ecosystems to a properly functioning structure, species composition, and stand density (Dunster and Dunster 1996). Two restoration approaches have been recognized: (1) Active restoration: an approach involving implementation of management activities (prescribed fire, thinning, etc.) to restore appropriate conditions; and (2) Passive restoration: an approach involving removal of stresses that caused ecosystem degradation in the first place, such as cessation of fire exclusion in fire-dependent ecosystems (Rapp 2002).

Although it is not expected that park-like ponderosa pine forest can be fully restored to its historical abundance, some amount of thinning and prescribed fire, applied in proper places and at appropriate times, is needed to help recover the integrity and resilience of this important ecosystem (Agee 1997, Arno and Allison-Bunnell 2002, Covington 2000, Fiedler et al. 2001).

Thinning and prescribed fire, used alone or in combination, are two active restoration treatments that can compensate somewhat for suppression of the historical surface fire regime because they are useful for reducing high stand density levels, addressing successional advancement (from early- to late-seral tree species), and to jump-start stagnant nutrient-cycling processes (Gundale et al. 2005) (fig. 20).

We need to remember, however, that some species find optimum habitat in early-seral conditions, others in late-seral plant communities, and some in either situation. So when compared with the historical situation, significant changes in disturbance levels (either an increase or decrease) can ultimately degrade biodiversity by affecting the proportion and distribution of seral stages at a landscape scale (White et al. 1999).

However, fire can be highly stressful to old-growth ponderosa pines, particularly on sites where existing tree density is many times greater than presettlement stocking levels. In these uncharacteristically crowded forests containing low-vigor trees, it may be wise to thin first and allow the old-growth pines to recover or release before subjecting them to the additional stress of a prescribed fire (Covington 2003, Fiedler et al. 1996, Scott 1998, Swezy and Agee 1991).

Much of the byproduct from fuel-reduction thinnings will be too small or poor in quality to be commercially valuable for conventional wood products (Fiedler et al. 1999). These thinnings are typically accomplished by using a service contract where the contractor is paid a specified amount per acre, or per tree, to cut or otherwise treat the unwanted trees and leave them on-site (Powell et al. 2001).

But leaving unwanted vegetation on-site contributes to an immediate and often unacceptable increase in short-term fuel loading and fire risk (Arno and Allison-Bunnell 2002, Brown et al. 2003, Mutch et al. 1993). The ideal solution, albeit a costly one in an economic context, is to use stewardship contracting for the vegetation treatments, and then remove the resulting fuel to an off-site biomass facility for ultimate disposal (the fuel could also be treated using pyrolysis to create bio-oil for energy, and biochar for carbon sequestration) (Lehmann and Joseph 2009).

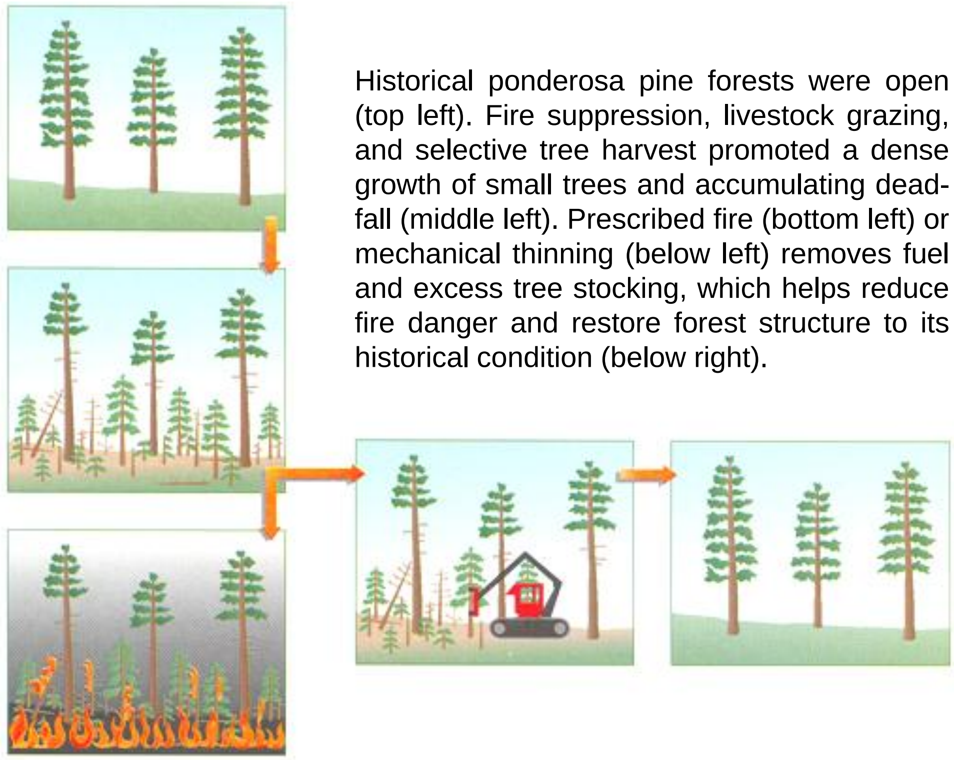


Figure 20 – Correcting a history of fire exclusion (adapted from Phillips 1995). Mechanical thinning and prescribed fire are examples of stand-maintaining disturbances that kill from the bottom up (Smith et al. 1997). Application of both these treatments needs to increase as one way to address forest health issues resulting from changes in species composition and forest structure caused by fire exclusion, livestock grazing, and selective tree harvest.

Several efforts are underway around the western United States to develop processing methods and markets for ever-smaller material. If these efforts succeed, then future thinnings may eventually become commercial by producing biomass material for distillation of ethanol (a gasoline additive) from cellulose, or to generate electricity (Barbour and Skog 1997).

On forest sites in eastern Washington, residual trees increased growth following surface fires that killed trees in the intermediate and suppressed crown classes, but growth increases were greater when thinning was used to reduce stand density. Unlike fire, manual thinning did not damage the fine roots, so residual trees reoccupied the growing space quickly. After overstory trees claimed the additional growing space provided by the thinning, grasses did not readily invade the site (Oliver and Larson 1996).

Avoiding root damage is important, particularly on dry, rocky, low-productivity sites. For poor-quality sites, up to 40% of a tree's annual net production is invested in fine roots (Keyes and Grier 1981). Since fine roots are concentrated near the soil surface, this means that the heat produced by prescribed fires has the potential to damage or kill them.

For spring prescribed fires, heat effects could "be amplified by the high thermal diffusivity of moist soils, and the low soil temperatures to which roots are adapted after winter. Late summer or early fall fires, on the other hand, occur when roots are inactive, soils are dry and thus good insulators, and roots are adapted to higher soil temperatures" (Grier 1989).

Thinning and prescribed fire also have site disturbance differences. Thinning tends to have minimal soil disturbance, so it favors native understory species more than exotics (non-natives). A combination thinning and burning treatment has intermediate amounts of soil disturbance, and this option favors native and exotic species equally. Burning tends to increase exotic species with little effect (either favorable or unfavorable) on native understory species (Fiedler et al. 2006, Griffis et al. 2001, Kerns et al. 2006).

7.7 Stewardship Tree Harvest

Stewardship tree harvest, depending on the techniques used and the woody debris left behind, may reduce fuels and wildfire hazard in the short term, or it may not. Harvest alone, without also thinning the small unmerchantable trees, treating the woody debris produced by the harvest and thinning, and then using prescribed fire, seldom reduces wildfire hazard over the long term (Gruell 2001).

Fuel hazard studies often came to similar conclusions regarding the importance of treating post-treatment woody debris (slash). 'Lopping and scattering' is a common treatment for thinning slash. In this method, branches are cut from the felled trees and scattered to reduce fuel concentrations; if needed, slash is pulled away from residual green trees. Research found that "lopping and scattering still managed to reduce fire behavior levels (mainly because of fuel depth reduction), but application of this treatment should be limited to areas with light fuel accumulations – less than 9 tons per acre" (Kalabokidis and Omi 1998).

Treating or removing post-harvest woody debris provides definite physiological advantages if a wildfire occurs soon after the treatment: when fire occurred in a thinned stand in Arizona, with the woody debris having been removed prior to the fire, fire improved resin production as compared to the unthinned control (Feeney et al. 1998), and improved resin production promotes defensive chemical compounds enhancing bark beetle resistance (Kolb et al. 1998).

7.8 More Use of Prescribed Fire?

In the early 1990s, Bob Mutch and other fire scientists recommended that prescribed fire use (fig. 21) be increased tenfold as one way to address forest health concerns for national forests in northeastern Oregon and southeastern Washington (Mutch et al. 1993). A recent survey by Oregon State University, however, showed a stronger public preference for thinning (79% of respondents) than for prescribed fire (20%) as alternative treatments for addressing forest health concerns in the Blue Mountains (Shindler and Reed 1996, Shindler and Toman 2003).

A proposal to greatly expand use of prescribed fire (Mutch et al. 1993, Mutch 1994) raised concerns about potential impacts on forest productivity, wildlife habitat, and biodiversity. One response to this proposal was that mechanical fuel treatment might be preferable to a dramatic increase in prescribed fire because it offers more control than fire, and more control translates into better protection for dead wood (down logs and snags) (Tiedemann et al. 2000).

Figure 21 – A prescribed fire burning at night. Using prescribed fire on dry sites is intended to emulate the historical fire regime that sustained open, park-like forests of ponderosa pine. In the Blue Mountains, surface fires with short flame lengths (less than 3 feet) occurred at an interval of 10 to 25 years, a fire frequency favoring the thick-barked ponderosa pines and western larches while discriminating against the thin-barked grand firs and Douglas-firs (Agee 1996b, Hall 1976, Heyerdahl 1997, Maruoka 1994).

When considering fuel reduction options, mechanical methods might be more expensive than prescribed fire in the short term but are probably more economical over the long run, especially if wildlife habitat (snags and down logs) must be mitigated or replaced after burning (Tiedemann et al. 2000). But down-wood objectives for dry-forest sites need to be compatible with inherent ecosystem processes. The widely used models of dead-tree (snag) and down-log dynamics developed over 25 years ago for the Blue Mountains (Maser et al. 1979, Thomas et al. 1979) are not fully compatible with inherent, dry-forest disturbance regimes.

The Thomas et al. (1979) snag model portrays snags as going through nine stages of decay corresponding to the length of time a dead tree has been standing, eventually culminating in 'snag mortality' when the snag falls (fig. 22). Fallen snags then become downed logs, which go through another series of five classes of decomposition and decay (Maser et al. 1979).

Frequent fires on dry-forest sites tended to burn snags before they could progress through all the stages of the Thomas et al. (1979) snag model (Agee 2002a). The few fallen snags that did become downed logs also did not progress through all the stages of the Maser et al. (1979) model because they typically burned when in decomposition class 1 or 2, seldom avoiding fire long enough to reach class 5 (fig. 22).

For the dry-forest climatic zone of the interior Pacific Northwest, the short-interval fire regime functioned as the primary wood and nutrient cycling process because microbial decomposition was too slow on these arid environments to keep pace with woody biomass accumulation (fig. 8 explains the nutrient cycling differences between dry and moist forest sites).

When considered through the prism of ecosystem adaptation, dry mixed-conifer forests had low down-wood potential because frequent fire consumed much of the system biomass (Agee 2002a), leaving what biomass that did accumulate to do so in the most persistent component of the ecosystem – large, old trees (see fig. 26). Because of the evolutionary adaptations of the dominant trees (thick bark and an elevated canopy), the resilience of these ecosystems was high, even when considering the frequency of low-severity surface fire as a disturbance process.

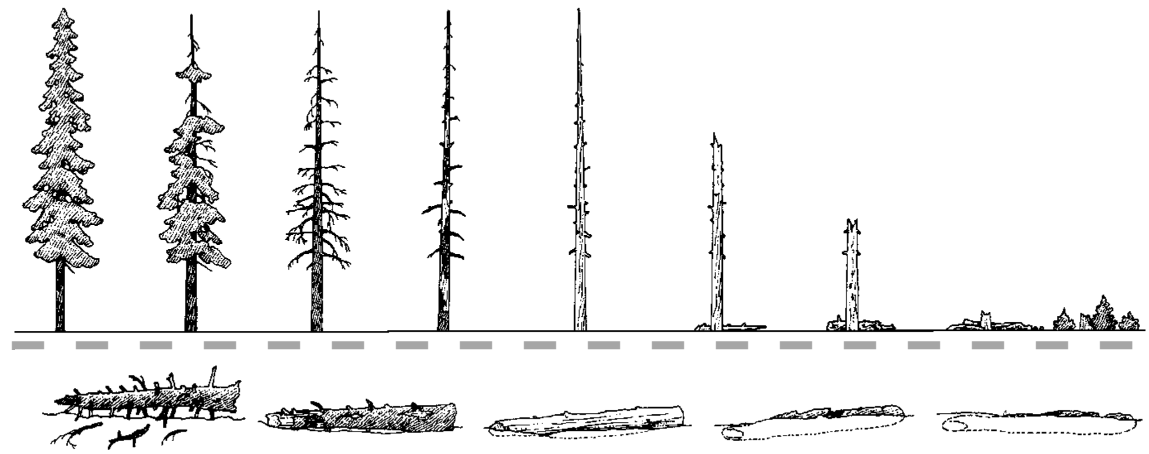


Figure 22 – Diagrams illustrating succession and evolution of snags (above dashed gray line) and down logs (below gray line) through time (from Maser et al. 1979 and Thomas et al. 1979). These models of snag and down wood succession are most appropriate for ecological environments where decomposition is primarily accomplished by microbes (e.g., moist and cold upland forests; see fig. 8). They are probably less appropriate for sites where woody detritus was cycled mainly by a frequent fire regime. Note that for dry sites, fire functioned ecologically as a coarse filter for both snags and logs (Agee 2002a).

What would happen if prescribed fire, rather than thinning, was applied to contemporary dry-site forests? In general, the outcome would be undesirable whenever a cohort of post-fire-exclusion trees is present (Bonnicksen 2000b). This post-exclusion cohort serves as ladder fuel (fig. 23), allowing a low-intensity surface fire to climb into the upper canopy and kill most of the dominant trees, including the fire-resistant species (Arno et al. 1997, Steele et al. 1986).

Where large quantities of standing dead trees are present following stand-replacing fire on dry sites, salvage harvest should be encouraged to remove some portion of this uncharacteristic fuel accumulation (Harvey et al. 1999, Mutch et al. 1993). [Despite the controversy surrounding post-fire salvage harvest (see Beschta et al. 2004), I maintain that if a dry forest's live-tree density is uncharacteristically high, and if it burns with uncharacteristic fire severity (lethal rather than non-lethal), then the resulting dead-tree density is also uncharacteristic and salvage harvest is appropriate for reducing the number of dead trees to characteristic levels by retaining the large-diameter, pre-fire-exclusion trees (see Brown et al. 2003 for post-fire fuel levels). What was uncharacteristic when alive does not automatically become characteristic when dead.]

Many land managers agree that wildfire exclusion was a policy with good intentions, but it was a policy that failed to consider the ecological implications of a major shift in species composition. Grand firs and Douglas-firs can get established under ponderosa pines in the absence of surface fire, but they may not have enough resilience to make it over the long run, let alone survive the next drought. This means that many of the mixed-conifer stands that replaced ponderosa pine are destined to become weak, and weak forests are susceptible to insect outbreaks and disease epidemics.

Effects of the 1980s western spruce budworm outbreak (Powell 1994)

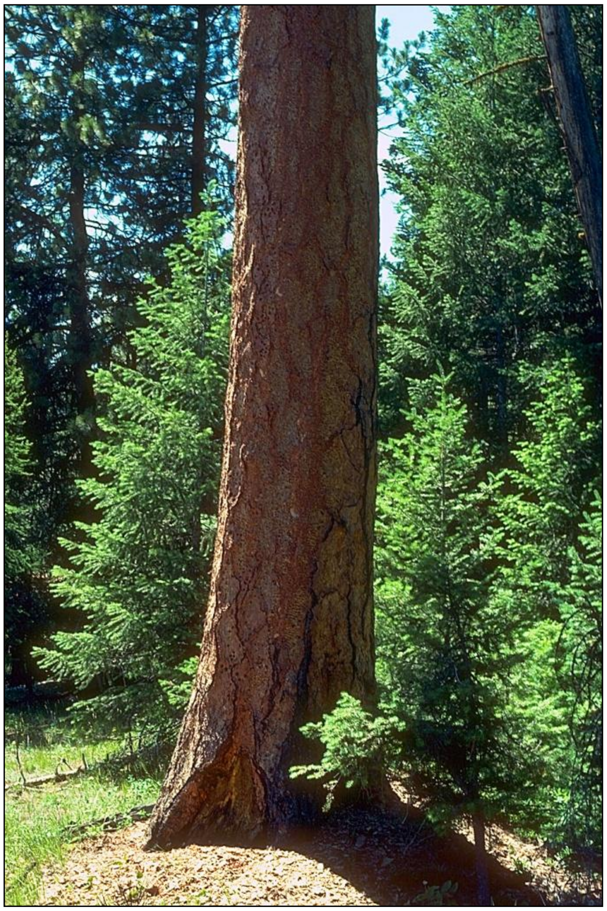


Figure 23 – Shade-tolerant trees can get established on dry sites in the absence of surface fire (from Powell 1994). Grand firs and Douglas-firs are clustered around the base of the ponderosa pine in this image. Eighty or more years of fire exclusion promoted this successional progression on millions of acres in western North America (Schmidt et al. 2002). If selective harvest removes the overstory trees, a multi-layered stand of late-seral species remains and most of them are highly susceptible to drought and damage from defoliating insects (Wickman 1992). On dry sites where grand fir or Douglas-fir is climax, prescribed fire is effective for managing ingrowth of late-seral species (Kalabokidis and Omi 1998). The mound of bark flakes at the base of this old pine is a common indicator of long-term fire exclusion; fire can smolder there and kill the fine roots (Ryan and Frandsen 1991). Note: Experience in the American southwest suggests it requires about 100 years for ponderosa pine to develop the characteristic orange, platy bark shown here (White 1985).

7.9 Restoration Alternatives

To be healthy, trees need a place in the sun and some soil to call their own (Society of American Foresters 1981). When crowded by too many neighbors, trees may not have enough soil and sun to maintain high vigor. Trees die after their vigor drops so low they can no longer heal injuries, resist attack by insects and diseases (by producing phenols, monoterpenes and other terpenoid resins, and similar defensive chemicals), or otherwise sustain life (fig. 24; Christiansen et al. 1987, Franklin et al. 1987, Kelsey 2001, Kolb et al. 1998, Langenheim 1990, McDowell et al. 2007, Nebeker et al. 1995, Peet and Christensen 1987, Wallin et al. 2008, Waring 1987).

Once a forest stand occupies all of its growing space, intertree competition causes some trees to die and the survivors immediately claim the growing space relinquished by their dead neighbors. In nature, this self-thinning process eventually results in relatively few large trees occupying the growing space that once supported many small trees (Long and Smith 1984).

Land managers can emulate this natural competition process by intentionally reducing the number of trees on a site, a silvicultural practice called thinning. Thinning has been used to describe practices ranging from light removal of small understory trees to moderate removal of large overstory trees. On dry-forest sites where thinning is designed to emulate surface fire (Perera et al. 2004), any reference to thinning is assumed to be "understory thinning, thinning from below, or low thinning," all of which refer to cutting or removal of subordinate (understory) trees (fig. 25; Smith et al. 1997).

Critics of active management often characterize thinning as a silvicultural practice designed for commodity wood (timber) production, rather than acknowledging what it truly is – application of a restoration tool in proper places and at appropriate times to achieve specific land management objectives through active vegetation management. One contemporary objective is to create fire-safe forest conditions, particularly for highly developed areas referred to as the wildland-urban interface, and thinning addresses three of four fire-safe principles (table 5).

Section 6 describes how selective cutting was one of three important factors contributing to dry-forest deterioration (the other two are fire exclusion and ungulate herbivory). Selective cutting, however, must not be confused with thinning. Not only are the two activities implemented in different ways, but selective cutting was directed at short-term (economic) objectives, while thinning is designed to meet long-term (silvicultural) objectives. These differences demonstrate that mechanical treatments are not all the same – low thinning is an ideal restoration activity for dry forests, whereas selective cutting contributed to the problem in the first place.

By removing some trees and thereby increasing the space around those that remain, thinning provides more sunlight, water, and nutrients for the residual trees. A reduction in tree density quickly improves the physiological condition and vigor of the residual trees. High-vigor trees produce more resin and defensive chemicals than low-vigor trees, allowing them to better repel insect and disease attacks (Christiansen et al. 1987; Feeney et al. 1998; Kolb et al. 1998; McDowell et al. 2003, 2007; Mitchell et al. 1983; Stoszek 1988; Vité 1961; Waring and Pitman 1985).

To capitalize on its forest health benefits, thinning was emphasized in Oregon Governor John Kitzhaber's strategy for restoring eastern Oregon forests, watersheds, and communities: "Understory thinning of green trees to restore forests to a healthy condition more representative of historic conditions is an important component of active management for forest health" (Kitzhaber et al. 2001).

Selective cutting. A system in which groups of trees, or individual trees, are periodically removed from the forest as based on economic criteria aimed at maximizing commodity revenues rather than trying to meet silvicultural objectives such as regeneration (Dunster and Dunster 1996).
Selection cutting. A regeneration cutting method designed to maintain and perpetuate a multi-aged structure by removing some trees in all size (age) classes either singly (single-tree selection) or in groups (group selection) (Helms 1998).
Thinning. A treatment designed to reduce tree density and thereby improve growth of the residual trees, enhance forest health, or recover potential mortality resulting from intertree competition. Two types of thinning are recognized – commercial thinning where the trees being removed are large enough to have economic value, and noncommercial thinning where trees are too small to be sold for conventional wood products, so the excess trees are cut and left on-site (Powell et al. 2001).
Prescribed fire. Deliberate burning of wildland fuels in either a natural or modified state, and under specified environmental conditions, in order to confine the fire to a predetermined area, and to produce a fireline intensity and rate of spread meeting land management objectives (Powell et al. 2001).

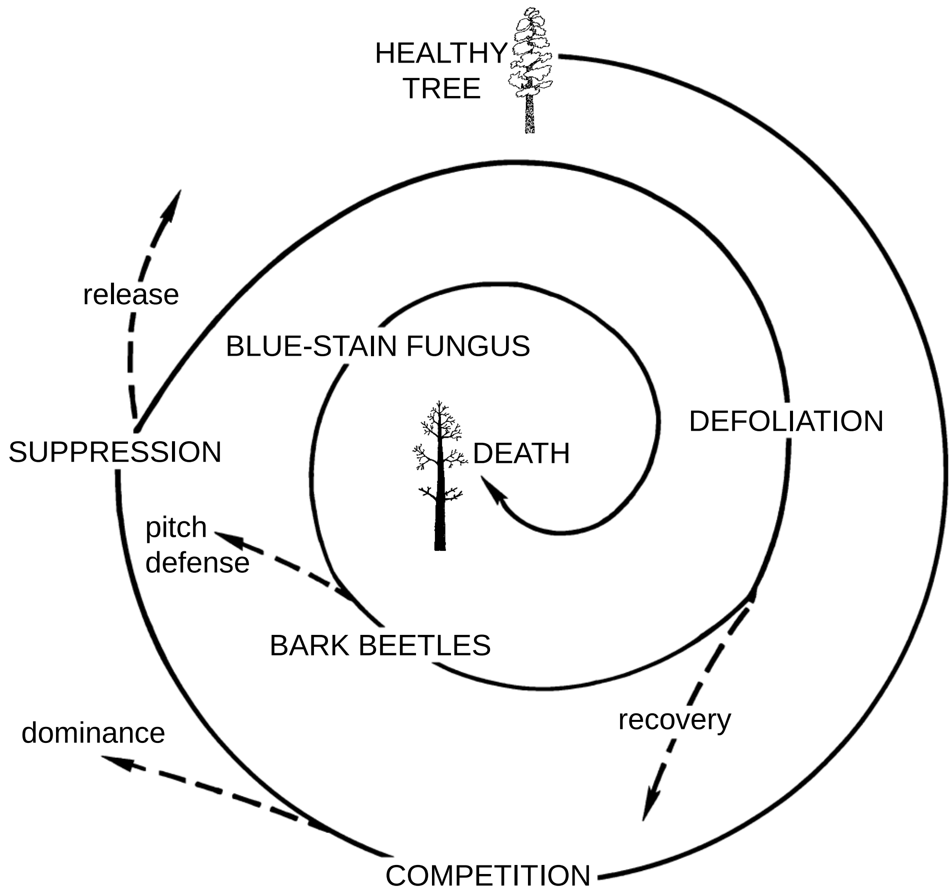


Figure 24 – Death spiral for a Douglas-fir tree in the Blue Mountains (adapted from Franklin et al. 1987). In this death spiral, a slightly taller tree suppresses a shorter but otherwise healthy tree. If not released from competition, the suppressed tree is predisposed to attack by defoliators. Once partially defoliated, the weakened tree is attractive to bark beetles such as Douglas-fir beetle (Wickman 1978) that carry blue-stain fungus. The fungus blocks water and sap movement and causes foliage desiccation. In this model of tree decline, suppression is a predisposing stressor; bark beetles and defoliation function as culminating or inciting stressors (Pedersen 1998).

When comparing mechanical thinning and prescribed fire as active restoration alternatives for dry-forest sites, mechanical thinning offers several advantages: (1) it provides the most control over species composition, vertical structure, tree density, and spatial pattern for the residual forest; (2) it provides more control over the amounts and distribution of standing and down wood (Tiedemann et al. 2000); (3) it is not constrained to short, unpredictable weather windows like prescribed fire; and (4) it may produce economically valuable wood products that could help defray the cost of restoration treatments (Barbour et al. 2007).

Regardless of which method is used, fuel treatments need to account for wildlife needs and similar resource concerns. For example, research found that treated stands provided better elk forage during spring, whereas untreated stands provided better summer forage. These results suggest that maintaining a mosaic of treated and untreated habitat may provide better long-term foraging opportunities than treating a large proportion of a landscape (Long et al. 2008).

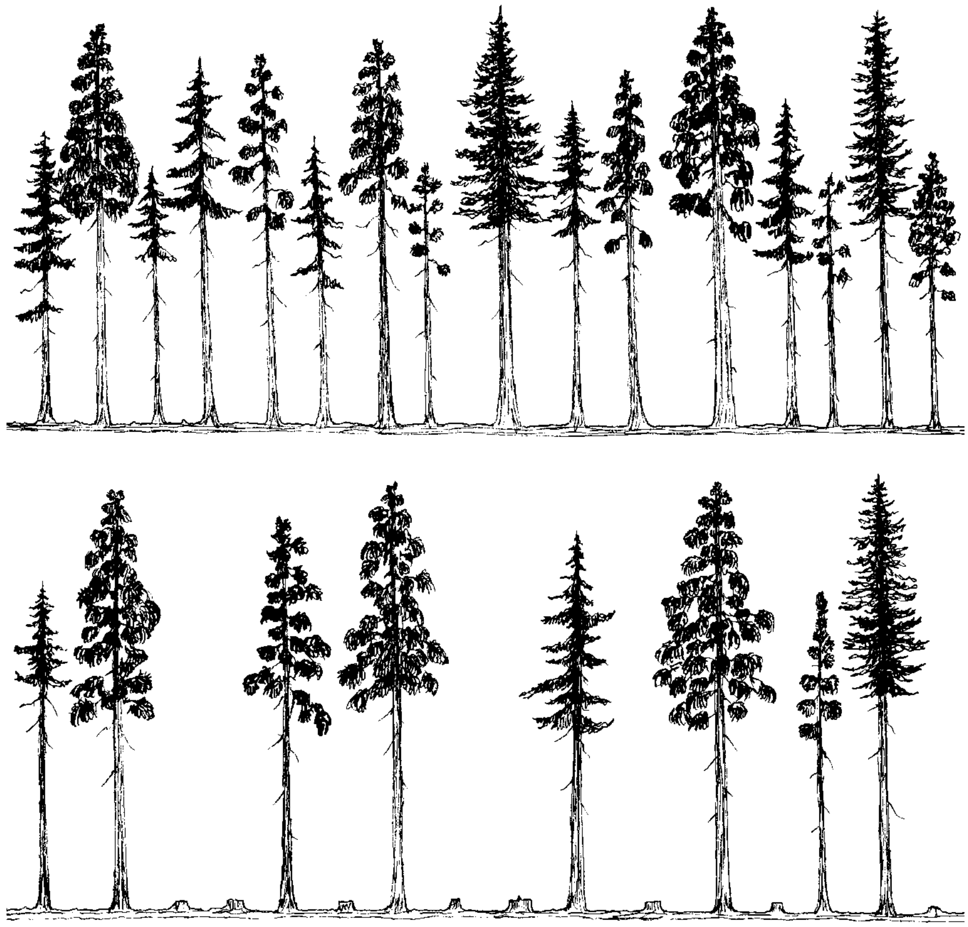


Figure 25 – Example of low thinning in a mixed-conifer forest (from Powell 1999). Low thinning is defined as the removal of trees from lower crown classes or canopy layers in order to favor those in upper crown classes or layers. Low thinning is also referred to as 'thinning from below.' Note how smaller trees were removed in every instance but one: the western larch in the center of the top panel was infected with dwarf mistletoe to an extent that threatened its continued survival. Because of its canopy position, the larch would not have been removed except for insect or disease reasons.

Some critics of active management view thinning or any other incarnation of 'logging' as less natural (and therefore less desirable) than prescribed burning. Their expectation is that prescribed fire should serve as the primary tool to remove excess fuel, preferring to see a forest burn rather than receive protection from a chain saw. The main problem with this expectation is its incompatibility with ecosystem changes wrought by fire exclusion over the last 75 years. Applying fire on sites with a cohort of post-fire-suppression trees could allow the burn to use this 'ladder fuel' to climb into the upper canopy and kill the mature trees, including fire-resistant species. Although fire was nature's most common way to reduce excess vegetation on dry-forest sites, we can't pretend we're dealing with a 'natural' situation now.

Forest density management: recent history and trends (Powell et al. 2001)

Table 5: Principles of fire-safe forests.

PRINCIPLE	EFFECT	ADVANTAGE	CONCERNS
Reduce surface fuels	Reduces potential flame length	Fire control is easier; less torching of individual trees	Soil disturbance: less with prescribed burning, more with certain mechanical treatments
Increase height to live crown	Requires longer flame length to begin torching	Less torching of individual trees	Opens understory, possibly allowing surface winds to increase
Decrease canopy bulk density (foliage biomass)	Makes tree-to-tree crown fire spread less likely	Reduces crown fire potential	Surface winds may increase and surface fuels may become drier
Favor fire-tolerant tree species	Reduces potential tree mortality	Improves vegetation tolerance to low- and mixed-severity fire	If used too broadly, it could simplify composition at landscape scale

Sources: Adapted from Agee et al. (2000) and Agee (2002b).

Although contentious debate over the merits of salvage tree harvest following stand-replacing fire on dry-forest sites (Beschta et al. 2004, McIver and Starr 2000) continues to distract decision makers from the pressing issues of forest health and ecosystem restoration, there is some agreement among foresters, fire ecologists, and conservationists about eight cost-effective ways to expedite restoration actions (Phillips 1995):

1. **Rethink local air-quality regulations including 'nuisance smoke' ordinances.** This would allow more use of prescribed fire and reduce pressure to extinguish natural fire ignitions that could be allowed to burn under prescribed conditions.
2. **Resolve liability issues** – fear of lawsuits over property damage from escaped prescribed fires prevents many forest managers from using this tool.
3. **Increase funding for hazardous fuels reduction.** The National Fire Plan has an objective of reducing hazardous fuels but funding for this type of work has not increased to the same extent as it has for fire suppression activities.
4. **Determine the extent to which environmental regulations are inhibiting forest restoration.** Legislation such as the Endangered Species Act and the National Environmental Policy Act have broad public support, but their implementing regulations could be modified to expedite fuel management treatments.
5. **Plan better for residential development in the wildland-urban interface (WUI).** This is more a political issue than a forest health issue, but the presence of WUI increasingly affects how surrounding forests are managed (or not managed).
6. **Restrict herbivory in forestlands.** Domestic livestock grazing has been reduced from its early-1900s levels, but the combined effect of domestic and wild ungulates has contributed to replacement of some meadows and grasslands with woody, flammable vegetation.
7. **Create new markets for wood chips from mechanical thinning of small-diameter trees.** One option would be to use federal revenues from tree harvest to help local communities

develop technology for producing veneers, fiberboard, or biomass material to generate ethanol, electricity, and thermal energy (LeVan-Green and Livingston 2001).

8. **Plan for landscape restoration.** Many computer models, decision support systems, and visualization systems can be used to help balance public expectations for forest uses with the need to reestablish landscapes supporting characteristic levels of fire, insect, or disease hazard. We must first identify landscapes with the highest priority for restoration treatments, and then seek to create a vegetation mosaic that functions within its range of variation.

7.10 Restoring Old Forest on Dry Sites

Old growth. Forest stands distinguished by old trees and related structural attributes such as tree size, accumulations of large dead woody material, number of canopy layers, species composition, and ecosystem function (Newton 2007).

Old forest. A structural stage characterized by a predominance of large trees (> 21" dbh) in a forest having either one or multiple canopy layers. On warm dry sites historically influenced by frequent surface fire, a single tree stratum may be present. On cool moist sites without recurring underburning, multi-layer stands with large trees in the uppermost stratum are typically found.

Restoration. Holistic action taken to modify an ecosystem to achieve desired, healthy, and functioning conditions and processes. Generally refers to the process of enabling a system to resume acting, or continue to act, following disturbance as if disturbance had not occurred (Powell et al. 2001).

In the interior Pacific Northwest, old forest structure occurs predominantly on two site types: dry sites and moist sites. Old forests were developed and maintained by different disturbance regimes on each of these biophysical environments (Camp et al. 1997, Everett et al. 1994, Habeck 1990, O'Hara et al. 1996, Oliver and Larson 1996).

On dry mixed-conifer sites, frequent surface fires historically interrupted plant succession toward a climatic climax, thereby preventing eventual domination by interior Douglas-fir or grand fir. This short-interval fire regime maintained an early-seral species composition consisting of ponderosa pine (fig. 26); these stands were stable and resilient because ecosystems shaped by frequent disturbance exhibit a relatively narrow range of plant communities (Steele and Geier-Hayes 1995). The old forest structure produced by frequent fire is termed old forest single stratum (table 6).

Because cyclic fire remained relatively constant on dry mixed-conifer sites, ponderosa pine forests came to depend on a particular fire frequency and intensity (Sloan 1998). Fire frequency must be maintained at an appropriate periodicity if ponderosa pine is to persist, and this is the reason why fire frequency and not occurrence has so much ecological influence. Species composition remembers fire, but abundance (tree density) forgets (Allen and Wyleto 1983).

A historic condition on dry sites was old ponderosa pine trees occurring in a park-like, savanna setting (fig. 27). This park-like structure did not occupy the entire landscape; dry mixed-conifer forest communities also supported snags, fallen logs, mid-size blackjack pines, and small seedlings and saplings. All of these stand attributes were influenced and sculpted by fire (Agee 2002a; Cooper 1960, 1961; Harrod et al. 1999; Munger 1917; Woolsey 1911; White 1985).

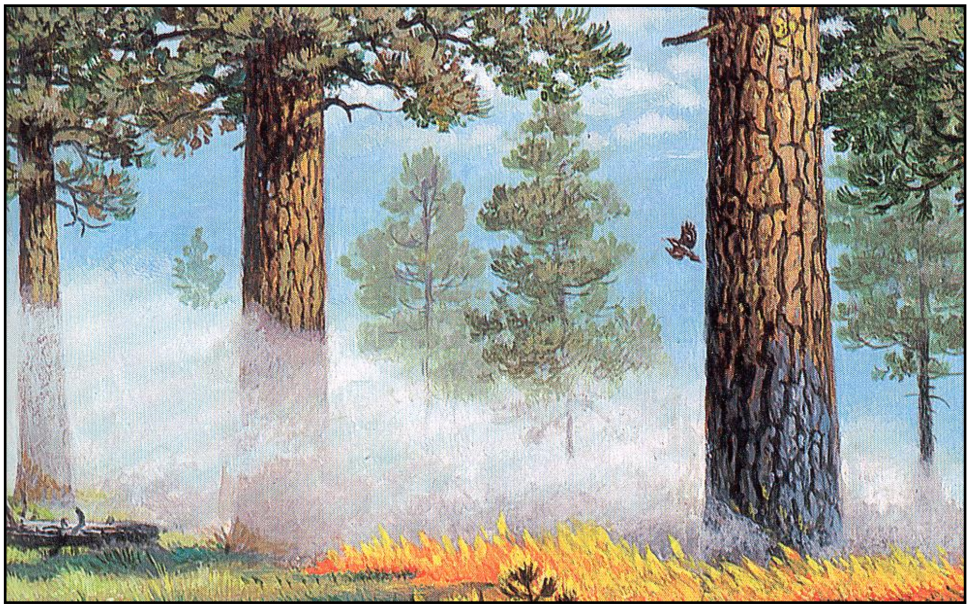
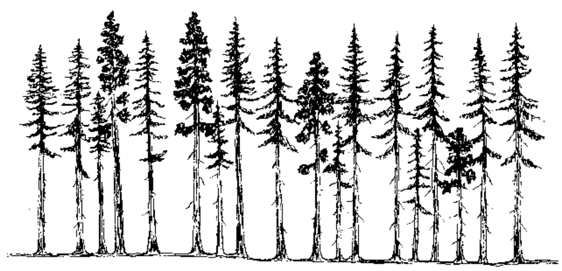


Figure 26 – Low severity surface fire in ponderosa pine forest (from Powell et al. 2001). In eastern Oregon, the presettlement fire regime created stable old forest referred to as 'park-like pine forest.' These ecosystems featured big, widely spaced ponderosa pines above a dense herb layer (also see fig. 5). This condition owed its stability to recurring visits by relatively benign wildfire every 5-20 years (Cooper 1960; Hall 1976, 1980; Munger 1917; Parfit 1996) (illustration by John D. Dawson, National Geographic Society).

Figure 27 – An open ponderosa pine stand with a grassy undergrowth (from Powell 1994). By suppressing low-severity, high-frequency surface fire, land managers were inadvertently swapping ponderosa pines for grand firs and Douglas-firs. This successional progression has important implications on susceptibility to defoliating insects such as Douglas-fir tussock moth and western spruce budworm because the replacement tree species provide habitat for these insects (Mason and Wickman 1994, Wickman 1992).

Table 6: Description of forest structural stages.

	**Stand Initiation**. Following a stand-replacing disturbance, growing space is occupied rapidly by vegetation that either survives the disturbance or colonizes the area. Survivors literally survive the disturbance above ground, or initiate new growth from their underground organs or from seeds on the site. Colonizers disperse seed into disturbed areas, it germinates, and new seedlings establish and grow. One stratum of tree seedlings and saplings is present in this stage.
	**Stem Exclusion**. Trees initially grow fast and quickly occupy their growing space, competing strongly for sunlight and moisture. Because trees are tall and reduce light, understory plants (including small trees) are shaded and grow slowly. Species needing sunlight usually die; shrubs and herbs may go dormant. In this stage, establishment of new trees is precluded by a lack of sunlight (stem exclusion closed canopy) or by a lack of moisture (stem exclusion open canopy).
	**Understory Reinitiation**. A new tree cohort eventually gets established after overstory trees begin to die or because they no longer fully occupy their growing space. This period of overstory crown shyness occurs when tall trees abrade each other in the wind (Putz et al. 1984). Regrowth of understory vegetation occurs, trees begin stratifying into vertical layers, and a moderately dense overstory with small trees beneath is eventually produced.
	**Old Forest**. Many age classes and tree layers mark this stage featuring large, old trees. Snags and fallen trees may also be present, leaving a discontinuous overstory canopy. The drawing shows single-layer ponderosa pine created by frequent surface fire on dry sites (old forest single stratum). Cold or moist sites, however, generally have multilayer stands with large trees in the uppermost stratum (old forest multi strata).

Sources: Based on O'Hara et al. (1996), Oliver and Larson (1996), and Spies (1997).

Thinning to develop an old-forest structure on dry sites differs from thinning to maximize tree growth and timber production. The complex structure of old forests is a product of their variability. Variable-density thinning promotes complexity by: (1) thinning to different densities across a range of patch sizes; (2) leaving some patches or portions of patches unthinned; and (3) creating small gaps (up to ½ acre in size) in other areas (Armleder 1999). "Studies show that when variable-density thinning is used, thinned stands usually have better developed understories, higher shrub densities, a greater richness of understory plant species, and more plant cover than unthinned stands" (McDowell et al. 2003, Rapp 2002).

7.11 Principles of Old Forest Restoration

An old-forest restoration program for the Blue Mountains should incorporate the following concepts relating to the landscape ecology of eastern Oregon (Camp et al. 1997, Everett et al. 1994):

- Current anomalous landscapes and disturbance regimes need to be restored to a more sustainable state if old-forest remnants are to be conserved, and old-forest networks created and maintained (Hessburg et al. 2005).
- Today, a mosaic of young forest types with heightened fire and insect hazard surrounds many old-forest remnants.
- Given the limited contribution from any individual old-forest patch, additional old-forest stands need to be continually created to maintain a dynamic equilibrium through time.
- Efforts to conserve old forest should not sacrifice contributions from other limited structures or components in the landscape.
- Conserving the disturbance processes that influence ecosystems is every bit as important as conserving individual plant and animal species or old forest structure – a lack of disturbance can be as threatening to biological diversity as excessive disturbance (Noss 1983).
- Management regimes for old-forest patches should be congruent with the disturbance regimes characteristic of their associated landscape.
- Any plan to sustain old forests must also sustain the landscape of which they are a part (Hessburg et al. 2005).
- In managing old forests, a landscape perspective is needed that coordinates species requirements with ecological processes and other functional ecosystem attributes (box 1) (Hessburg et al. 2005).
- Forest ecosystems of the interior Pacific Northwest exist in a constant cycle of change, and it should be acknowledged that the successional pathway of a certain proportion of the forest stands will be interrupted by fire, windthrow, insect attack, or disease before they can reach an old-forest condition.

7.12 Strategies for Restoring Old Forest

An effective restoration strategy for old forests in dynamic landscapes of the Blue Mountains and interior Pacific Northwest should incorporate many of these considerations (Camp et al. 1997, Everett et al. 1994):

- Conservation of the remaining old-forest patches is the cornerstone of any management scheme, if for no other reason than it best maintains future options.
- Sites that do not have a full complement of old forest attributes can partially function as old forest for any attributes that are present.
- Dry old-growth forest differs dramatically from west-side Douglas-fir/hemlock old-growth (Franklin et al. 1981), and it may seem decrepit if evaluated by using those standards.

- In some parts of the landscape it may be necessary to designate areas of younger forest as old-forest management areas (stands having priority for old-forest development) in order to meet desired future objectives with respect to a structural-stage distribution.
- Silvicultural practices can be used to accelerate development of old-forest characteristics in young stands, particularly with respect to practices influencing regeneration density, stocking levels, or competing vegetation (Gottfried 1992, Spies et al. 1991).
- Research showed that tree growth increases rapidly after stand density levels are reduced (Barrett 1979; Seidel and Cochran 1981), suggesting that thinning will accelerate production of the large-tree component of old forest (Sullivan et al. 2001, Tappeiner et al. 1997).
- When identifying possible candidates for future 'old forest multi strata' stands in large landscapes containing dry forests, stands should be selected with the highest survival potential to an old forest condition – specifically areas on north-facing aspects and at high elevations, particularly if they also occur within valley bottoms and drainage headwalls because these physiographic positions are semi-stable environmental settings (Camp et al. 1997).
- Although mid- to late-seral stands are 'in the pipeline' to replace old forests lost to disturbance, we still do not know the appropriate ratio of late-seral to old forest patches to ensure that current or desired levels of old forest are maintained in perpetuity.
- Evaluating historical amounts of old forest (as is often done when analyzing the range of variation for forest structural stages) provides a first approximation for how much old forest was sustainable, and in which old-forest-dependent plant and animal species evolved.
- Ideally, historical evaluations should incorporate several reference points in time, and at sufficient spatial scales, to ensure that major disturbance regimes have been accounted for.
- A successful strategy would allow flexibility in specific on-the-ground locations over time. The 'shifting mosaic' landscape concept (Clark 1991) suggests a dynamic framework in which old forest patches are lost and created at appropriate spatial and temporal scales.
- Restoration of old forests carries long-term management costs with little expectation of substantial commodity production. Creation of an old-forest network explicitly assumes that biological diversity and other old-forest objectives are supported socially and economically.
- A dynamic ecosystems philosophy should be the foundation of an old-forest strategy – an ecologically sustainable representation of old forest structure in the landscape is more important than perpetuation of old forest patches in a specific location. Old-growth should be perceived as a dynamic entity influenced primarily by fine-scale mortality and recruitment.
- Research suggests that light fuel treatments across a portion of the landscape provide a considerable reduction in overall landscape fire risk, although they may not lower the risk for individual reserves containing large trees and multi-layered canopies (Wilson and Baker 1998).
- Efforts to protect individual old-forest stands through moderate or intense management of adjacent stands apparently provide minimal reductions in fire risk, although reducing surface fuels by using a combination of thinning and prescribed fire might qualify as the only treatment regime providing at least a modicum of fire protection (Wilson and Baker 1998).
- Low thinning and prescribed fire treatments in an old-forest stand will lower its fire risk substantially; however, some of the stand's old-forest characteristics (such as multi-layered canopies) are altered by such practices (Wilson and Baker 1998).

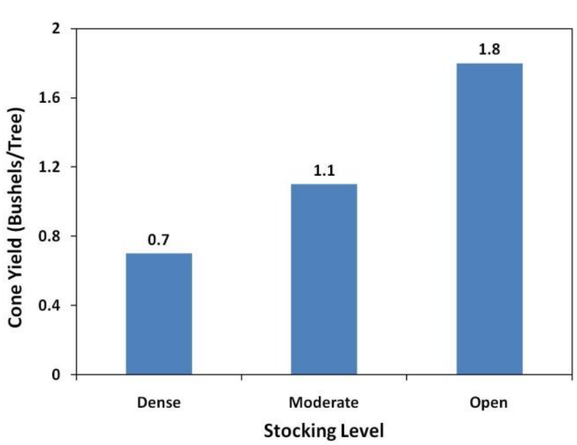
Box 1: Stand Density and White-Headed Woodpecker

This white paper describes how fire exclusion, ungulate herbivory, and selective timber harvest contributed to significant changes in dry-forest ecosystems. These changes resulted in a current emphasis on restoration of dry forests, not just for the Blue Mountains but throughout western North America. One restoration strategy involves reducing tree density to levels approximating the presettlement stand-density situation. Density reductions contribute to lower fire and insect susceptibility, rejuvenation of undergrowth plant abundance and species diversity, and improved wildlife habitat for species dependent on presettlement ponderosa pine forest conditions. One species of interest in this regard is the currently uncommon white-headed woodpecker (*Picoides albolarvatus*). In the Oregon portion of this woodpecker's west-wide range, ponderosa pine cones are believed to provide the primary suitable food source during non-breeding periods (Garrett et al. 1996).

Dry-forest restoration activities in Washington that focused on reintroduction of fire were apparently successful at increasing woodpecker abundance (Krannitz and Duralia 2004). Another restoration option is thinning, an active management practice believed to be especially applicable to white-headed woodpecker because it addresses the bird's food base by increasing cone and seed production (see chart below, showing cone yield by stocking level). Ponderosa pine cone production was observed to vary consistently with stand density in the southwestern United States (Pearson 1912). "Since trees of larger diameter produce the majority of cones, increased cone production may be a longer-term benefit of thinning" (Krannitz and Duralia 2004).

It is also believed that an interaction between thinning and fire can increase the cone production benefits of active management on dry sites: when wildfire occurred in a thinned stand in Arizona, and the woody debris had been removed before it occurred, the fire improved resin production as compared with an unthinned control (Feeney et al. 1998). Similar results were reported in other studies (Kolb et al. 1998). Large-diameter ponderosa pine trees with increased capacity for producing resin and other defensive chemicals (Christiansen et al. 1987, Franklin et al. 1987, Kelsey 2001, Kolb et al. 1998, Langenheim 1990, McDowell et al. 2007, Nebeker et al. 1995, Peet and Christensen 1987, Waring 1987) are more likely to resist attack by western pine beetle, the primary bark beetle species known to prey on low-vigor, old-growth ponderosa pines.

Trees respond to thinning by producing more foliage and developing a higher level of photosynthate reserves, both of which improve their capability to resist and recover from insect or disease attack (Franceschi et al. 2005). A tree allocates photosynthate to its growth processes in an order of precedence: (1) maintenance respiration; (2) fine root and foliage production; (3) flower and seed production; (4) height, branch, and large-root growth; (5) diameter growth; and (6) insect and disease resistance. Since seed production and insect resistance rank fairly low in the hierarchy (#3 and #6, respectively), management practices can be used to sustain tree vigor at levels high enough to ensure that sufficient photosynthate is available to satisfy these physiological needs.



7.13 Range of Variation Concept

Range of variation. A characterization of fluctuations in ecosystem conditions or processes over time; an analytical technique used to define the bounds of ecosystem behavior that remain relatively consistent through time (Morgan et al. 1994). The values of an attribute, such as composition or structure, that fall within the upper and lower bounds determined for the attribute (Jennings et al. 2003).

The range of variation (RV) is an analytical technique to characterize inherent variation in ecosystem composition, structure, and function, reflecting recent evolutionary history and the dynamic interplay of biotic and abiotic factors (fig. 28). “Study of past ecosystem behavior can provide the framework for understanding the structure and behavior of contemporary ecosystems, and is the basis for predicting future conditions” (Morgan et al. 1994).

RV is meant to reflect ecosystem properties free of major influence by Euro-American humans, thereby providing an insight into ecosystem resilience (Kaufmann et al. 1994). It helps us to understand what an ecosystem is capable of, how historical disturbance regimes functioned, and the underlying variation in ecosystem processes and functions – the patterns, connectivity, seral stages, and cover types produced by ecological processes operating at a landscape scale (USDA Forest Service 1997).

Perhaps the best yardstick for evaluating the health of dry forests is historical variation – are changes caused by insects, diseases, and wildfire consistent with what would be expected (the RV) for similar ecosystems and vegetative conditions? Since ecosystems are constantly changing, we need to assess their health in a similar context. Resilient forests not only tolerate periodic disturbance, they may depend on it for rejuvenation and renewal (Johnson et al. 1994). Significant changes in the magnitude (extent), intensity, or pattern of disturbance, however, may function as warning signals of impaired ecosystem integrity (Sampson and Adams 1994).

The range of variation concept has been proposed as a way to identify restoration needs and opportunities. Using reference conditions to guide our restoration programs will continue into the future because this approach is explicitly required by certain laws governing dry-forest management, such as the Healthy Forests Restoration Act: “In carrying out a covered project, the Secretary shall fully maintain, or contribute toward the restoration of the structure and composition of old growth stands according to the pre-fire suppression old growth conditions” (http://agriculture.senate.gov/forest/forhxadtsec.pdf).

Both now and in the future, a desirable landscape condition for the Blue Mountains province is a diverse, heterogeneous vegetation mosaic more consistent with the historical range of variation, less susceptible to uncharacteristic disturbance events, and thus more sustainable (Mutch et al. 1993, Sampson et al. 1994). Using an RV approach to help restore vegetation diversity means providing a full spectrum of structural elements in variable configurations and quantities, with the ultimate objective being maintenance of the dynamic patterns and processes that are integral to resilient ecosystems (Aplet and Keeton 1999).

Range of variation information for species composition, structural stage, tree density, and insect and disease susceptibility is provided in tables 7-10.

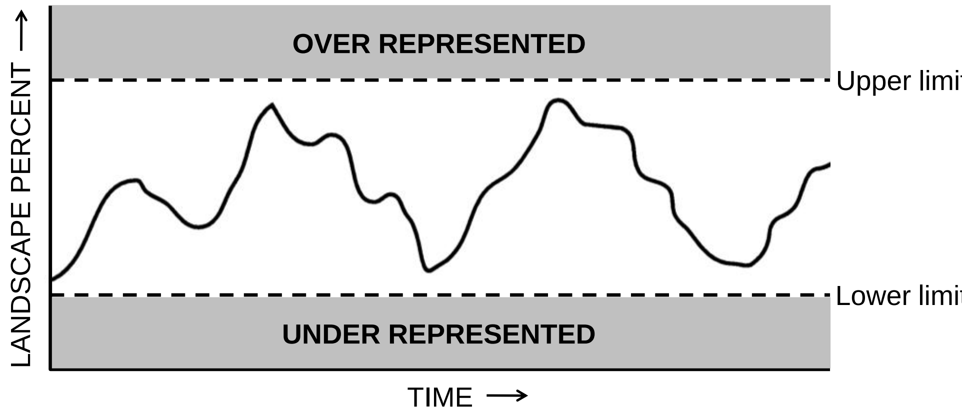


Figure 28 – The range of variation (RV) helps us decide whether existing amounts of vegetation composition, structure, and density, when summarized for a landscape-scale analysis area, are occurring within a characteristic range (Aplet and Keeton 1999, Morgan et al. 1994, Swanson et al. 1994). This diagram shows the ecological trajectory of an ecosystem component (the solid line) varying through time because the phrase 'range of variation' is meant to encompass more than just the extreme values (the upper and lower limits, shown as dashed lines) (diagram modified from Morgan et al. 1994).

RV is a good example of the dynamic equilibrium concept because modal or central-tendency conditions obviously vary over time (shown by the squiggly solid line in the center), and yet they vary within an equilibrium zone whose limits (the dashed lines) are confined within a range of potential ecological expressions. Note that conditions occurring above the upper limit are considered to be over-represented; conditions below the lower limit are considered to be under-represented (the representation zones are gray).

Table 7: RV information for species composition (vegetation cover type) on the dry upland forest PVG.

Vegetation Cover Type	Range of Variation (Percent)
Grass-forb	0-5
Shrub	0-5
Western juniper	0-5
Ponderosa pine	50-80
Douglas-fir	5-20
Western larch	1-10
Broadleaved trees	0-5
Lodgepole pine	N.A.
Western white pine	0-5
Grand fir	1-10
Whitebark pine	N.A.
Subalpine fir and spruce	N.A.

Sources/Notes: Derived from disturbance process modeling using the Vegetation Dynamics Development Tool (VDDT) (Powell 2010). N.A. is Not Applicable.

Cover types reflect the existing vegetation composition of a polygon (Eyre 1980, Shiflet 1994). Cover type codes are described in Powell (2004); cover types consist of these coding combinations:

Grass-forb: all grass and forb codes;
Shrub: all shrub codes;
Western juniper: JUOC and mix-JUOC;
Ponderosa pine: PIPO and mix-PIPO;
Douglas-fir: PSME and mix-PSME;
Western larch: LAOC and mix-LAOC;
Broadleaved trees: POTR, POTR2, mix-POTR, and mix-POTR2;
Lodgepole pine: PICO and mix-PICO;
Western white pine: PIMO and mix-PIMO;
Grand fir: ABGR and mix-ABGR;
Whitebark pine: PIAL and mix-PIAL;
Subalpine fir and spruce: ABLA, PIEN, mix- ABLA, and mix-PIEN.

Table 8: Range of variation information for forest structural stage on the dry upland forest PVG.

Forest Structural Stage	Range of Variation (Percent)
Stand initiation	15-25
Stem exclusion	10-20
Understory reinitiation	5-10
Old forest single stratum	40-60
Old forest multi strata	5-15

Sources/Notes: Derived from disturbance process modeling using the Vegetation Dynamics Development Tool (VDDT) (Powell 2010). Forest structural stages are illustrated and described in table 6.

Table 9: Range of variation information for tree density on the dry upland forest PVG.

Tree Density Class (mixed species composition at a quadratic mean diameter of 10")	Range of Variation (Percent)
Low (<40% canopy cover; <45 ft^2/ac basal area; <81 tpa or sdi)	40-85
Moderate (40-50% canopy cover; 45-70 ft^2/ac basal area; 81-121 tpa or sdi)	15-30
High (>50% canopy cover; >70 ft^2/ac basal area; >121 tpa or sdi)	5-15

Sources/Notes: Derived from Powell (2010). Note that tpa refers to trees per acre; sdi refers to stand density index. All 'tree density class' values pertain to an irregular stand structure. The tpa and sdi values are the same because the stand density index system uses a 10" quadratic mean diameter (QMD) as its reference tree size, so sdi and tpa values are the same when QMD is 10 inches (if QMD had been any value other than 10", the tpa and sdi values would not have been identical).

Table 10: Range of variation information for insect and disease susceptibility on the dry upland forest PVG.

Insect and Disease Agents[1]	Range of Variation (Percent)
Defoliating insects	
Low susceptibility	40-85
Moderate susceptibility	15-30
High susceptibility	5-15

Insect and Disease Agents[1]	Range of Variation (Percent)
Douglas-fir beetle	
Low susceptibility	35-75
Moderate susceptibility	15-30
High susceptibility	10-25
Fir engraver	
Low susceptibility	45-90
Moderate susceptibility	10-25
High susceptibility	5-10
Bark beetles in ponderosa pine	
Low susceptibility	5-10
Moderate susceptibility	15-30
High susceptibility	40-90
Mountain pine beetle in lodgepole pine	
Low susceptibility	55-90
Moderate susceptibility	5-25
High susceptibility	0-5
Douglas-fir dwarf mistletoe	
Low susceptibility	25-55
Moderate susceptibility	15-40
High susceptibility	20-35
Western larch dwarf mistletoe	
Low susceptibility	55-95
Moderate susceptibility	5-30
High susceptibility	0-5
Root diseases	
Low susceptibility	30-60
Moderate susceptibility	25-50
High susceptibility	5-25

Sources/Notes: Derived from Schmitt and Powell (2008). Queries for calculating susceptibility ratings for forest polygons are available from Schmitt and Powell (2005).

[1] Defoliating insects includes western spruce budworm and Douglas-fir tussock moth; bark beetles in ponderosa pine includes western and mountain pine beetles; root diseases include laminated root rot and Armillaria root disease.

7.14 Climate Change Considerations

A pressing environmental matter of critical concern is the threat of a long-term increase in the surface temperature of the earth. This threat goes under several names – climate change and global warming are probably the most common terms. Global warming exacerbates a natural process called the ‘greenhouse effect,’ referring to the principle of a greenhouse in that the enclosing shell allows passage of incoming sunlight but traps a portion of the reflected infrared radiation, warming the greenhouse’s interior above the outside temperature. Greenhouse gases in earth’s atmosphere play a similar role to a greenhouse’s shell – they function to raise the temperature of the earth and make it habitable. Without greenhouse gases, the surface of the earth would be about 30°C (54°F) cooler than it is today, rendering human life impossible.

Since the beginning of what is termed the 'industrial era' (mid 1700s), combustion of fossil fuels, together with permanent deforestation and a few other anthropogenic activities, has led to an increase in the carbon dioxide content of the atmosphere by about 40 percent. In the last three decades alone, it has increased by almost 20 percent. An approximate doubling of carbon dioxide levels could occur by the middle of the 21st century, depending on the rate of fossil fuel burning over the next few decades. [Note: after excluding water vapor, which is the most abundant 'greenhouse gas,' carbon dioxide is currently about 77% of all remaining greenhouse gases, with others being methane (14%), nitrous oxide (8%), and several trace gases (carbon monoxide, ozone-depleting chemicals, halocarbons, etc.).]

Instrumented temperature records, along with the gas composition of ice associated with long-lived glaciers and ice fields, show that the earth has warmed about 0.7°C (1.3°F) over the past 100 years. Some climate models predict that during this century, temperatures could rise by 1.5 to 4.5°C, or about 0.3°C per decade. This might not sound like very rapid change, but historical studies have shown that past episodes of warming and cooling occurred at a rate of only about 0.05°C per decade, and some of this change was sufficient to cause major dislocations for human agrarian societies (Mann 2006).

Climate change effects are not expected to be uniform – in the northern hemisphere, polar regions will warm faster than equatorial zones, and the centers of continental landmasses are expected to become drier than the peripheries. In ice ages of the past, weather changed gradually enough to allow plants and animals to migrate and survive; the rapid pace of change occurring now may be too quick for many organisms to adjust to modified habitats. For this reason, some of the biggest impacts of climate change on humans could involve agriculture and forestry and, of the two, forestry probably has fewer mitigation or adaptation options than agriculture (narrative to this point in section 7.14 is based primarily on Karl et al. 2009).

Much of the concern about climate change relates to how it might affect baseline climate conditions. But would climate change effects be additive, subtractive, or neutral on baseline temperature and moisture relationships, and would their magnitude be great enough to exceed the environmental tolerances of existing plant species (table 11)? If the answer to the second question is yes, then one likely effect of climate change could be extirpation of certain plant species, and their related fauna and ecosystem services, from portions of the Blue Mountains.

When considering precipitation patterns, it's not just the potential for more and longer droughts in the future that is problematic – it's the projected change in precipitation form, with less being received as snow and more as rain (fig. 29). Modeling suggests this trend might improve forest growth in the short-term because the growing season would lengthen into early spring, which could possibly synchronize spring rains with tree growth initiation (now, tree growth tends to commence in late spring after rainfall has ceased). But since the Blue Mountains occur in a 'summer-dry' zone where soil storage of snowmelt is critical for sustaining plant growth, this change in precipitation form could induce earlier summer plant dormancy, lengthen fire season, shorten the wetland saturation period, and affect other ecosystem services (van Mantgem et al. 2009).

Table 11: Selected life history traits for four primary conifers of dry forests.

	Ponderosa pine	Western larch	Douglas-fir	Grand fir
Tolerance to shading	L	L	M	H
Tolerance to full sunlight	H	H	M	L
Seral status	Early	Early	Mid	Late
Tolerance to frost	L	L	L	M
Tolerance to drought	H	M	M	M
Rooting habit (depth)	D	D	D	S
Fire resistance	H	H	M/H	L/M
Evolutionary mode	Inter.	Inter.	Spec.	N.R.
Seed germination on charred or ashy soil	IN	NE	IN	IN
Maximum seed dispersal distance (feet)	120	150	330	200
Potential for regeneration in the open	H	H	H	L
Overall reproductive capacity	H	H	H	M
Potential initial growth rate (first 5 years)	H	H	M	M

Sources/Notes: Ratings derived from a variety of literature sources. Rating codes are: L, low; M, moderate; H, High; D, deep; S, shallow; IN, increased; and NE, no effect. Overall reproductive capacity considers minimum cone-bearing age, seed crop frequency and size, seed soundness, and related factors. Evolutionary mode refers to the amount of genetic differentiation and is an indicator of how well a species could adapt to future climates (Inter. is intermediate; Spec. is specialist; N.R. is not rated; source = Rehfeldt 1994).

Certain life history traits in table 11, such as 'tolerance to frost,' might seem unrelated to climate change. But the cold hardiness of trees has apparently been influenced by climate change, with boreal forests experiencing earlier loss of cold hardiness in response to early-spring warming (late April to early May), followed by severe frost damage during subsequent cold snaps in mid spring (mid to late May) (Man et al. 2009). Before the onset of climate change, frost damage in mid May was an unusual event because boreal trees had not lost cold hardiness at that point in the spring.

The ecological changes described earlier in this white paper, as related to fire exclusion, ungulate herbivory, and selective cutting (Harrod et al. 1999, Mast et al. 1999, Sloan 1998, Turner and Krannitz 2001), have put dry-forest ecosystems on precisely the wrong trajectory when considering the warm, fire-favoring climate expected in the 21st century (Brown et al. 2004, Flannigan et al. 2005, Gillett et al. 2004, Macias Fauria and Johnson 2006, Miller et al. 2009, Running 2006, Spracklen et al. 2007, van Mantgem et al. 2009, Westerling et al. 2006).

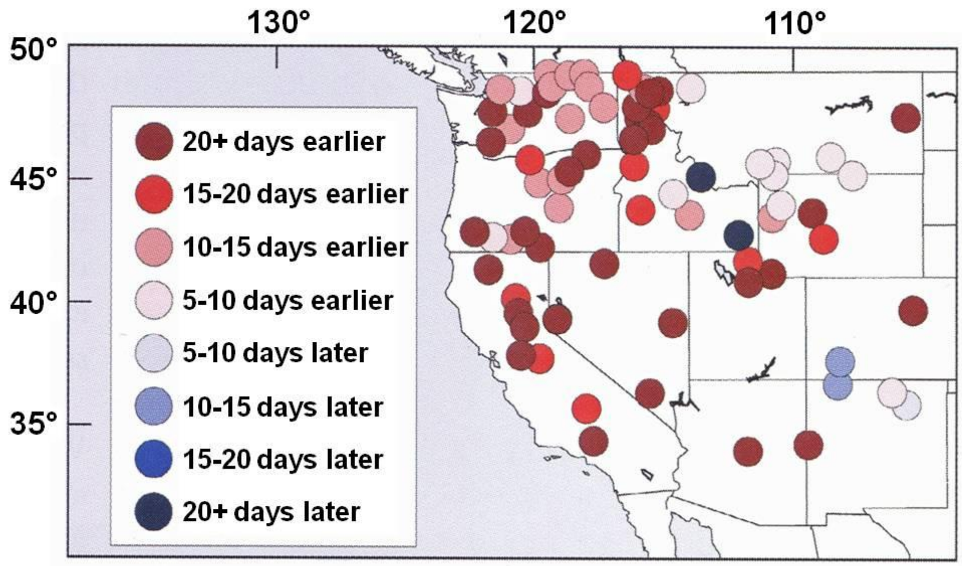


Figure 29 – Recent changes in spring snowmelt timing for the western United States (from Karl et al. 2009, page 33). This chart shows trends in streamflow runoff timing for 1948-2000, as the number of days runoff occurs earlier. According to this analysis, river basins of the northern Blue Mountains occur in a zone where runoff occurred 10-20+ days earlier for the 1948-2000 period than it did previously. Future climate change is expected to exacerbate this trend (Furniss et al. 2010, Stewart et al. 2004).

If dry mixed-conifer forests are to have a reasonable opportunity for persistence under the future climate regime, restoring conditions more similar to the historical characteristics of the frequently burned, open forests of the past is likely to function as a useful startpoint (Fiedler 2000b, Harrod et al. 1999, Munger 1917). Sustainable, dry-forest conditions can be achieved by reintroducing surface fire to change fire-free intervals from centuries to decades, reduce surface fuels, thin canopy and ladder fuels, counteract a species composition trend toward reduced representation by fire-resistant trees (fig. 30), and reestablish spatial heterogeneity (fig. 31).

Forest trees in a superior canopy position have better access to sunlight, nutrients, and moisture than trees in subordinate canopy positions. Since dominant trees use a disproportionate share of site resources, it seems logical that they are little influenced by subordinate trees (Daniel et al. 1979, Smith et al. 1997). Research in central Oregon (Barrett 1963, 1972) and elsewhere in the West (Dolph et al. 1995, Stone et al. 1999), however, showed that competition functions as a two-way process, and that removing subordinate trees (particularly the intermediate and subcanopy crown classes) results in dramatic vigor increases for dominant ponderosa pines (Woodall et al. 2003), particularly in old-growth stands and during drought periods.

Note that dry-forest restoration activities are envisioned for implementation on Blue Mountain areas **currently** classified as dry upland forest; no attempt has yet been made to predict how this biophysical environment might expand, contract, or migrate in response to future climate change. Although any attempt to model how the Dry Upland Forest (UF) PVG might increase at the expense of the Moist UF or Cold UF PVGs would be speculative at this point, several of the climate change scenarios examined for the interior Pacific Northwest suggest that this is a likely outcome (Dello and Mote 2010).

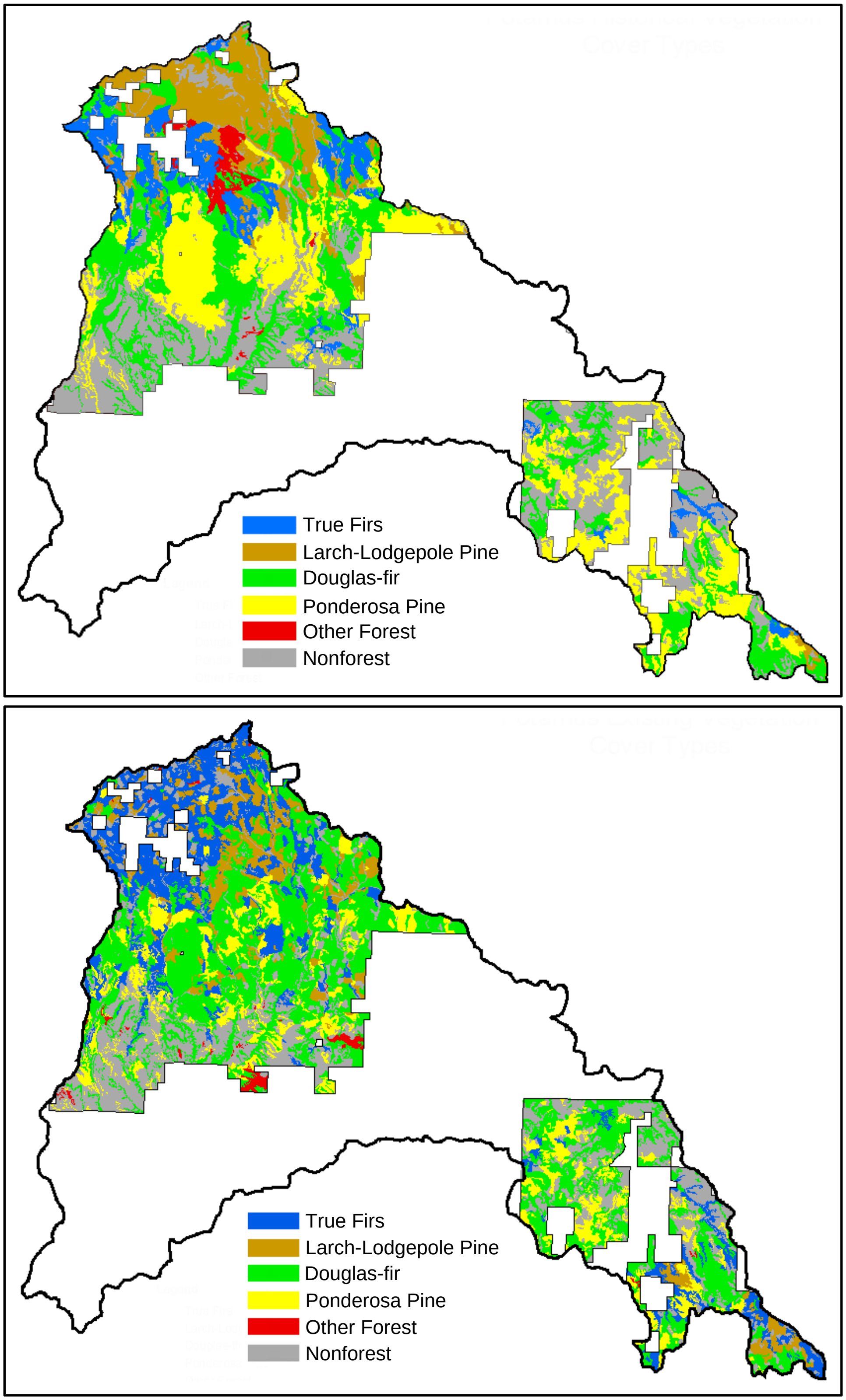


Figure 30 – Historical (upper) and existing (lower) vegetation cover types for the Potamus watershed, Heppner Ranger District. Note that when comparing the two maps, ponderosa pine declined, and Douglas-fir/true firs increased, through time.

Figure 31 – Reintroduction of spatial heterogeneity into a portion of the Wild Horse prescribed fire project area on the Heppner Ranger District. Experience in the Blue Mountains over the past 20 years suggests that adopting a very conservative approach to restoration of dry forests is not a choice of 'no action' because such a strategy accepts the risk of high-severity wildfire and other uncharacteristic disturbance events (see fig. 15). Upon recognizing that the risks of no action are probably unacceptable for many scenarios, land managers must design flexible, adaptive treatments to restore high levels of spatial heterogeneity for dry forests (Allen et al. 2002; Hessburg et al. 2000, 2007; Wright and Agee 2004). And if the scale of management activity does not emulate the scale of native disturbance processes, then we can expect ecosystem responses such as reduced biological diversity and impaired nutrient cycling (Baydack et al. 1999, Eng 1998).

There is also no assurance that the current amount and spatial configuration of dry forest will remain the same under climate change. Research suggests that changes in fire regimes due to climate feedbacks led to the expansion of savanna environments (open tree stands whose physiognomy is more reminiscent of grassland than forest) in response to hotter and drier conditions (Bond et al. 2005, Bowman et al. 2009). Based on the circumstances under which it has occurred elsewhere, a savanna outcome is certainly plausible for some proportion of the dry-forest acreage located within the Blue Mountains ecoregion.

Many of the policy proposals being considered to address climate change are based on mitigation – reducing greenhouse gas emissions from energy use and land-use changes in order to minimize the pace and magnitude of climate change. While mitigation is crucial, adaptation to climate change is increasingly viewed as a necessary and complementary strategy to mitigation (Joyce et al. 2009). Table 12 provides adaptation strategies proposed for the National Forest System and pertaining to upland forest vegetation. Table 12 also describes the predicted compatibility of active management treatments with climate change adaptation strategies. Note: some sources frame the mitigation/adaptation concept as resistance/resilience – near-term measures to increase resistance (thinning) need to be coordinated with longer-term resilience strategies.

Table 12: Estimated compatibility of climate change adaptation strategies and active management of dry upland forests.

Climate Change Adaptation Strategies	Compatibility With Dry-Forest Management
Improve the capability of ecosystems to withstand uncharacteristically severe drought, wildfires, and insect outbreaks at landscape scales.	Thinning and similar active management practices might be necessary to improve the resistance and resilience of dry-forest vegetation, upon which many ecosystem services depend.
Facilitate natural (evolutionary) adaptation through silvicultural treatments that shorten regeneration times and promote interspecific competition.	Adaptation strategies often recommend regeneration cutting because existing stands are adapted to century-old climates, so new seedlings would then become adapted to future (changed) climates.
Where ecosystems will very likely become more water limited, manage for drought- and heat-tolerant species.	When circumstances permit, composition could be changed to favor species with high tolerance to drought, open conditions, and fire (table 11).
Reduce homogeneity of stand structure and synchrony of disturbance patterns across broad landscapes by promoting diverse age classes and species mixes, stand diversities, and genetic diversity.	This strategy could best be addressed by perpetuating age-class diversity, introducing additional species diversity when appropriate, and trying new genotypes offering better environmental fitness.
Reset ecological trajectories to take advantage of early successional stages that are adaptive to present rather than past climates.	Composition could be changed to favor early-seral species with high tolerance or resistance to drought, open conditions, and fire (table 11).
Use historical ecological information to identify environments buffered against climate change and which would be good candidates for conservation.	Many literature sources provide historical information with relevance for dry-forest ecosystems (Gannett 1902, Munger 1917, and others).
Encourage local industries that can adapt to or cope with variable types of forest products because of the uncertainty about which tree species will prosper in the future.	Small-diameter trees will be removed periodically as restoration activities are implemented and, depending on the circumstances, they could be used for biomass purposes.
Reforestation after disturbance may require different species than were present before the disturbance to better match site-level changes associated with climate change.	We can use life-history data such as fire resistance and drought tolerance (table 11) to reforest with species having high resilience to future climates. But should we also consider new species?
After a disturbance event, use intensive site preparation activities to remove competing vegetation and replant with high-quality, genetically appropriate and diverse stock.	This recommendation is similar to the one just before it, but with additional detail. It is feasible to use site preparation before planting, but any 'intensive' measures need to protect soil integrity.
To promote climate resilience for existing stands, use widely spaced thinnings or shelterwood cuttings and rapid response to forest mortality from fire or insects.	Wide thinning spacings and shelterwood seed cuttings are compatible with dry upland forests. A rapid response to mortality would help address increased fire and insect risk as related to climate change impact.
Plan for higher-elevation insect outbreaks, species mortality events, and altered fire regimes.	It is expected that some fire regime 3 (mixed-severity) areas could transition to fire regime 1 (low severity) as the future climate warms and dries.

Sources/Notes: Adaptation strategies pertain to forest environments only, and were derived from Joyce et al. (2008, 2009) and West et al. (2009). Only forest-centric strategies were addressed in this table.

The information in table 12 suggests that active management practices reducing stand vulnerability to uncharacteristically severe wildfire and other climate-influenced disturbance processes could satisfy multiple goals of near-term mitigation (by minimizing fire-related carbon emissions) and mid-term adaptation if such practices also reflect goals for other ecosystem services such as late-old structure and water quality (Joyce et al. 2009). The potential for uncharacteristically severe wildfire is particularly high – of the 47 million acres of federal land in the Pacific Northwest, approximately 47 percent (22.6 million acres) were historically affected by short interval fire (these are dry sites once dominated by ponderosa pine, shrubs, or bunchgrasses). The majority of these lands are located east of the Cascade Mountains in Washington and Oregon. Of the acres with a short-interval fire regime, 71 percent (16 million acres) currently have a higher predicted fire severity (risk) than would have existed historically (fig. 32).

Future activities could be designed to favor species whose life-history traits are most compatible with future climatic conditions. These traits are presented in table 11 for four primary species of dry upland forest sites. But even so, we also need to realize that as stock brokers like to say: "past performance is no guarantee of future success." Proposed restoration activities are expected to improve the adaptive capacity (Olsson et al. 2004) of dry mixed-conifer forests in the Blue Mountains, particularly by alleviating the chronic stress associated with high tree density levels and reestablishing an historically appropriate structural condition (fig. 33). The general goal of dry-forest restoration across the western United States is to develop more open-stand structures consistent with the historical disturbance regime (Arno et al. 1995), and this outcome is now considered to be most compatible with a warmer and dryer future (Brown 2008).

Climatic drought is projected to be more common in the future because mid-summer temperatures are expected to be higher, and summer precipitation amounts lower, than at present. Dense stands exist in a sort of perpetual physiological drought because there is not enough soil moisture to meet the water needs of all trees; silvicultural treatments are used to alleviate this moisture stress and allow residual trees to survive and continue growing. Since climate change could amplify the effects of density-caused stress, the need for thinning is expected to be much greater in the future than at present, particularly because thinning improves physiological vigor, and trees with improved vigor produce more of the resins used to repel insect and disease attacks (Kolb et al. 1998, Langenheim 1990, Mitchell et al. 1983, Nebeker et al. 1995, Phillips and Croteau 1999, Pitman et al. 1982, Safranyik et al. 1998).

The direct effects of climate change on temperature and precipitation, in combination with indirect effects related to wildfires, insect outbreaks, and other disturbance processes, could detrimentally affect the future provision of old-forest structure, properly functioning soil and water services, wildlife habitat, animal and plant diversity, recreational opportunities, and carbon storage (fig. 34). Climate modeling, for example, suggests that western larch could be extirpated from the Blue Mountains by early in the 22nd century (Rehfeldt et al. 2006). Obviously, changes of this magnitude could cause substantial 'ripple effects' across many biological webs and trophic levels (Perry et al. 2008). If climate change precludes us from sustaining our current level of ecosystem components (composition, structure, density), then how can we expect to sustain the ecosystem services relying on these components?

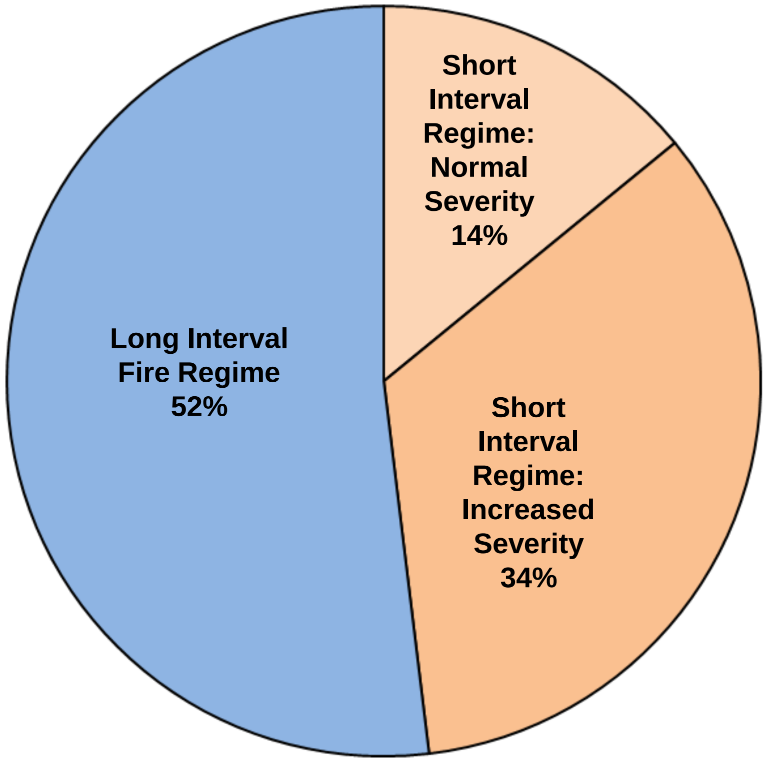


Figure 32 – Trend toward increasing fire susceptibility for short-interval fire regimes of the Pacific Northwest.[1] Many sources show that the scale of predicted change in fire severity for fire regime 1 sites has been enormous (Hessburg et al. 2005, Hann et al. 1997, Quigley et al. 1996, and other sources in the References section). When considering the 47 million acres of federal lands in the Pacific Northwest, which are administered by the Bureau of Indian Affairs, Bureau of Land Management, National Park Service, U.S. Fish and Wildlife Service, and U.S. Forest Service, about 48% of the total acreage can be assigned to a short-interval fire regime (primarily fire regimes 1 and 2). Of this dry-site acreage, 71 percent (16 million acres, or 34% of the total acreage) currently has a higher predicted fire severity (stand-replacing) than would have existed historically (stand-maintaining severity). When the General Accounting Office evaluated catastrophic fire risk for the western U.S., its report concluded that "the most extensive and serious problem related to the health of national forests in the interior West is the over-accumulation of vegetation." GAO estimated that about 39 million acres of national forests in the West have high fire risk due to excessive fuel buildup; they estimated that $12 billion would be needed between 1995 and 2015 to reduce excess fuel accumulations, an average expenditure of $725 million annually (GAO 1999).

Mitigation. A near-term climate change strategy adopting tactics such as reducing greenhouse gas emissions (by reducing wildfire emissions, for example), or by enhancing carbon uptake and storage. **Adaptation.** A far-term climate change strategy adopting tactics such as minimizing negative ecosystem effects (reforest now with tree species expected to be tolerant of future droughts), or by exploiting potential opportunities. **Both** strategies are important, in some combination, to address climate change.

[1] This figure includes federal lands administered by the Bureau of Indian Affairs, Bureau of Land Management, National Park Service, U.S. Fish and Wildlife Service, and the U.S. Forest Service (data derived from a draft report released by the Pacific Northwest Wildfire Coordinating Group in June 2000; 28 p.).

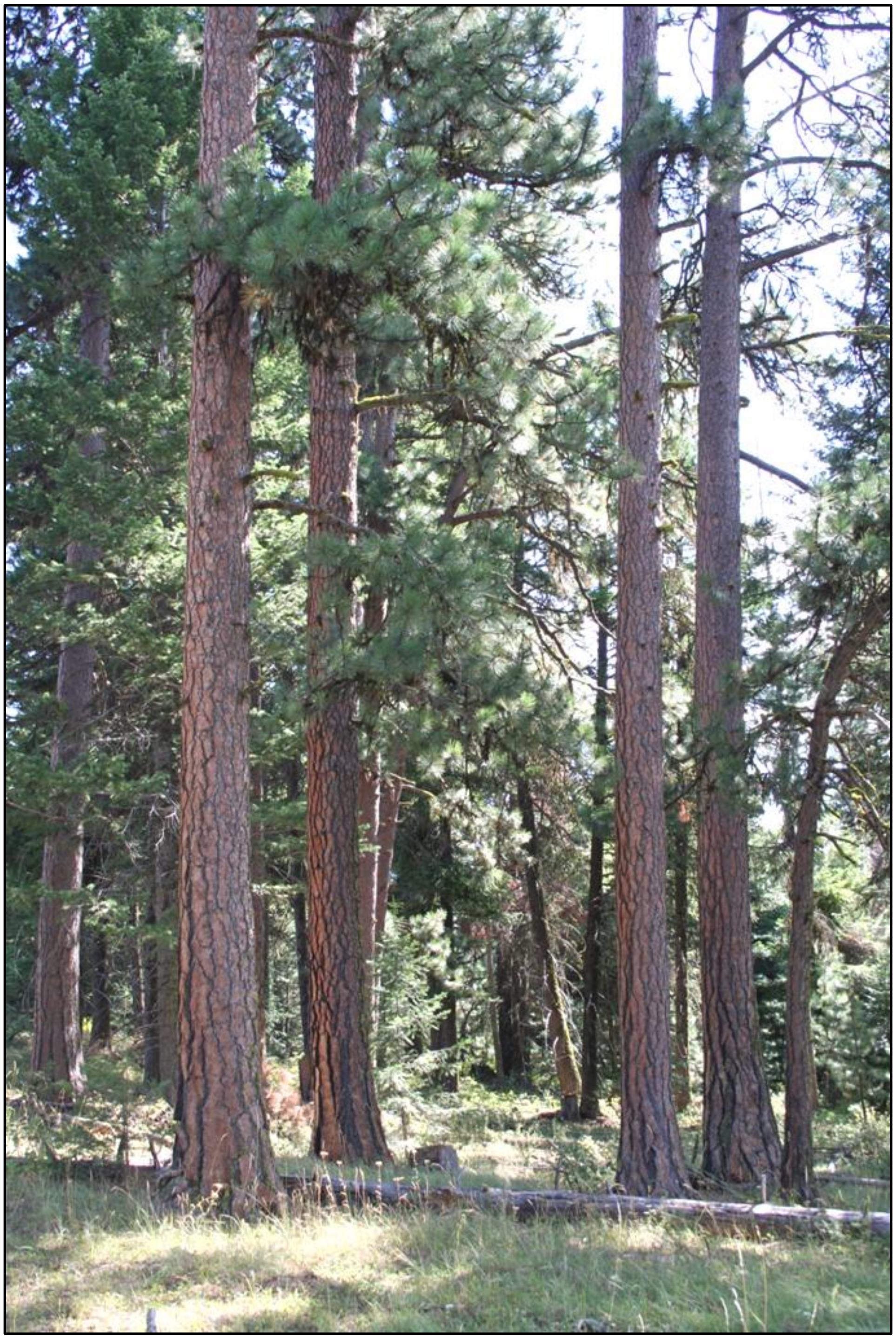


Figure 33 – Many dry-forest sites still have remnant historical structure such as this clump of mature ponderosa pine. Historically, dry forests tended to be uneven-aged at the stand level, with stands consisting of small even-aged patches, each differing in age from the others (Cooper 1960, 1961a; Munger 1917; White 1985). Historical patch sizes for dry forests ranged from 0.01-0.05 acres for central Washington (Harrod et al. 1999) to 0.37-0.44 acres for central Oregon and northeastern California (Youngblood et al. 2004). Restoration treatments can be used to remove the late-seral species (Douglas-fir and grand fir) from this group, thereby improving its resistance and resilience to climate change.

Figure 34 – Fenced clone of quaking aspen located along the 5316 road on the North Fork John Day Ranger District, Umatilla National Forest (aspen is the short vegetation with yellow and green foliage located just behind the large ponderosa pines). Aspen reproduces almost entirely from root suckers, resulting in a clonal life history where the root system functions as what is referred to as the genet, and it produces successive generations of root suckers called ramets. The ramets develop into mature trees. Although an individual cohort of aspen ramets is relatively short-lived (60 to 100 years is common), the underground genet may be thousands of years old. Some clones in the intermountain West might approach 10,000 years of age (and perhaps more than a million years according to Barnes 1975), thus producing a hundred or more generations of ramets from a single root system. Recent genetic testing, for example, indicates that an ancient aspen clone has existed for perhaps thousands of years in the Morsay Creek drainage of the North Fork John Day Ranger District (Shirley and Erickson 2001).

Aspen is classed as a very intolerant tree species (Daniel et al. 1979, p. 297), which means it can regenerate and develop acceptably in open environments only. When competing with other species that are more tolerant than it is, particularly conifers, aspen will quickly lose vigor as shading, soil acidity, and other environmental conditions evolve to favor the competitors. This means that one restoration tactic for maintaining and sustaining aspen in the Blue Mountains is to remove invading conifers. This practice is controversial, particularly when the competing conifers are old ponderosa pines (but, 200-year-old pines are obviously younger than a 1000-year-old aspen root system).

The photograph also shows buck-and-pole, A-frame style fencing installed around the aspen clone (including both its current extent, and some expansion space) as a way to address ungulate herbivory caused primarily by cattle and elk. The concept of fencing is to exclude ungulates for a long enough period to allow aspen suckers to reach a large enough size (in both height and stem caliper) where they can withstand some browsing pressure and still develop into a viable overstory cohort.

Although aspen is not an obligate riparian species, it tends to occupy swales and other landforms accumulating subsurface soil moisture. Aspen has a surprising affinity for dry-forest environments in the Blue Mountains (e.g., it occupies moist microsites within a broader warm dry biophysical setting), as illustrated by a recent classification of quaking aspen types – of the 15 aspen plant community types identified, 8 of them occur in the dry-forest potential vegetation group (see appendix; Swanson et al. 2010).

ACKNOWLEDGMENTS

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PHOTO CREDITS

The photographs used for figures 7, 9, 15, 21, 23, 27, 31, 33, and 34 were all acquired by David C. Powell. The white-headed woodpecker photograph used in Box 1 was acquired by Al and Elaine Wilson (www.BirdsISaw.com). Historical, black-and-white photographs (figures 5, 12, and 19) are acknowledged in the figure captions. Unless noted otherwise in the figure caption, non-photographic charts, diagrams, or images were prepared by David C. Powell.

APPENDIX

The dry upland forest PVG includes dry mixed-conifer forests occurring in the lower montane vegetation zone (see fig. 3). Portions of three potential vegetation series are represented in the dry upland forest PVG – grand fir, ponderosa pine, and Douglas-fir. Note that quaking aspen plant community types, which are successional (non-climax) stages of a plant association, are also common in the dry upland forest PVG – eight aspen types are included in the list below.

Only three grand fir plant associations are included in the dry upland forest PVG (two 'sod-grass' types: the elk sedge and pinegrass associations, and the birchleaf spiraea type), but they occupy substantial acreage in the central and southern Blue Mountains. Douglas-fir plant associations are well represented in this potential vegetation group, with Douglas-fir/low shrub types being especially common (the snowberry, birchleaf spiraea, and ninebark associations).

Although ponderosa pine is a ubiquitous species in the Blue Mountains, it is the climax species on a surprisingly small percentage of the area (certainly less than 10% in the northern Blue Mountains, but a higher percentage than that for the southern Blue Mountains). Many ponderosa pine plant associations were described for the Blue Mountains, and all of them were assigned to this potential vegetation group (Powell et al. 2007), indicating that the environmental tolerances of ponderosa pine do not allow it to become dominant on cold or moist forest sites.

Potential vegetation type (PVT) codes and names, and plant association group (PAG) assignments, for the dry upland forest potential vegetation group (PVG).[1]

PVT Code	PVT Name	PAG
ABGR/CAGE	grand fir/elk sedge	warm dry
ABGR/CARU	grand fir/pinegrass	warm dry

PVT Code	PVT Name	PAG
ABGR/SPBE	grand fir/birchleaf spiraea	warm dry
JUSC/CELE	Rocky Mountain juniper/mountain mahogany	warm dry
PIPO/AGSP	ponderosa pine/bluebunch wheatgrass	hot dry
PIPO/ARAR	ponderosa pine/low sagebrush	hot moist
PIPO/ARTRV/CAGE	ponderosa pine/mountain big sagebrush/elk sedge	hot dry
PIPO/ARTRV/FEID-AGSP	pond. pine/mtn. big sage/Idaho fescue-bluebunch wheatgrass	hot dry
PIPO/CAGE	ponderosa pine/elk sedge	warm dry
PIPO/CARU	ponderosa pine/pinegrass	warm dry
PIPO/CELE/CAGE	ponderosa pine/mountain mahogany/elk sedge	warm dry
PIPO/CELE/FEID-AGSP	pond. pine/mtn. mahog./Idaho fescue-bluebunch wheat.	hot dry
PIPO/CELE/PONE	ponderosa pine/mountain mahogany/Wheeler bluegrass	hot dry
PIPO/FEID	ponderosa pine/Idaho fescue	hot dry
PIPO/PERA	ponderosa pine/squaw apple	hot dry
PIPO/PUTR/AGSP	ponderosa pine/bitterbrush/bluebunch wheatgrass	hot dry
PIPO/PUTR/AGSP-POSA	pond. pine/bitterbrush/bluebunch wheat./Sandberg bluegrass	hot dry
PIPO/PUTR/CAGE	ponderosa pine/bitterbrush/elk sedge	warm dry
PIPO/PUTR/CARO	ponderosa pine/bitterbrush/Ross sedge	warm dry
PIPO/PUTR/FEID-AGSP	pond. pine/bitterbrush/Idaho fescue-bluebunch wheat.	hot dry
PIPO/RHGL	ponderosa pine/smooth sumac	hot dry
PIPO/SPBE	ponderosa pine/birchleaf spiraea	warm dry
PIPO/SYAL	ponderosa pine/common snowberry	warm dry
PIPO/SYOR	ponderosa pine/mountain snowberry	warm dry
PIPO-JUOC/CELE-SYOR	pond. pine/western juniper/mtn. mahog.-mtn. snowberry	hot dry
POTR5/CAGE2	aspen/elk sedge	warm dry
POTR5/CARU	aspen/pinegrass	warm dry
POTR5/EXOTIC GRASS	aspen/exotic grass	warm dry
POTR5/PRVI	aspen/chokecherry	warm dry
POTR5 (RUBBLE, LOW)	aspen (rubble, low)	warm, dry
POTR5(ABGR)/SYMPH	aspen(grand fir)/snowberry	warm dry
POTR5(PIPO-PSME)/SYMPH	aspen(ponderosa pine-Douglas-fir)/snowberry	warm dry
POTR5(PSME)/PREM	aspen(Douglas-fir)/bitter cherry	warm dry
PSME/ARNE/CAGE	Douglas-fir/pinemat manzanita/elk sedge	warm dry
PSME/CAGE	Douglas-fir/elk sedge	warm dry
PSME/CARU	Douglas-fir/pinegrass	warm dry
PSME/CELE/CAGE	Douglas-fir/mountain mahogany/elk sedge	warm dry
PSME/PHMA	Douglas-fir/mallow ninebark	warm dry
PSME/SPBE	Douglas-fir/birchleaf spiraea	warm dry
PSME/SYAL	Douglas-fir/common snowberry	warm dry
PSME/SYOR	Douglas-fir/mountain snowberry	warm dry
PSME/SYOR/CAGE	Douglas-fir/mountain snowberry/elk sedge	warm dry
PSME/VAME	Douglas-fir/big huckleberry	warm dry
PSME-PIPO-JUOC/FEID	Douglas-fir/ponderosa pine/western juniper/Idaho fescue	warm dry

[1] Potential vegetation type codes and names, and plant association group assignments, were taken from Powell et al. (2007) except for the aspen community types, which came from Swanson et al. (2010).

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