WHITE PAPER

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Active Management of Blue Mountains Dry Forests: Silvicultural Considerations

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Silvicultural treatment in dry forest to restore old forest single stratum structure

CONTENTS

1. INTRODUCTION

A Blue Mountains ecoregion extends from Ochoco Mountains in central Oregon to Hells Canyon of Snake River in extreme northeastern Oregon, and then north to deeply carved canyons and basalt rimrock of southeastern Washington (fig. 1).

An objective of this white paper is to discuss silvicultural considerations associated with active management of Blue Mountains dry forests.

A companion white paper (F14-SO-WP-Silv-7) discusses silvicultural considerations for active management of Blue Mountains moist upland forests (Powell 2019a).

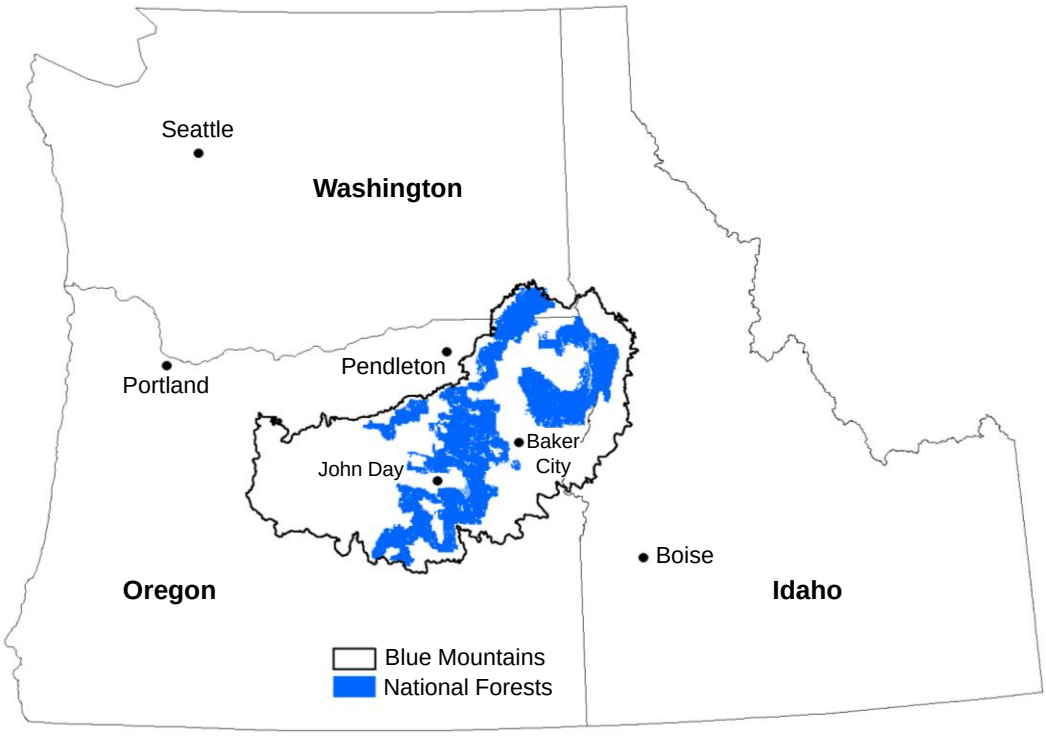


Figure 1 – Blue Mountains ecoregion of northeastern Oregon, southeastern Washington, and west-central Idaho. This ecoregion consists of a series of mountain ranges in a southwest to northeast orientation, extending from Ochoco Mountains in central Oregon, southwestern portion of the ecoregion, to western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

Beginning in mid-1960s, 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. Blue Mountains eventually gained a dubious distinction of having perhaps the worst forest health in 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 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, 1990s scientific assessments concluded that:

- In 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 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 also occurring throughout interior Pacific Northwest and elsewhere in western United States, particularly for dry forest environments (Everett et al. 1994; Hessburg et al. 1994, 1999a; Lehmkuhl et al. 1994; O'Laughlin et al. 1993, Oliver et al. 1994c; Quigley et al. 1996; Sampson and Adams 1994).
- Fine-scale project planning corroborated findings from broad-scale assessments by suggesting that certain symptoms of impaired forest health (such as uncharacteristic wildfire and insect effects) were largely related to species composition, forest structure, and tree density being outside their historical range of variation. Once again, this finding pertained mostly to dry-forest portions of Blue Mountains ecoregion.

We know that many of our fire-dependent, dry-forest ecosystems are deteriorated, with wildfire and other disturbance processes behaving much differently now than they did historically. This white paper 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 an ecological setting and historical context for dry-forest ecosystem components (species composition, forest structure, and tree density).
2. Identify some factors (fire suppression, ungulate herbivory, selective cutting) that may have contributed to dry-forest ecosystem changes through time.
3. Describe what needs to be done to restore dry-forest ecosystem components.
4. Develop criteria for measuring success of restoration activities.

Initial sections of this white paper characterize an ecological setting and provide an historical narrative for dry forests. Middle sections examine how fire exclusion, plant succession in an absence of recurrent fire, domestic and native ungulate herbivory, and selective timber harvest allowed historically high resilience of dry-forest ecosystems to erode 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 southern half of Umatilla National Forest and elsewhere in the central and southern Blue Mountains, and to a lesser extent on northern half of Umatilla National Forest and elsewhere in the northern Blue Mountains (fig. 2, table 1).

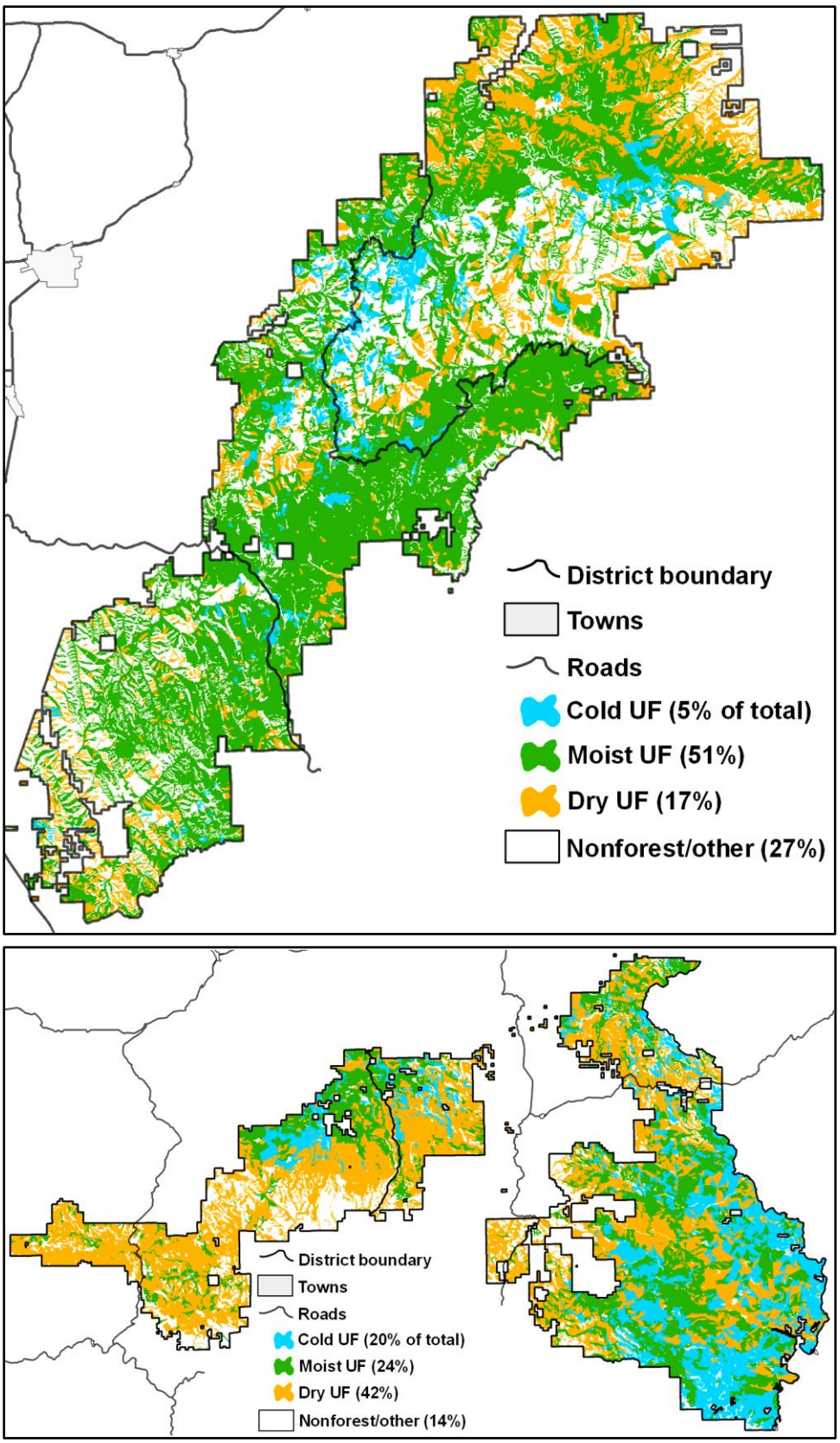


Figure 2 – Distribution of upland forest (UF) potential vegetation groups on Umatilla National Forest (north-end districts above; south-end districts below).

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

Potential Vegetation Group	North Half	South Half	Total
Cold Upland Forest	34,832 ac (21%)	132,314 ac (79%)	167,145
Pct. Of Forested	7%	23%	15%
Pct. Of Total	5%	20%	12%
Moist Upland Forest	368,847 ac (70%)	162,283 ac (30%)	531,130
Pct. Of Forested	70%	28%	48%
Pct. Of Total	51%	24%	38%
Dry Upland Forest	123,129 ac (30%)	286,316 ac (70%)	409,445
Pct. Of Forested	23%	49%	37%
Pct. Of Total	17%	42%	29%
Nonforest	201,481 ac (68%)	94,667 ac (32%)	296,147
Pct. Of Total	27%	14%	21%

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

Appendix 1 provides a list of potential vegetation types (plant associations, plant community types, plant communities) occurring in a dry upland forest potential vegetation group.

Appendix 1 demonstrates that to establish a context for this white paper, dry forest is defined by using units of potential vegetation (e.g., plant associations, plant community types, plant communities), rather than by adopting an alternative approach relying on categories of existing vegetation (such as ponderosa pine stands or cover types, etc.).

This strategy for defining dry forest is necessary because potential vegetation reflects site potential – vegetation types a site can support under contemporary climate and its associated temperature and moisture regimes. Existing vegetation, however, describes what is present now, regardless of whether it represents climatic (permanent) vegetation or temporary types resulting from wildfire, timber harvest, ungulate grazing, and other disturbance processes.

Formatting note: glossary terms are dispersed throughout this white paper by separating them from text in gray-shaded sections (see first example, below). This approach was adopted to provide definitions in the chapter in which a term is first used, rather than combining all glossary terms in one section at the end of this document.

Active management. Human intervention into nature, extent, and timing of disturbance to forest ecosystems for the purpose of obtaining desired goods and services (Haeussler and Kneeshaw 2003).
Resilience. Intrinsic properties allowing 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, a system may rapidly transform to a different state or developmental trajectory (Gunderson et al. 2010).

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 by 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).

A 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, 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 an 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 effect of fire, insects, pathogens, and other disturbance agents 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.

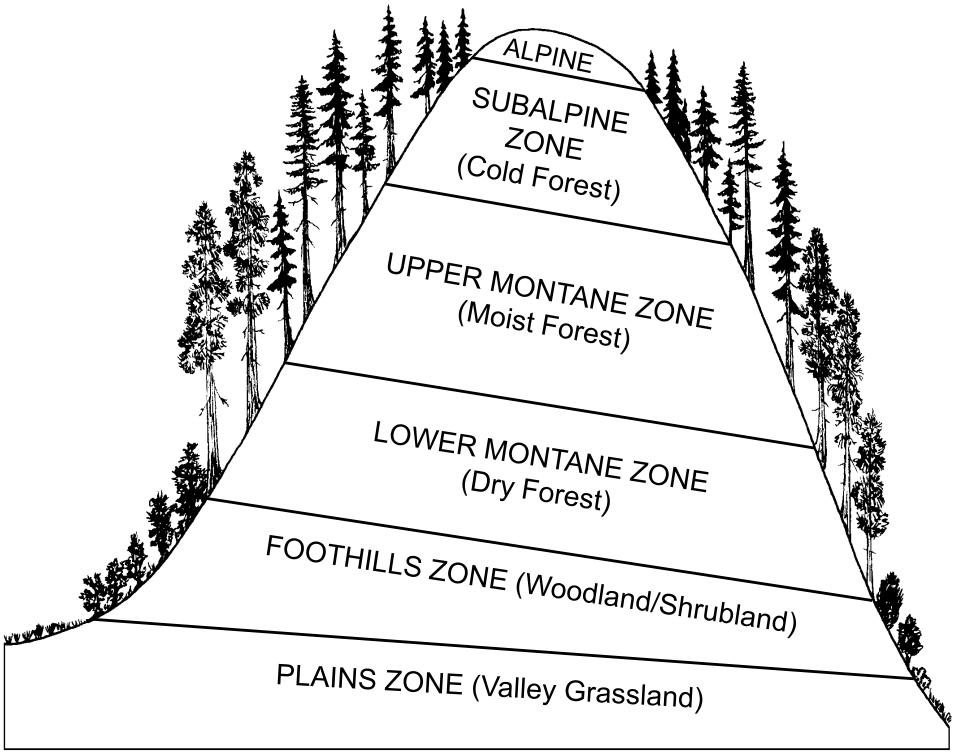


Figure 3 – Vegetation zones of the Blue Mountains. In the northern hemisphere, a south-facing slope receives more solar radiation than a flat surface, and a north-facing slope receives less (south slope is to the left, and north is to the right). These solar radiation patterns result in vegetation zones 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).

A plains zone contains grasslands and shrublands because moisture is too low to support forests except along waterways. A foothills zone is usually dominated by western juniper, often with a mixture of mountain-mahogany shrublands. Located just above western juniper woodlands is a lower montane zone containing 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 1). An 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).]

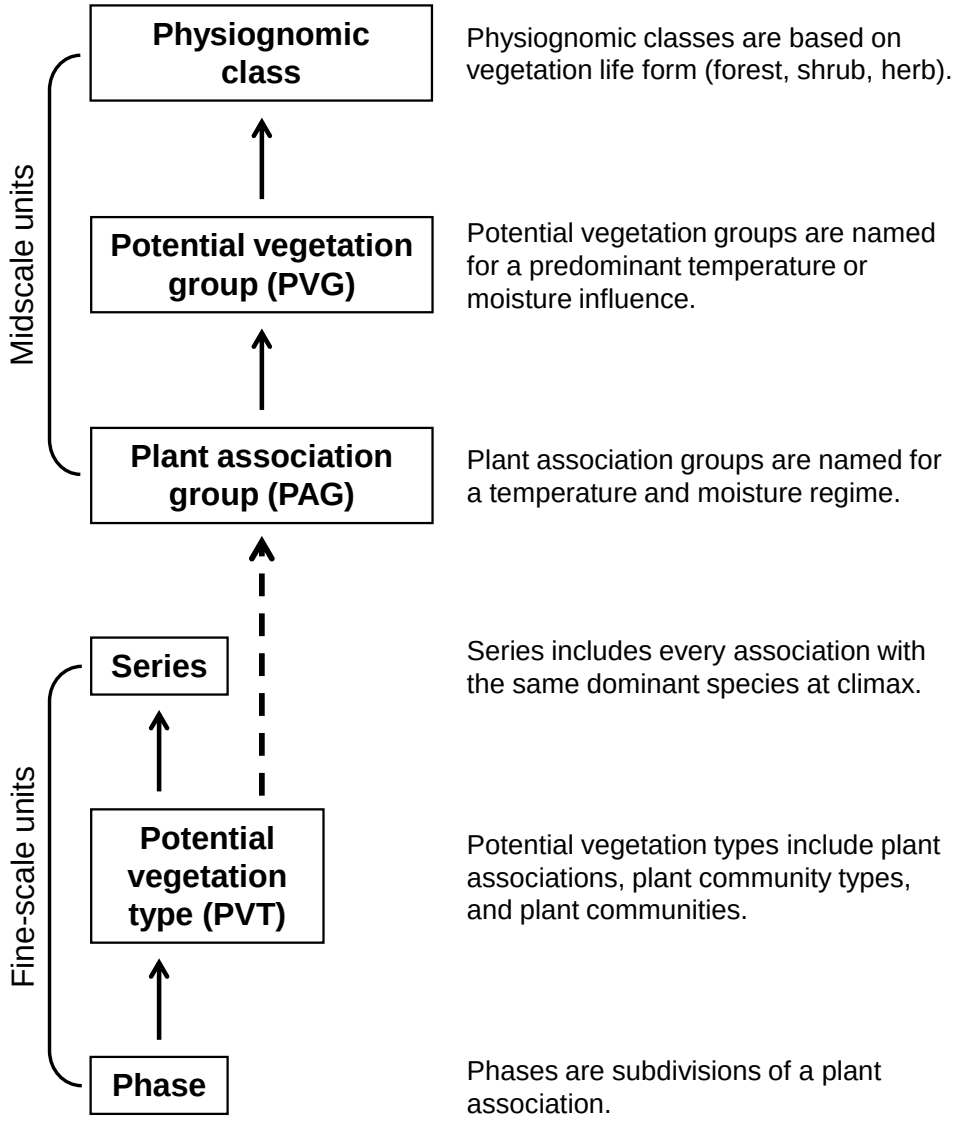


Figure 4 – Hierarchy of potential vegetation (PV) for the Blue Mountains (from Powell et al. 2007). PV taxonomic units have been organized as two integrated portions of a hierarchy. The fine-scale hierarchical units are described in PV classification reports and their associated keys (Crowe and Clausnitzer 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 biophysical environment covered by this white paper is referred to as the Dry Upland Forest potential vegetation group (PVG). Dry Upland Forest is one of three potential vegetation groups occurring in the Upland Forest physiognomic class (the other two PVGs in this physiognomic class are Moist Upland Forest and Cold Upland Forest). PVT codes and names, and the plant association group that each of the Dry Upland Forest PVTs has been assigned to, are provided in appendix 1 of this white paper. [As illustrated in this figure, plant association groups (PAGs) occur at a lower level in the hierarchy than PVGs – the Dry Upland Forest PVG contains three PAGs, each of which is named for a combination of temperature and moisture conditions – warm dry (by far the most common PAG in the Dry Upland Forest PVG), hot moist, and hot dry.]

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 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 below the montane zone. Dry forests are adjoined by moist upland forests at their upper edge, and by woodlands and shrublands of a foothills vegetation zone at their lower edge (fig. 3).

For the Blue Mountains, a Dry Upland Forest PVG consists of three plant association groups (PAG) – one from a warm temperature regime (Warm Dry PAG), and two from a hot temperature regime (Hot Moist and Hot Dry PAGs). Of the three PAGs, Warm Dry is most common for the Dry UF potential vegetation group. A 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 of 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.

Why Discuss Dry Forest As A Separate Entity?

Why prepare a white paper focused just on dry forests? After all, dry forests exist in a mosaic – sometimes they are a dominant landscape element (matrix) and, at other times, they exist as patches within a moist-forest matrix. From many perspectives, it makes sense to examine the entire mosaic rather than its constituent parts (dry forest, moist forest, etc.). Dry-forest ecosystems, however, are molded by disturbance processes differing in important ways from those shaping moist- or cold-forest environments. Thus, the science informing dry forests also differs from the science relating to moist-forest management. This truth is demonstrated by examining the dry- and moist-forest white papers – there is little science overlap between them (except for science pertaining to concepts and principles), and substantial overlap would not be expected when considering their ecological and historical (management) settings.

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 their 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 similar to 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, pine forests were open and park-like with grass and herbs as predominant undergrowth vegetation, and fire was a common occurrence in late summer and autumn (Evans 1991, Wickman et al. 1994). We can surmise that a 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). Two journal entries below (Beckham 1991, Evans 1991), however, speculate that their name commemorates a blue-green color imparted by extensive pine stands. In either case, fire was an important reason for a ‘Blue Mountains’ name because it was not only responsible for smoke, but

also for maintaining 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 USDA Forest Service's Pacific Northwest Research Station, examined eastern Oregon's ponderosa pine forests more than a decade after Gannett's survey. Munger made insightful observations about forest structure and composition, including frequent comments about fire's obvious influence (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."

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: 20 trees per acre results in an equilateral (triangular) spacing of 50 feet between trees, most assuredly an open stand condition. This equilateral spacing calculation is provided for tree density context – presettlement ponderosa pine stands did NOT feature individual trees growing at a regular spacing of 50 feet apart. A common structural condition was large ponderosa pine trees occurring in clumps or groups, as illustrated in photographs used for figures 44-46 later in this white paper. Since trees were aggregated into clumps, spacing between individual clumps was often much greater than 50 feet.]

George Bright offered similar observations about the open character of ponderosa pine forests, along with five specific causes for its open structure (Bright 1914):

> "The most striking feature of a stand of Western Yellow pine is its open character. The peculiarity is the first thing which strikes anyone looking upon such a forest for the first time. Even growing on the best soils and under favorable climatic conditions, it would be difficult, if not quite impossible, to find a full or normal stand of Yellow pine over an area of forty or even ten acres. There appear openings even where the very best conditions for growth of this tree occur, as well as in localities where conditions are less favorable. The peculiarity of Yellow pine stands is due to five primary causes, as follows: (1) fire, (2) insect infestation, (3) windfall, (4) root competition and (5) light competition."

George Bright and Thornton Munger worked together to install plots in Blue Mountains ponderosa pine type in 1910 and 1911 (Bright 1912, Munger 1912). Bright summarized plot results in a published journal paper (Bright 1914). Unfortunately, plot results were only provided for merchantable trees, which included stems 12 inches in diameter and greater.

Even though it lacks information for trees smaller than 12 inches, Bright's summary provides a useful stand density reference condition for mature, large-diameter ponderosa pines. A lack of reference condition information for the regeneration component (e.g., seedlings, saplings, and poles) is unfortunate because it would have provided a more comprehensive characterization of presettlement dry-forest structure.

Presettlement stand densities of mature ponderosa pines on eight dry-forest plots from three localities in the Blue Mountains are provided in table 2.

Table 2: Historical tree density information for 8 plots and 3 localities in the Blue Mountains.

Plot Locality	Plot Size (Acres)	Total Number of Trees on Plot[1]	Trees per Acre[1]	Average Tree Diameter (Inches)[1]	Basal Area per Acre[1]	SDI
Palmer Junction	5	170	34	21	82	126
Palmer Junction	5	190	38	23	110	166
Palmer Junction	4	119	30	19	59	93
Palmer Junction	4	176	44	19	87	137
Palmer Junction	6	159	27	21	63	99
Whitney	20	669	33	21	80	124
Whitney	10	301	30	22	79	122
Austin	4	124	31	22	82	125
Mean	7	239	33	21	80	124

[1] Due to the parent sources, all numerical values pertain only to trees 12 inches dbh and greater. Basal area per acre was calculated by multiplying trees-per-acre values (4th column) by basal area (in square feet) of a tree with the diameter shown in Average Tree Diameter (5th column). SDI (Stand Density Index) (Reineke 1933) is calculated by using Trees per Acre and Average Tree Diameter columns.
Sources/Notes: Adapted from Table V in Bright (1914). Austin and Whitney plots were likely established in same general area as an Austin-Whitney tract described in figure 22 later in this white paper; Palmer Junction plots were likely established in same general area as a Lookingglass Creek tract described in figure 22. Note that figure 22 provides similar tree density information as is presented here, although it includes tree density for trees greater than 1 inch in diameter (rather than the 12-inch limit used here), and it provides summary calculations for trees greater than 21 inches in diameter.

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 in biophysical settings where most of them would have been eliminated by the historical fire regime (Agee 1996b, 1998; Cooper 1960; Munger 1917; Mutch et al. 1993; Sloan 1998b; 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 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 Blue Mountains or 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, Vestal 1917), 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, Thornton Munger, and George Bright could hardly fail to notice its impact on forest composition and structure (this paragraph refers to quoted material from Ketcham, Gannett, Munger, and Bright in section 3 – Historical Context).

If Ketcham, Gannett, Munger, or Bright 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 big yellow pines, some of which were harvested to make moldings, window sashes, and doors, as well as 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 suggests that humans 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, impacts of native peoples were relatively minor, and the landscape was pristine (Cronon 1996, Kay and Simmons 2002). Subsequent work shows 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 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, Scott 1928).

Even though their populations were already declining dramatically due to diseases introduced after European contact (Cook 1955), Native Americans of interior Pacific Northwest may have expanded their use of fire in early 1700s, perhaps to promote forage for horses they just acquired for the first time (Habeck 1987; Haines 1938; Humphrey 1943; Mosgrove 1980; Stewart 1951, 2009).

Recent investigations indicate that American Indians were far from passive hunters and gatherers often depicted in western movies and novels. Their actions had a profound influence on 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 habitats required by 'first foods' plants and animals (Boyd 1999, Denevan 1992, Kay 1994, Robbins 1997, Shinn 1980, Swetnam 1984, 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 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 an unintended consequence of allowing open stands of park-like ponderosa pine to be transformed into dense, thick forests of grand fir and Douglas-fir (Harrod et al. 1999, Mast et al. 1999, Sloan 1998b, Turner and Krannitz 2001) (fig. 6). Fire suppression also transformed the structure of dry forests by shifting much of the canopy leaf area from an overstory layer to one or more understory layers.

Ironically, many of these thick, multi-layered, dry forests may present a more attractive appearance than the park-like pine stands they replaced – there seems to be an intuitive human sense that when it comes to forests, lush is better (Gruell 2001, Hjerpe et al. 2016, Scott 1998a).

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 3). With thick bark and few branches close to the ground, ponderosa pine and western larch easily resist surface fires that eliminated firs and other invading tree species (Agee 1994, Cooper 1960, Dickman 1978, Weaver 1967b, White 1985).

When considering climate only (precipitation and temperature), Douglas-fir or grand fir are most assuredly climax species for dry, mixed-conifer sites of the Blue Mountains (dry mixed-conifer forests include ponderosa pine, grand fir, and Douglas-fir potential vegetation types; see appendix 1).

But when surface fire is superimposed on the climatic regime, it results in 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).

Light-water tradeoff theory (Smith and Huston 1989) maintains that plants cannot be optimally adapted to both light and water. Dry forests are water limited, with dominant conifers evolved to compete for water first and light second.

A surface fire regime creates an open stand of fire-resistant species. As long as fires continue, stands are thinned and competition for water is reduced.

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 4). A 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 late-seral grand firs and Douglas-firs can establish under ponderosa pine when underburning is absent, they may not have enough resilience to make it over the long run, let alone survive the next drought. This means that many late-seral stands of grand fir and Douglas-fir, which replaced an original stand of early-seral ponderosa pine, are destined to become weak – and weak forests are susceptible to insect, disease, and fire outbreaks (fig. 7; Agee 1996b, 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).

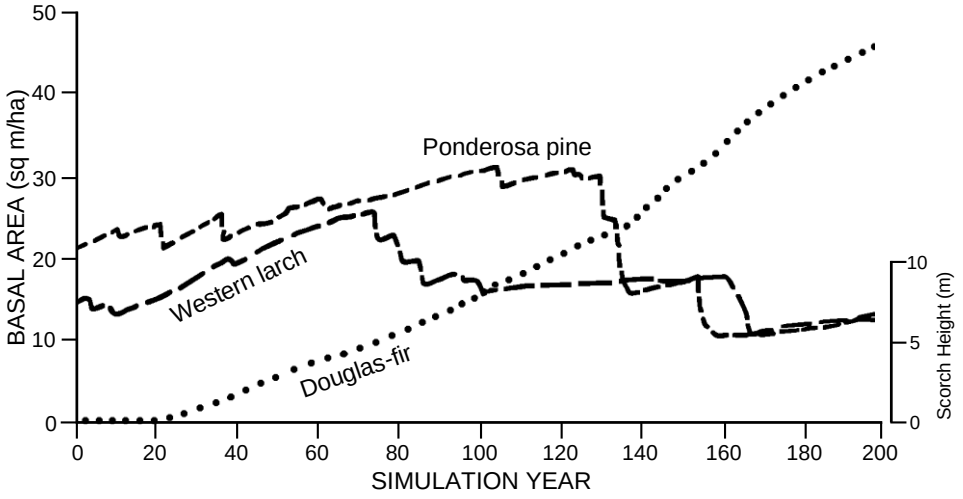


Figure 6 – Forest succession for Douglas-fir/mallow ninebark (PSME/PHMA) plant association in the absence of recurring wildfire (adapted from Keane et al. 1990). In this simulation study, "compositional shifts from ponderosa pine and larch to Douglas-fir occurred in simulations of 50-yr fire intervals 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 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 1996b). Since dry-forest surface fire occurred in Blue Mountains on a cycle of 5-20 years (Hall 1976, 1980), this study helps explain why species composition has changed dramatically for this biophysical environment. This and other studies show that fire exclusion in dry-forest types results in greater canopy cover and density of shade-tolerant trees, higher fuel loads, and increased fuel continuity, which combine to increase potential for high-severity, stand-replacing fires (Agee and Skinner 2005, Parsons and DeBenedetti 1979, North et al. 2005, Zald et al. 2008).

Table 3: 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 a 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 a 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.

Table 4: 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 a predominant species associated with each seral stage by PVG. Tree Longevity age values are from Powell (2000).

Successional roles of ponderosa pine and white (grand) fir were recognized by early silvicultural researchers, as demonstrated by these comments about forest succession and development for 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."

Early-seral communities developing under an influence of recurring disturbance can be ecologically resilient. A disturbance regime for these dry-forest sites was generally dominated by frequent, low-severity fire, resulting in open, multi-aged stands with a vigorous herbaceous undergrowth.

Disturbance frequency determines the length of successional cycles for a particular ecological system. Ecosystems with frequent disturbance 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 presettlement, park-like ponderosa pine forest (see fig. 5, and fig. 34 later in this paper).

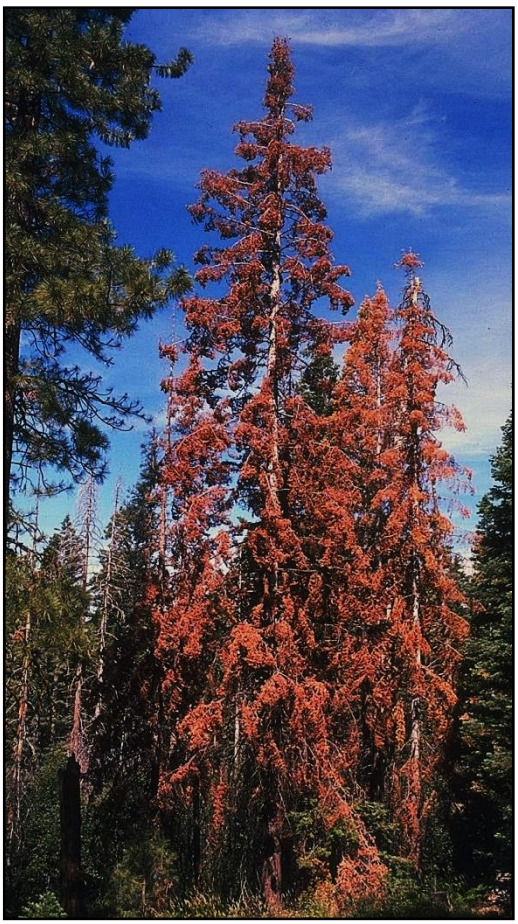


Figure 7 – Grand fir trees killed by fir engraver bark beetles (from Powell 1994). Defoliation, drought, root disease, dwarf mistletoe, overstocking, and other stressors increase a tree's susceptibility to bark beetle attack (Filip 1994, Filip and Schmitt 1990). Fir engraver and Douglas-fir beetles caused widespread damage in the Blue Mountains during 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 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).

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 (figs. 5 and 34). "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).

Plant succession. A 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). A 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 fire tends to cause minimal damage to larger trees; historically, this was a prevailing fire type for ponderosa pine ecosystems throughout the western United States.

4.2 Fire's Influence On Site Nutrition

After frequent fires were suppressed following Euro-American settlement, microbial decomposition has been unable to process rapidly accumulating organic debris (needles, twigs, and

branches) on dry sites. Impaired decomposition and nutrient cycling rates can be initial 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 decomposition and nutrient cycling processes are not functioning properly (Yazvenko and Rapport 1997).

Numerous studies have documented slow decomposition rates for woody biomass of the western United States. This means that interior Pacific Northwest forests 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).

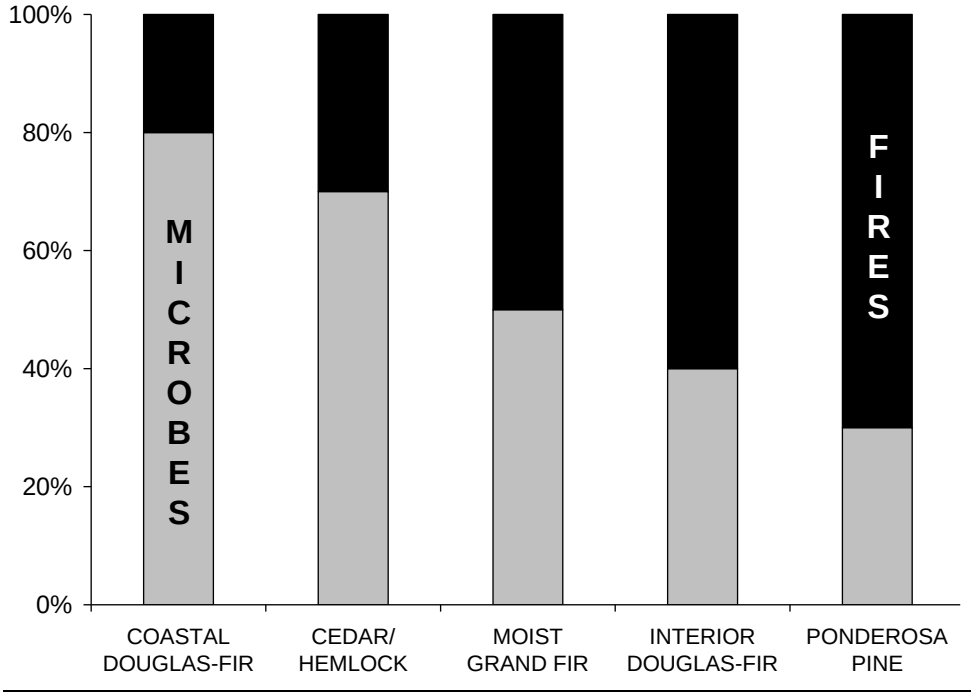


Figure 8 – Microbes and fire as agents of decomposition (adapted from Harvey et al. 1994). Fire (black portion of bars) and microbes (gray portion) are important decomposition and nutrient cycling agents. For the dry-forest climatic zone of the interior Pacific Northwest (the interior Douglas-fir and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

And, these two nutrient-cycling processes – microbes and fire – are obviously related because frequent fire not only converted litter to its mineral elements (calcium, etc.), but it also 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 more slowly than grass litter in a summer-dry Mediterranean climate of the interior Pacific Northwest (Hart et al. 1992), means that nutrient cycling has undoubtedly deteriorated for contemporary forests when compared with historical conditions (Cooper 1960; Covington and Moore 1994a, 1994b; Weaver 1943).

This trend also means that dry mixed-conifer forests are accumulating biomass faster than it is being removed by surface fire, timber harvest, or microbial decomposition, leaving millions of acres vulnerable to drought stress, insects and diseases, and uncharacteristic wildfire (Sampson and Adams 1994). [But this trend may change as wildfire increases by orders of magnitude, potentially removing substantially more biomass – see figure 40 later in this white paper.]

Interactions Between Fire And Nutrients

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 has a negative effect on the growth of *Armillaria ostoyae*, cause of Armillaria root disease. These studies found that much of the Armillaria suppression is related to a fungus called *Trichoderma* – a strongly antagonistic competitor of Armillaria root disease – and *Trichoderma* apparently benefits 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. 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 solely 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).

Forest floor also plays 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. Chemicals produced during allelopathy are often referred to as phytotoxins (Kelsey and Harrington 1979, Rietveld 1975).

If phytotoxins are produced by a climax tree 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 any phytotoxins would obviously affect its own offspring. In situations where a dominant plant species produces chemicals limiting its own abundance, a phytotoxin is referred to as an autotoxin (e.g., 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 often is for dry-forest sites, and when growth-inhibiting conditions occur in an ecosystem where short-interval surface fire was a prevailing disturbance process, then adequate tree survival and growth can only be maintained at relatively low tree densities. Therefore, chemicals from mature trees could function as 'density regulators' by reducing germination and growth of its own progeny (Kelsey and Harrington 1979).

Does Allelopathy Help Regulate Seedling Density?

Trees with capability to use allelopathy (e.g., autotoxic chemicals) to regulate seedling density could possess an important evolutionary adaptation because this trait could effectively limit or prevent overcrowding, stagnation, and competition between individuals of the same species. This life-history trait would ensure that some small proportion of a seedling cohort would grow fast enough to reach a size conferring reasonable resistance to a 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?

It is clear that when plant succession occurs on dry-forest sites in the absence of recurring wildfire, it eventually results in reduced availability of mineral nitrogen and causes increased accumulation of polyphenolic compounds in the mineral soil (MacKenzie et al. 2006, Souto et al. 2000, Wardle et al. 2000). And these changes caused by fire suppression are superimposed on high levels of natural soil variation related to vegetation influences – in a study from the southwest, 69% of soils in openings between patches were Mollisols (a grassland soil) whereas 75% of soils in presettlement tree patches were Alfisols (a forest soil) (Abella et al. 2013).

4.3 Awareness Of Changes Caused By Fire Suppression

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 these changes were recognized early on, but they did not generate a response because of prevailing attitudes of the time.

Two studies described earlier illustrate differing attitudes about fire's role in ponderosa pine forests. Gannett (1902) surveyed 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."

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 crown base height above average flame length in 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 quote helps us remember that fire tolerance is primarily a species-specific life history trait (see table 3), but it also varies with size of individuals in a population.

Munger's (1917) comments about fire-caused damage reflect a commodity paradigm of his era; ponderosa pine forests were to be managed as a sustainable source of wood products, and fire was perceived as an obstacle to reaching that goal. William Greeley, an early Chief of USDA Forest Service, expressed a commodity philosophy in 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."

Munger's commodity orientation was shared by other Forest Service researchers working in western United States, as demonstrated by a 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 USDA Forest Service also recognized that fire-caused changes were occurring on the landscape, as described in these three 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" (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).
>
> "In times gone by the frequent fires killed out the patches of reproduction about as soon as they occurred, but since the fires have been in large measure stopped, reproduction has come in very thickly in most Yellow pine forests, and its abundance points to a heavier future stand than the existing stand. This abundance is decidedly out of proportion to the comparatively small number of old trees in most Yellow pine forests which make up the present stand" (Bright 1914).

When evaluated in a 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 – it was the fact that surrogates were not substituted for fire, fire surrogates providing similar ecosystem functions such as nutrient cycling, fuel reduction, and tree thinning.

"In the absence of fire, vegetation development generally increases ladder and canopy fuels as tree stands become denser (Hessburg et al. 2000), and more surface fuels accumulate as the vegetation shifts from herbaceous plants and shrubs to woody material (Pinol et al. 2005)" (Stephens et al. 2014).

Harold Weaver's Observations About Fire Protection

More than 60 years ago, an early fire ecologist (Harold Weaver) made 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).

Here are Harold Weaver's observations:

"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?"

4.4 Summary: Changes Caused By Fire Exclusion

Contemporary dry-forest landscapes reflect many long-term influences of fire exclusion:

1. Without frequent fire to retard plant succession, fire-sensitive grand fir and Douglas-fir invaded sites where ponderosa pine had been maintained as a fire disclimax (Lunan and Habeck 1973; Parsons and DeBenedetti 1979; Sloan 1998b; Stephens et al. 2016, 2018).
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 (Keen 1950, Miller 1926, 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. High stand density created elk thermal cover that is neither appropriate nor sustainable in an ecological context (Powell 2012).
   d. Dense forests produce less water for streams and springs than open forests (Bosch and Hewlett 1982, Covington and Moore 1994b, Grant et al. 2013, 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 'height to live crown base' by pruning lower branches of overstory trees, reducing potential for crown-fire initiation (Agee 1996c, Keyes 1996).
6. By maintaining open stands and allowing perennial herbs to persist, surface fire provided forage for both livestock and wildlife (Hedrick et al. 1968, Irwin et al. 1994).
7. Fire supported nutrient cycling 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.

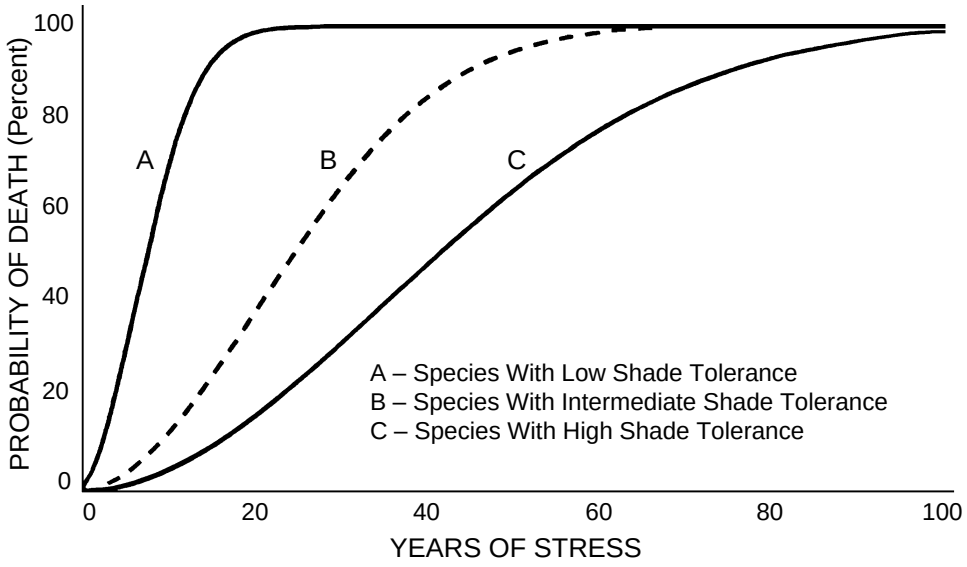


Figure 10 – Tree resistance to stress varies with shade tolerance (adapted from Keane et al. 1996). Intolerant tree species (lodgepole pine, ponderosa pine, western larch) die relatively quickly when exposed to chronic stress such as high stand density. Trees with intermediate tolerance (Douglas-fir 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 dry-forest undergrowth plant communities (Burkhardt and Tisdale 1976, Gedney et al. 1999).
11. Western juniper increased with fire exclusion (fig. 11) (Gedney et al. 1999), reducing water yields because juniper uses more water than grasses and shrubs (Miller et al. 1987).
12. Loss of an open park-like structure had negative impacts on blue grouse (Pelren and Crawford 1999) and white-headed woodpecker (Buchanan et al. 2003, Casey et al. Undated).
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-sensitive conifers displaced fruit-bearing shrubs, deciduous trees, and herbaceous plants – important food sources for wildlife (Bartos and Campbell 1998, Gruell 2001).
15. Native Americans burned the landscape to promote forage for horses, and to maintain important habitat for 'first foods' plant species (Habeck 1987; Haines 1938; Humphrey 1943; Mosgrove 1980; Stewart 1951, 2009).

    [Note: Since responsibility for provision of plant-based 'first foods' tended to reside with women, they often possessed much of a tribe's prescribed-fire expertise, as noted in this account: "On the way, they met an old squaw, with a large firebrand in her hand, with which she had just set the grass and bushes on fire; when surprised, she stood motionless, and appeared to be heedless of anything that was passing around her" (Wilkes 1844).]
16. Fire exclusion created landscapes that are more homogeneous, with fewer vegetation types and lower patch densities (Lehmkuhl et al. 1994, Miller and Urban 2000).
17. Landscape diversity declined after fire was prevented from periodically creating early-seral plant communities (Hessburg et al. 1999b, Taylor and Skinner 1998).

Figure 11 – Western juniper expansion on a dry-forest site, likely as a result of fire exclusion (Kahler planning area, Heppner Ranger District). This image portrays a dry forest example of the ponderosa pine/bitterbrush/Ross' sedge (PIPO/PUTR/CARO) plant association (Johnson and Clausnitzer 1992). Juniper is occasionally associated with late-seral communities in this plant association, but it typically occurs at low canopy coverage (2% mean cover for seven PIPO/PUTR/CARO stands sampled by Johnson and Clausnitzer 1992), and it is not found in every stand (juniper occurred in 42% of the samples). The amount of juniper shown here is greater than what was encountered by Johnson and Clausnitzer (1992, appendix C) in their late-seral sample stands. Juniper has increased in areal extent from historical levels – the interior Columbia Basin ecosystem management project reported increases of 243% for the juniper/sagebrush cover type in the Blue Mountains ecological reporting unit (Quigley and Arbelbide 1997, p. 676). Although much of this reported increase involves juniper expansion into rangelands, juniper also increased on dry-forest sites. Manifold increases in western juniper abundance have been reported in many studies examining eastern Oregon vegetation conditions (Azuma et al. 2005, Gedney et al. 1999, Knapp and Soulé 1998, Miller et al. 2005).

4.5 Active Management Implications Of Fire Exclusion

Results from many scientific assessments completed over the past four decades concluded that impacts associated with wildfire, insects, and diseases 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 ecosystem process, and its exclusion by human society has many consequences – some of which were intended, but many of which were not. In this context, adopting an active management (restoration) approach (see section 7, "Restoration of Dry-Forest Ecosystems") is a reasonable response to an historical paradigm of fire exclusion.

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 trees are killed by fire (see fig. 19 later in this white paper).

A list in section 4.4 enumerates 17 ecosystem changes relating to fire exclusion on dry-forest sites. Although extensive, it still may not furnish 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 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, especially for dry forests (Belsky and Blumenthal 1997, Fleischner 1994, Hatton 1920, Madany and West 1983, Oliver et al. 1994c, 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, USDA Soil Conservation Service 1941).

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 (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 summer and fall of 1861, large numbers of sheep and cattle were driven into eastern Oregon and Washington from the Willamette 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. 12 and 13), causing persistent changes in vegetation composition (Bright 1914, 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 during autumn (Darlington 1915, Oliver et al. 1994c).

Sheep grazing caused conflict between cattle ranchers, homesteaders, and sheepherders because 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:

> "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" (Langille 1906).

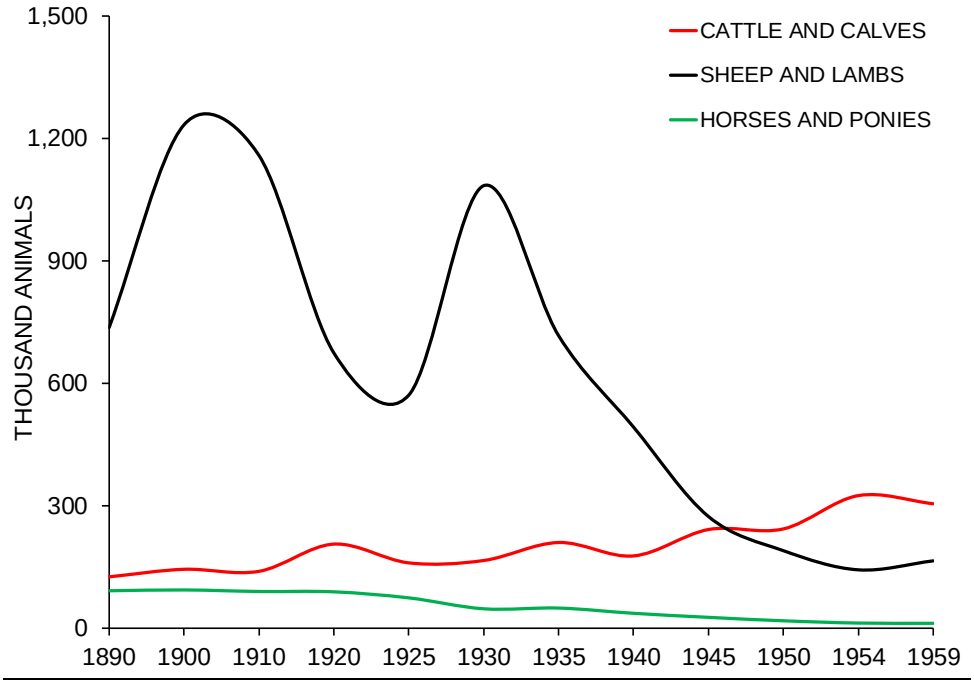


Figure 12 – Number of domestic grazing animals, summarized for three livestock categories, for nine counties in northeastern Oregon and southeastern Washington (Asotin, Columbia and Garfield in Washington; Grant, Morrow, Umatilla, Union, Wallowa and Wheeler in Oregon). Data derived from Bureau of Census agricultural summaries (Bureau of Census 1895, 1902, 1913, 1922, 1927, 1932, 1942, 1946, 1952, 1956, 1961).

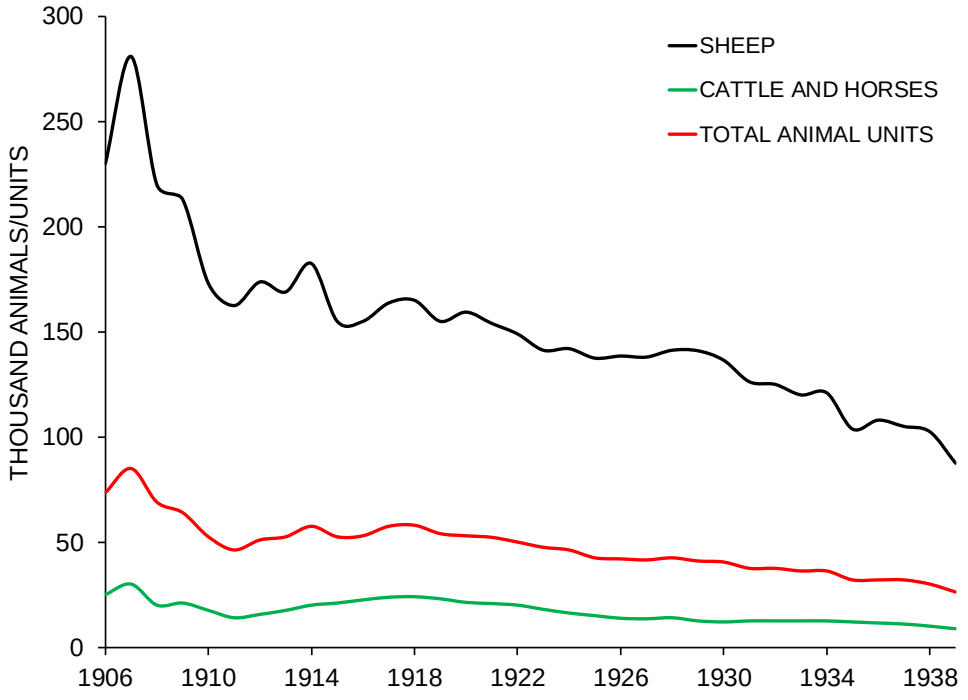


Figure 13 – Grazing summary for Umatilla National Forest, 1906-1939. Data derived from USDA Soil Conservation Service (1941) (note: little information is provided by this source about the basis for calculation of 'Total Animal Units').

An early survey of sheep ranges found moist mountain meadows entirely devoid of vegetation and experiencing severe soil erosion (fig. 14). 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 (*Ceanothus velutinus*); even the small ponderosa pines were fed upon (Griffiths 1903, Langille 1903).

When the Blue Mountains were surveyed early in the 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 an examination of 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 12 summarizes historical grazing trends for three classes of livestock and nine counties in northeastern Oregon and southeastern Washington. Figure 13 provides early grazing trend information for Umatilla National Forest from 1906 to 1939 (USDA Soil Conservation Service 1941).

Domestic livestock grazing in early 1900s was not the only factor that may have affected forest regeneration; in some areas, impact from native ungulates (deer, elk) was more pervasive and, unlike domestic animals, continues at moderate or high levels today (Case and Kauffman 1997, Endress et al. 2012, 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 they were reintroduced to the Blue Mountains, increasing from 360 animals in 1921 to 13,000 animals by 1941 (fig. 15).

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 establishment of tree seedlings (Cooper 1960, Kolb and Robberecht 1996, Pearson 1942, Rummell 1951).

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. 16; Covington and Moore 1994a, Sloan and Ryker 1986, Yazvenko and Rapport 1997).

Figure 14a – Historical photograph showing a band of sheep feeding in dry forest. Sheepherders made an annual migration, following snow as it retreated from lowlands in spring to high country in summer, and then back down to valleys in autumn. It was often noted that peak sheep numbers, and associated damage, occurred from 1890 to 1910 (Tisdale 1961).

Figure 14b – Historical photograph showing a herd of cattle grazing in transitional forest between dry and moist ecological settings (Plenty Bear Ridge). Although cattle grazing occurred later, overall, than sheep grazing, it still caused impacts on dry-forest conditions. As described in this "Influence of Ungulate Herbivory" section, grazing by domestic ungulates (cattle and calves; sheep and lambs; and horses and ponies) changed herbaceous undergrowth plant composition (cattle) and modified woody browse species and production (sheep), and these changes influenced the potential for dry forests to support low-severity surface fire.

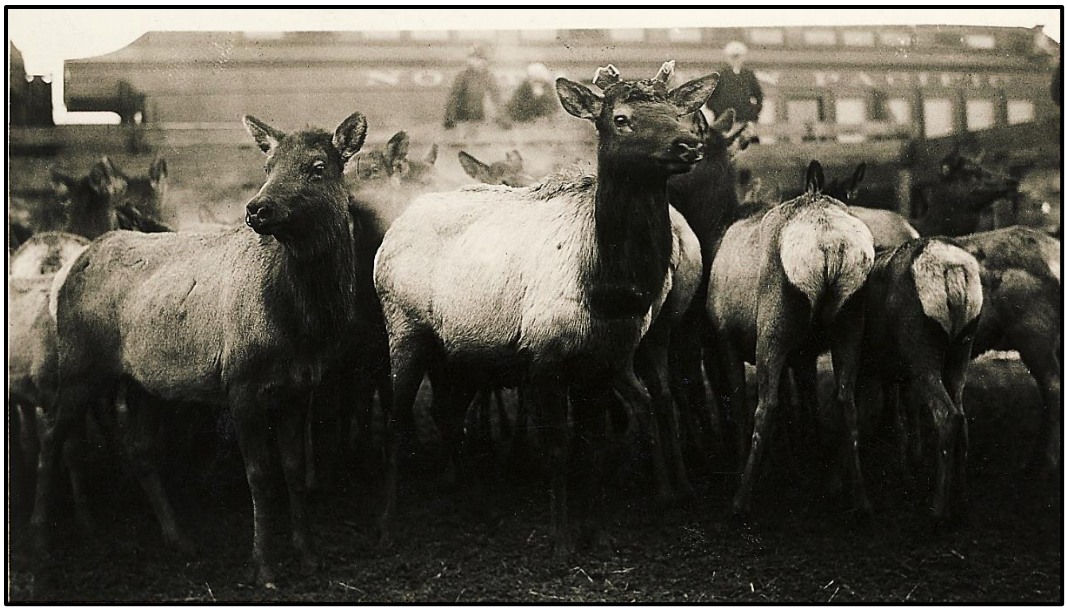


Figure 15a – Elk being dropped off in Dayton, Washington, northern Blue Mountains, on February 1, 1930. These 30 head of elk were shipped from Montana to Dayton, Washington by railcar on a Northern Pacific train. Cost of shipment was approximately $700. Note: Importing elk into the northern Blues began in 1909, when a Game Commission was formed and 4 railcars of elk were shipped west from Yellowstone National Park – 1 carload was delivered to Dayton, 2 carloads to Pomeroy, and 1 carload to Clarkston for the Lewiston Flats area. Each carload contained 36 cows and 4 bulls, and 1909 shipping costs for elk ran $4.95 per head.

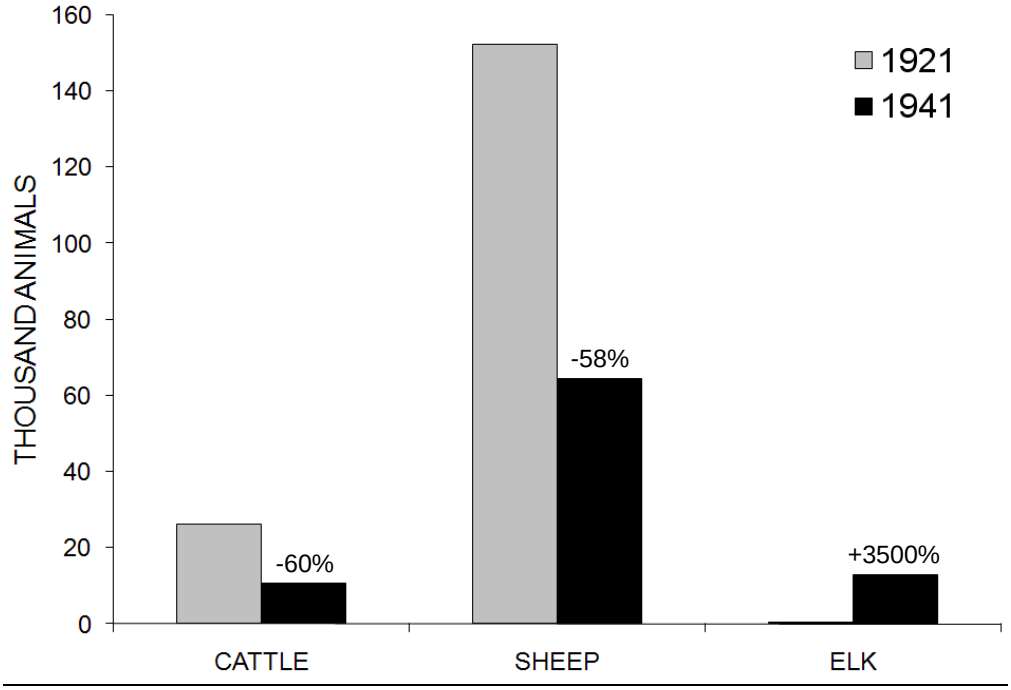


Figure 15b – Ungulate trends for Whitman National Forest in Blue Mountains of northeastern Oregon (data from Pickford and Reid 1943). This chart shows cattle and sheep numbers declining dramatically between 1921 and 1941, and elk numbers increasing from only 360 animals in 1921 to more than 13,000 by 1941.

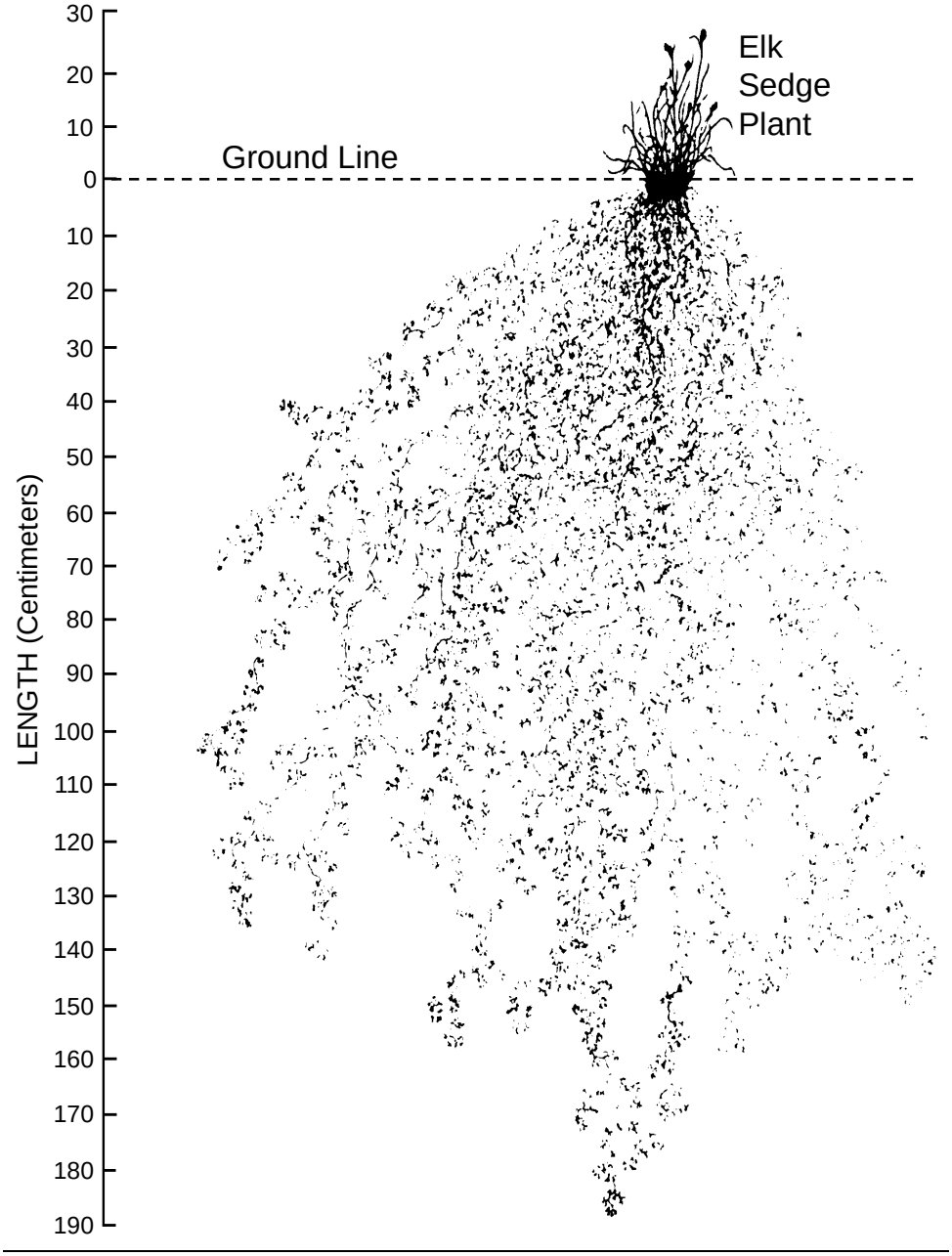


Figure 16 – Elk sedge has a fibrous root system occupying an impressive volume of soil (adapted from Sloan and Ryker 1986). The plant in this diagram is 12 inches tall and 10 inches wide, but its roots spread 56 inches wide and 75 inches deep; dashed line shows ground level. Competition from extensive 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, inhibitory effects of rhizomatous herbs is viewed as a management problem because herbs function as 'competing vegetation' by limiting 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 upland forest) where large numbers of seedlings (and eventually mature trees) would easily exceed an area's capacity to support sustainable levels of tree stocking (Cochran et al. 1994, Powell 1999).

As often happens, grazing effects were likely influenced by interactions with other factors. Heavy grazing early-on by domestic livestock (particularly sheep; see figs. 12-15) 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 might otherwise 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).

A restoration section (section 7) describes how it may 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 (fig. 17). And since prescribed fire occupies a primary position in a hierarchy of active management treatments considered for dry-forest restoration, grazing management should play a critical role in any effort to craft suitable habitat for restoring fire exhibiting 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. 16). High tree density is a common problem throughout eastern Oregon and eastern Washington (Powell et al. 2001), so the importance of this inhibitory effect on tree establishment should not be underestimated.

Speculation about an interaction between livestock grazing and tree regeneration is frequently mentioned in the scientific literature (Cooper 1960, Madany and West 1983, Steele et al. 1986, Zimmerman and Neuenschwander 1984, and others). However, it is often difficult to establish a cause-and-effect relationship between grazing and tree regeneration, perhaps because of difficulty in establishing a carefully controlled research framework accounting for potential influences of confounding factors.

Rummell (1951) studied tree regeneration patterns for grazed and ungrazed areas in central Washington. He implied there was a direct relationship between the degree of forage utilization by livestock and the density of ponderosa pine reproduction. But there were important differences in the representation of pinegrass and elk sedge between grazed and ungrazed areas, and the differences were not necessarily explained by palatability or other grazing factors.

Rummell's (1951) study suggests that grazing may not have been a primary factor affecting tree regeneration because elk sedge and pinegrass – two plant species known to limit tree seedling establishment on dry-forest sites (Sloan and Ryker 1986) (fig. 16) – were apparently reacting to some influence other than grazing (perhaps reflecting variations or differences in site potential (e.g., plant association) from one portion of his study area to another?).

Figure 17 – Fenceline contrast related to livestock grazing. Domestic and wild ungulates (see figs. 12-15) sometimes caused dramatic impact on the abundance and vigor of herbaceous plants. Grazing animals caused many secondary influences for dry-forest sites: (1) they disrupted the prevailing surface fire regime by removing much of the fine-fuel component (herbs) functioning as a fire carrier; (2) they reduced herbaceous competition for tree seedlings, thereby allowing many more trees to become established than would have occurred otherwise; and (3) they reduced the abundance of aspen, cottonwood, serviceberry, and other broadleaf trees and shrubs (Endress et al. 2012).

When Miller and Halpern (1998) studied effects of grazing and environment (climate) on tree establishment for Cascade Range in Oregon, they noted that "the strongest support for the absence of grazing-induced changes comes from establishment trends on south-facing slopes. Here, despite widely varying dates of closure to sheep, tree invasion remained relatively synchronous among transects and was closely timed to the onset of wetter weather" (Miller and Halpern 1998, p. 280). In other words, climate apparently had more influence on tree regeneration patterns for this Cascade Mountains study area than livestock grazing or the magnitude of its impact.

Jon Skovlin and others (1976) studied cattle grazing methods on ponderosa pine ranges and, although it was not the primary objective of their investigation, they noted an impressive, 12-fold increase in tree seedling density during the 13-year period of their study. Once again, however, this increase was apparently unrelated to cattle use since the same response occurred in units grazed by wildlife only, and because there was no statistically significant seedling density difference between different cattle-grazing intensities (Skovlin et al. 1976).

Livestock grazing on national forest lands was sanctioned after creation of USDA Forest Service by Transfer Act of 1905. In my opinion, high grazing levels of the early 1900s, particularly by sheep (figs. 12-15), were sufficient to affect tree regeneration by reducing herbaceous vegetation (by reducing herbaceous competition with seedlings), and by exposing mineral soil for tree-seed germination; I would not expect substantially reduced livestock grazing levels of today to have a significant influence on tree regeneration patterns for these lands.

6. INFLUENCE OF SELECTIVE CUTTING

Fire exclusion allowed a multi-layered structure to develop on a majority of dry-forest sites (influence of fire exclusion is discussed in section 4 of this white paper). 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, limited amounts of lodgepole pine. When wood products were harvested from these stands beginning in 1940s and 1950s (see appendix 2), many overstory trees were removed for these reasons (Powell 1994):

- 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 may have low vigor and little resistance to insect attack, they were often 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 a frequent surface fire regime had been allowed to continue its historical role (see fig. 46 later in this white paper).
- 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 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 an immature understory of Douglas-fir and grand fir (Dezellem 1983).
- An overstory removal strategy seemed to make good sense – it avoided the cost of tree planting, an expensive practice; it avoided an undesirable appearance associated with clearcutting; it maintained the pleasing aesthetics of a green, forested setting; and it capitalized on previous growth of understory trees existing for 60 years or more.
- 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 68). Heavy commercial timber harvests in a northwestern pine region (eastern Oregon and eastern Washington) began in the 1880s (fig. 18) (Weidman and Silcox 1936), although some previous harvesting occurred in conjunction with gold extraction and mineral development (Lindgren 1901).

Mining activity in lower Columbia River basin can trace its origins to discovery of gold on Canal Gulch of Orofino Creek, a tributary of 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).

Figure 18 – Early timber harvest in Blue Mountains (also see appendix 2). Relatively heavy commercial harvest began in limited portions of lower Columbia River basin in 1880s (Weidman and Silcox 1936), with early harvest concentrated near settlements, mining camps, and railroads. Large ponderosa pines and Douglas-firs were removed early on because of their desirable wood qualities and their abundance in easily-accessible, park-like stands at lower elevations (Oliver et al. 1994c). Settlers and homesteaders, however, often had a different species preference because their favorite tree tended to be 'tam-brack' (western larch) because it was durable (decay resistant), young trees furnished long, straight poles, and large trees split easily into the finest rails ever enclosing a pig pen or garden patch.

Note: early range was legally open, as much of it still is today, so settlers and farmers had to fence free-range livestock **out**, rather than livestock producers being responsible for fencing their animals in and thereby preventing damage to settlers' gardens or planted crops (Robbins 1997, Robbins and Wolf 1994).

For the Blue Mountains, gold was discovered in Griffiths Gulch, located a few miles southwest of Baker City, Oregon, in autumn 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 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 significant timber harvest near Sumpter by 1900 (Gannett 1902, Thompson and Johnson 1900).

Since an extensive road network was not present in the Blue Mountains during a mining era, widespread timber harvests did not occur. A far ranging road system eventually 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 unsuitable for house logs or fence rails, settlers needed access to mountain timber. The favorite mountain timber of early pioneers was tamarack or western larch (they called it 'tam-brack') because it was durable (decay resistant), young trees furnished long, straight poles, and large trees split easily into the finest rails ever enclosing 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 roaming free on adjoining foothills of bunchgrass. Once settlers began growing grain (Humphrey 1943) and needed more timber to fence their fields and 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, resinous, durable woods of ponderosa pine and western larch were found to be ideal for producing railroad ties (Robbins and Wolf 1994, Tucker 1940).

Beginning in 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 alters forest composition (Abella et al. 2006) by removing 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 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 valuable ponderosa pine and western larch? Under market (economic) conditions of that era, selective cutting was viewed as a wise use of natural resources because it captured 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 with the hope that some 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 the same concept 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" (Haig et al. 1941).

In many respects, selective cutting had the opposite effect of native disturbance processes operating on dry mixed-conifer sites. Surface fire was historically a dominant disturbance process (Agee 1993, Cooper 1960, Munger 1917, Sloan 1998a, White 1985, Wright and Agee 2004), and it discriminated against fire-intolerant invaders (grand fir and Douglas-fir) while favoring fire-tolerant trees with high, open crowns (ponderosa pine and western larch).

In contrast to surface fire, selective cutting on dry-forest sites removed fire-resistant ponderosa pines and western larches, while allowing grand firs and other fire-susceptible species to remain and flourish (Filip 1994, Filip and Schmitt 1990). [Appendix 2 provides pictures and descriptions for early 1940s selective cutting in the central and southern Blue Mountains.]

Dry forests of the interior Pacific Northwest have a history of high-grading (early selective cutting was often implemented as high-grading). High-grading did not seek to regulate stand structure; instead, harvesting simply removed timber. High-grading can be dysgenic by leaving 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).

Late-seral species 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).

Evaluating Disturbance Evidence To Interpret Successional Trends

Presence or absence of selective cutting evidence can be used when evaluating the successional history of dry-forest ecosystems. When first entering a dry-forest stand, look up into the highest canopy to see if widely-spaced, old-growth tree crowns are found there. If they are, their presence suggests a relatively stable stand structure long free of severe disturbance. Then, look around you at eye level – this generally reveals abundant young trees beneath the old overstory. This finding suggests an unstable structure because a new tree cohort often follows disturbance. But, does a careful search at ground level fail to reveal typical disturbance indicators, such as fire scars at the base of live trees or tree stumps from selective cutting?

At this point in your investigation, you might come to the following conclusions: a relatively open pine stand may have occupied the area more than a century ago (the upper cohort of old trees). Tree seedlings competed unsuccessfully with wildfire (section 4) and herbs (section 5), so tree density was kept relatively low. And, this rationale could help explain why the old trees have large diameter – low understory tree density contributes to rapid overstory growth; we should remember that large tree diameter is not solely a result of advanced age.

You then surmise that cattle grazing associated with Euro-American settlement may have weakened the ground cover. Tree seedlings, no longer held in check by severe herbaceous competition, established abundantly whenever a good seed crop and favorable germination conditions happened to coincide. As grazing continued, however, cattle destroyed or damaged many seedlings, and few of them reached sapling size. But once unfettered grazing was regulated, herbivore pressure declined (figs. 12-15), and more seedlings could then develop normally, eventually resulting in abundant small trees beneath an overstory canopy (see fig 46).

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 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 a tree species with high resistance to disturbance processes, but 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 removed larger-diameter trees, so they functioned as an overstory removal by releasing small seedlings and saplings in an understory. This means that selective harvests generally caused a pronounced change in vertical forest structure. [But, light selective harvest could be viewed as emulating tree mortality caused by western pine beetle (see fig. 36 later in this white paper).]

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 first be necessary to remove some ecologically inappropriate composition (grand fir and Douglas-fir) in order to free up growing space for occupancy by ponderosa pine (including planting ponderosa pine, if need be, to help restore its historical abundance).

7. RESTORATION OF DRY-FOREST ECOSYSTEMS

As a result of substantial reductions in park-like ponderosa pine forests throughout interior Pacific Northwest, they are now considered to be a threatened ecosystem of the United States. Reed Noss and others described 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 USDA Forest Service 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 normally 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 of dry-forest sites by suppressing frequent surface fire, many younger trees would have perished while still small (Barrett 1988, Powell 1994, Sloan 1998b, Steele et al. 1986). And since smaller trees function as 'ladder fuel,' easily lifting surface fire up into a forest canopy, crown fires are more common now than historically, leading to our contemporary perspective that *crown fire, not timber harvest, is currently the greatest threat to old forest on dry sites* (fig. 19). Climate change is a 'double-whammy' for these forests because crown fire and drought act synergistically (Savage et al. 2013).

7.1 Characterization Of Reference Conditions

Restoration efforts benefit from characterization of reference conditions, which disclose how vegetation has changed over time as a result of human influences and disturbance; they help us understand what an ecosystem is capable of, how disturbance functions, and how ecosystems recover after disturbance (Falk 1990, REO 1995). They also provide clues about how we got where we are now, and they help 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 (Egan and Howell 2001, Evans 1991, Powell 1999a). 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."

Figure 19 – Crown fire in Blue Mountains of northeastern Oregon (top photo from Powell 2010; bottom photo shows aftermath of crown fire at 1996 Wheeler Point fire site on Heppner Ranger District). In dense forests with large amounts of canopy fuel loading, fires are very intense and travel rapidly from one tree crown to another. Crown fires are an important process for perpetuating lodgepole pine, grand fir, and subalpine fir forests, although any particular area seldom experiences 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 past century (Arno and Allison-Bunnell 2002).

Seven decades ago, 74% of commercial forest in eastern Oregon and eastern Washington was classified as ponderosa pine, much of it old-growth (Cowlin et al. 1942). By the late 1970s, at least 25% of Pacific Northwest ponderosa pine type had been replaced by mixed-conifer forest (Barrett 1979); reductions were apparently greater for northeastern Oregon where ponderosa pine declined by more than 50% between 1936 and 1980 (fig. 20; Powell 1994).

These forest inventory trends demonstrate that dry mixed-conifer forest, frequently overstocked with Douglas-firs and true firs capable of persisting in overcrowded stand conditions for relatively long periods (see fig. 10), have replaced ponderosa pine and now cover many eastside landscapes (Mason and Wickman 1994) (note: in a potential vegetation context used for this white paper, 'dry mixed-conifer' and 'dry forest' are synonymous terms).

The following comments suggest that a trend of ponderosa pine being replaced by other species was recognized more than 50 years ago (Gedney 1963).

> "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

Altered disturbance regimes often result in forest health problems such as insect outbreaks or stand-initiating fires (figs. 7 and 19), but 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 is available. It found that 135 of these plants (92%) either benefit from fire or are found in fire-adapted ecosystems, suggesting that declines in their abundance or persistence are likely influenced by fire exclusion (Hessl and Spackman 1995) (and, such declines are often viewed as indicators of an 'unhealthy' ecosystem).

Plant succession, in combination with human influence including 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 1998b, Steele 1994). Forest ecosystems 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 effects of forest succession, tree physiology, and insect and disease susceptibility (Clark et al. 1998).

Forest health. 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. 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).

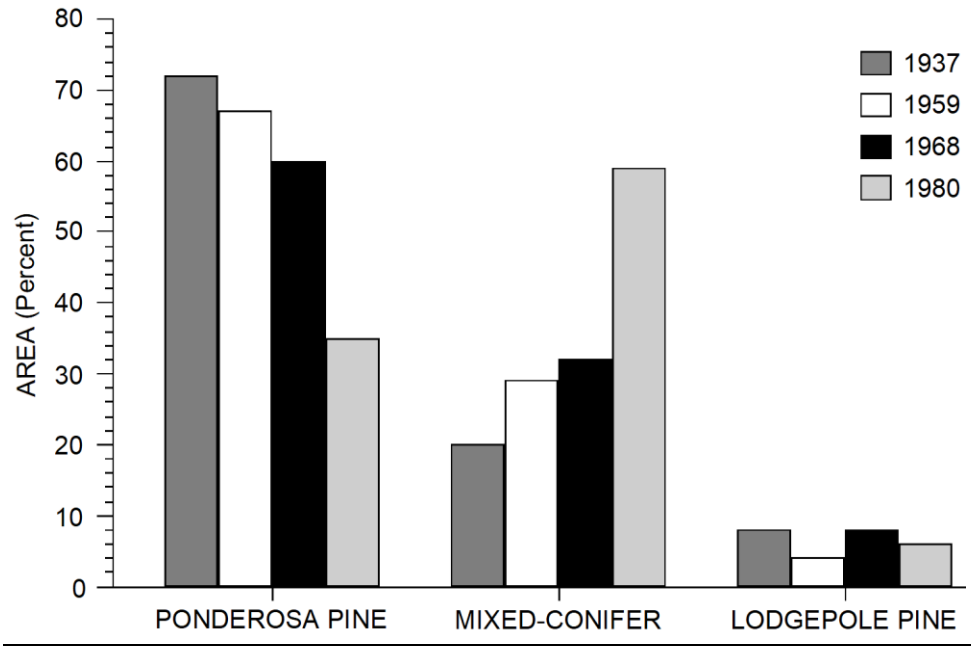


Figure 20 – Change in forest cover types for Malheur National Forest, 1937-1980 (from Powell 1994). Ponderosa pine forest declined by more than half between 1937 and 1980, mixed-conifer type increased by an equivalent amount during this period, and lodgepole pine type remained relatively constant. This figure shows that mixed-conifer forest – prime habitat for defoliating insects – increased by 195% between 1937 and 1980. An increase in mixed-conifer forest was an important reason for unprecedented magnitude of a Blue Mountains budworm outbreak between 1980 and 1992.

Once a forest's vigor falls to low levels, insects and diseases quickly become catalysts of change (Gast et al. 1991, Wickman 1992, and many other citations in References section). Without application of restoration treatments soon (during next 15-30 years), it is very likely that the Blue Mountains' legacy into the second half of 21st century will be large, homogenous landscapes recovering from uncharacteristic wildfires and other ecosystem setbacks on a scale unprecedented in recent evolutionary history (Mutch et al. 1993, 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 an ecosystem farther away from, rather than closer to, a desired future condition (Landres et al. 1999). In situations where current conditions deviate significantly from reference 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, Oliver et al. 1994b, Pickett and Parker 1994).

One example of this concept is that standing and surface fuels often accumulate 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). Caution about reintroducing fire is appropriate because fire exclusion, by itself, did not create our current problem, and fire's reinstatement will not cure it. Fire is an ecological

catalyst taking its character from whatever surrounds it. Ecosystems with uncharacteristic conditions will yield uncharacteristic fires (see fig. 19).

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 (see fig. 17), it may be difficult to reestablish a 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 homogenized eastside landscapes, especially for a montane, mixed-conifer zone at mid elevations (Hessburg et al. 1994, 1999; Lehmkuhl et al. 1994). In drier forests of eastern Oregon and eastern Washington, alteration of a disturbance regime by suppressing fire has de-fragmented inherent patterns of fuel distribution and accumulation, thereby increasing potential for large wildfires (Hessburg et al. 2005, Rochelle et al. 1999) (see fig. 19).

Unnaturally large, contiguous areas of densely stocked and highly stressed trees provide an increased food base for defoliating insects (Gast et al. 1991, Hessburg et al. 1994, Mason and Wickman 1988, Williams et al. 1980), and these forest conditions are also more favorable for occurrence of parasitic plants (Gast et al. 1991, Zimmerman and Laven 1984) 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 during episodic outbreak events (Hessburg et al. 1994, Powell 1994, Wickman 1994).

Reducing stand density to minimize moisture and nutrient stress for individual trees, and then reintroducing fire – a 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 on a landscape scale to reestablish a mosaic of forest types and structural stages that will, in turn, reduce 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).

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 of uncharacteristic fire effects.

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

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 desirable characteristics of presettlement fire regimes (Kimmins 1997).

Historically, spatial variation in fire intensity was important for providing diversity in landscape patterns (fig. 21). [Munger (1917) provides excellent observations about spatial pattern associated with pine forests; see pages 15-16.] Under a 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 (Churchill et al. 2017, del Moral 1972, Hessburg et al. 1999b, 2005; Lehmkuhl et al. 1994).

A 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).

7.3 Emulating Disturbance Processes

A primary focus of dry-forest restoration is to use silvicultural treatments to emulate intensity, scale, and pattern of historical disturbance regimes. An objective of active restoration is to address fire hazard and insect and disease problems – production of timber, water, and other commodities (if any) is only a by-product of meeting overall restoration objectives (DeGraaf and Healy 1993). Salvaging some dead trees produced by an uncharacteristic crown fire, for example, would be appropriate by "leaving an amount of CWD [coarse woody debris] sustainable under inherent disturbance regimes, not an excess that could set the stage for severe fires and subsequent loss of biological capacity" (Everett et al. 1996, p. 276).

Choice of silvicultural treatment can be important in both ecological and economic contexts. For example, a general trend over past decades has seen a transition from forest harvests producing relatively large, high-quality timber to entries 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 economic viability of using commodity revenues to offset costs of dry-forest restoration treatments (Rainville et al. 2008).

Current ecological conditions in dry forests of 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).

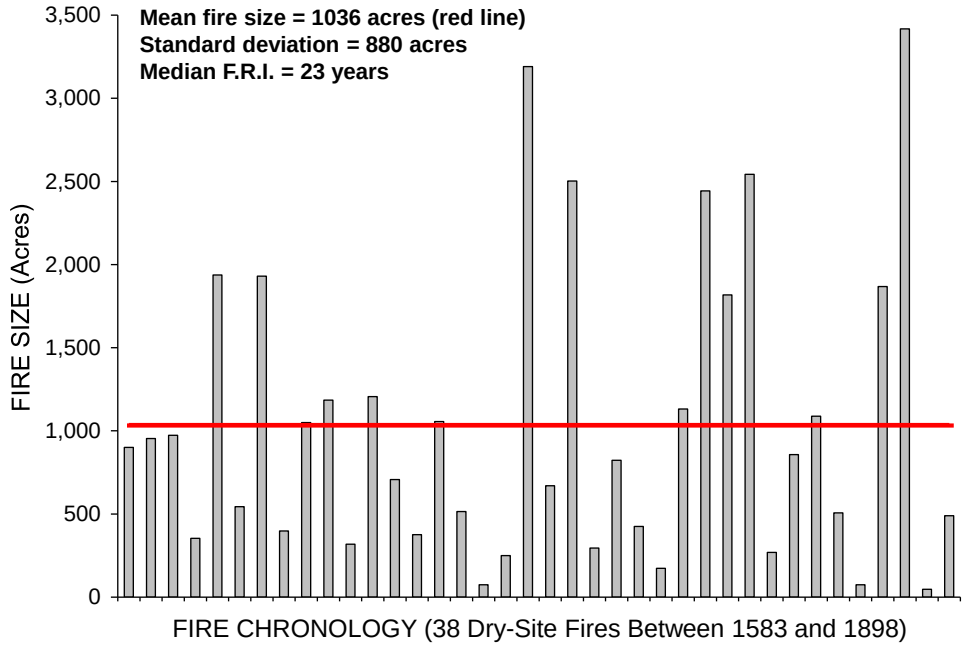


Figure 21 – Spatial variability in fire extent for dry-forest sites in Tucannon watershed, northern Blue Mountains, southeastern Washington (based on data from Heyerdahl 1997). Forty-two individual fire events were interpreted for the watershed, and 38 of them occurred on dry-forest sites. Smallest fire extent on dry-forest sites was 47 acres and largest was 3,417 acres. Average fire extent for 38 dry-site fires was 1,036 acres (red line shows an average). Note that the last recorded fire for this watershed occurred in 1898 (Heyerdahl 1997), although School Fire affected the study watershed in 2005. And, the other three study areas in Heyerdahl's (1997) study (Baker City watershed, Dugout Creek, and Imnaha Creek) also included dry-forest sites, with Dugout Creek area (Malheur NF) consisting entirely of dry-forest biophysical environments.

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 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 management would make an unfortunate situation even worse (Beschta et al. 2004). Many proponents of passive restoration contend that knowledge of reference conditions will never be complete, so we should rely on wildfire, insect outbreaks, and other disturbance processes to fix the problem (transform 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). For 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 mineral soil (MacKenzie et al. 2006, Souto et al. 2000, Wardle et al. 1998). And, waiting for more information fails to acknowledge that it has been estimated that up to 32% of all forests in the U.S. suffer high risk of wildfire (GAO 2003) (and the percentage for interior Northwest is much greater than 32% – see figure 43 later in this white paper).

7.4 Desired Conditions For Dry-Forest Sites

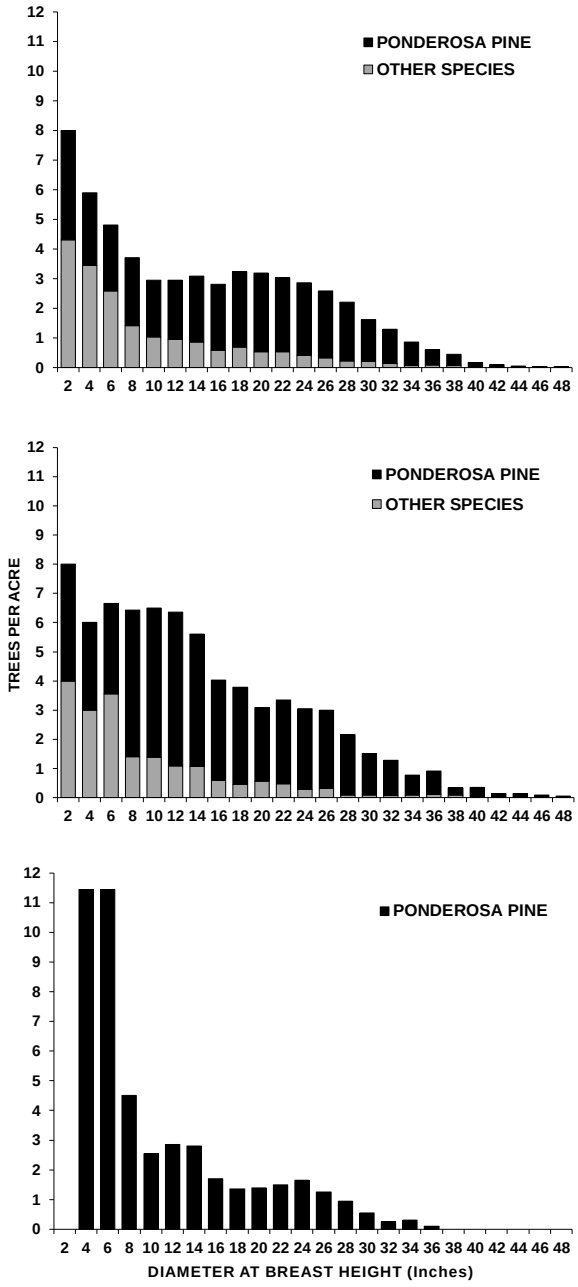
Desired conditions contributing to a sustainable composition, structure, and density for dry-forest sites include the following attributes (Fiedler 2000b).

- An open stand density (40 to 70 square feet per acre of basal area). Stands with a predominance of big trees (> 21" dbh) could be at the upper end of this stocking range and still be viewed as having a sustainable density level.
- A multi-cohort or uneven-aged structure at a 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 even-aged groups in an uneven-aged stand structure would have a single-layer structure (figs. 44-47 later in this paper illustrate groupy or clumpy structures). Typical group size should range from 0.1 to 0.6 acres (Harrod et al. 1999, Youngblood et al. 2004).
- A predominance of large trees – up to 60 percent of basal area per acre would occur in trees whose diameter at breast height was 21 inches or greater (see fig. 22).
- A composition dominated by ponderosa pine – up to 70% would consist of ponderosa pine (see fig. 22). At least ⅔ of 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 a 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 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 within an historical range of variation (RV), management activities should emphasize producing fewer but larger trees (Allen et al. 2002, Wright and Agee 2004). [Section 7.9 provides detailed RV information for composition, structure, and density.]

Numerical goals relating to a desired future condition depend on how a metric is quantified. Inventory data collected in 1910-1911 for three forest tracts in the Blue Mountains (fig. 22), for example, show that when tree density is expressed as basal area, 66% of it occurred in trees whose diameter is 21 inches or more. When forest density is expressed as trees-per-acre rather than basal area, stems with a diameter of 21 inches or more comprise only 23% of total stocking (Bright 1912; Munger 1912, 1917).

A characterization of desired conditions should account for a range of disturbance processes and biological legacies, rather than attempting to directly replicate any particular disturbance agent (Foster et al. 1998, Hansen et al. 1991, Urban et al. 1987). Moreover, land managers should focus 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).



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 22 – Selected stand attributes for three forest tracts in Blue Mountains (adapted from Powell 1999b). This data came from relatively large sample areas measured in 1910 or 1911 (sample areas were 258½ acres for Austin-Whitney, 44 acres for Lookingglass Creek, and 20 acres for Winlock's Mill). Data sources are Bright (1912) and Munger (1912, 1917). [Also see table 2 for similar data.]

Alan White (1985) suggests that ponderosa pine regeneration requires a 'safe site' such as the ash bed of a fire-consumed log, where at least a few 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 successfully making it through this initial fire filter was high. This regime produced low density of small-diameter ponderosa pine trees, so a resulting diameter distribution was relatively flat. This differs

from a classical, inverse-J distribution expected for uneven-aged hardwood forests on moist sites lacking a frequent-fire regime (Mast et al. 1999, Powell 2018, White 1985).

Munger (1917) also noted that "yellow pine grows commonly in many-aged stands" (page 16). Historical Blue Mountains inventory data (fig. 22) exhibits a flat diameter distribution expected for uneven-aged stands sustained 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 risk of high-severity wildfire and other uncharacteristic disturbance events. Upon recognizing that risks of no action are probably unacceptable for most scenarios, managers should 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; Churchill et al. 2013a, 2013b, 2017; Franklin et al. 2013; Wright and Agee 2004; and many others in References section).

A 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 onset of fire exclusion (sec. 4), selective cutting (sec. 6), and ungulate herbivory (sec. 5). These three management activities contributed to creating condition class 2 and 3 conditions described and illustrated in table 5 (Barrett et al. 2010, 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, a no-treatment scenario had nearly 30 percent of a landscape in a high-risk category, whereas active management (thinning and prescribed fire) had 100 percent of a landscape in a low-risk category (Wilson and Baker 1998).

No single restoration solution, however, can hope to precisely reproduce 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 (see fig. 19) (Covington 2003).

A challenge is to integrate a suite of active management treatments to effectively and appropriately emulate natural disturbance regimes of dry-forest landscapes (fig. 23). Successfully meeting this challenge will produce a semblance of historical forest structure and species composition – a desirable outcome not because resulting conditions are historical, but because they are sustainable (e.g., vigorous, self-perpetuating, pine-dominated, and at low risk to stand-replacing fire and defoliating insects) (Fiedler 2000a).

Note: thinning dense clumps of ponderosa pine regeneration (see middle panel of fig. 23) will produce a positive growth response in residual trees (Barrett 1963, 1968, 1970) and place them back on a developmental trajectory toward characteristic stand dynamics. Also note that when released by thinning after a long period in a very dense condition, other tree species seldom respond to additional growing space as well as ponderosa pine does.

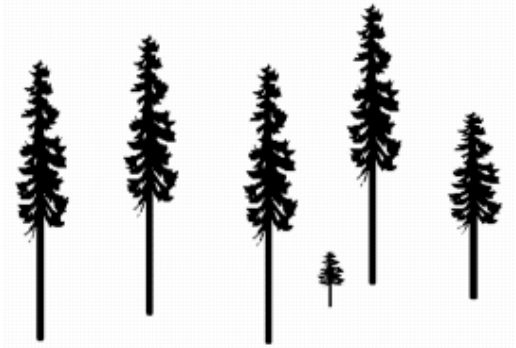
Using burlap to beat out a surface fire in ponderosa pine, Wallowa National Forest, about 1910. As Thornton Munger noted, “Light, slowly spreading fires that form a blaze not more than 2 or 3 feet high and that burn chiefly the dry grass, needles, and underbrush start freely in yellow pine forests. Practically 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 fire’s influence was suppressed during the past 100 years. On many dry sites, fire exclusion had an 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 historical composition and structure will go a long way toward allowing us to restore an ecologically important and valuable disturbance process – frequent surface fire.

Figure 23 – Restoration of ponderosa pine ecosystems (from Powell et al. 2001; top photograph from Boerker 1920, bottom two photographs from USDA Forest Service 2001). Soon after its inception in 1905, USDA 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 a properly functioning ecosystem process (bottom) (Arno and Allison-Bunnell 2002, Arno et al. 1995, Fiedler et al. 1996, Fiedler et al. 1999). Dry-forest landscapes are said to have strong ‘ecological memory’ due to the strength of an interaction between an ecological process (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 5: 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.

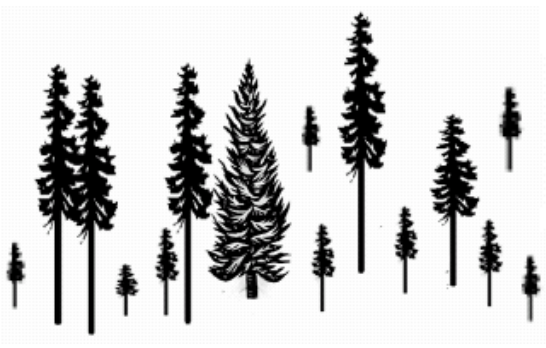
Tree density: stocking levels are within an historical range; density remains consistently below lower limit of a self-thinning zone.

Vigor: high seasonal energy activity; high capacity to repel or resist disturbance agents such as insects and pathogens.

Fire regime: maintained within or near an historical range; no departure from historical frequency or severity (nonlethal fire 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 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 historical range; density may exceed lower limit of a 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 upper half of historical range (10 to 20 tons per acre).

Resilience and risk: fairly high potential to return to condition class 1 by using prescribed fire; moderate risk of losing key ecosystem components following wildfire.

CONDITION CLASS 3

(ecosystem degradation stage)

Composition and structure: highly altered from reference conditions; fire-sensitive species common; open, park-like appearance completely lacking; multi-layer canopy structure.

Tree density: stocking levels exceed 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 by using prescribed fire; mechanical treatments needed before reintroducing fire; high risk of losing key ecosystem components to stand-replacing wildfire.

Note: A strategic assessment for 15 western states (Rummer et al. 2005) found that 30 million acres exist in class 1, 38.4 million in class 2, and 28.5 million in class 3.

7.5 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 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, can compensate somewhat for suppression of an historical surface fire regime by reducing high stand density levels, addressing successional advancement (from early- to late-seral tree species), and jump-starting stagnant nutrient-cycling processes (Gundale et al. 2005, Stephens et al. 2009) (fig. 24).

We should consider, however, that some plant and animal 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 proportion and distribution of seral stages at a landscape scale (White et al. 1999).

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 is wise to thin first and allow old-growth pines to recover their vigor before subjecting them to additional stress from a prescribed fire (Covington 2003, Fiedler et al. 1996, Scott 1998b, Swezy and Agee 1991). Increased vigor translates into increased resin production and bark-beetle defenses (Perrakis et al. 2011).

Much 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 a contractor is paid a specified amount per acre, or per tree, to cut or otherwise treat unwanted trees and leave them on-site (Powell et al. 2001).

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

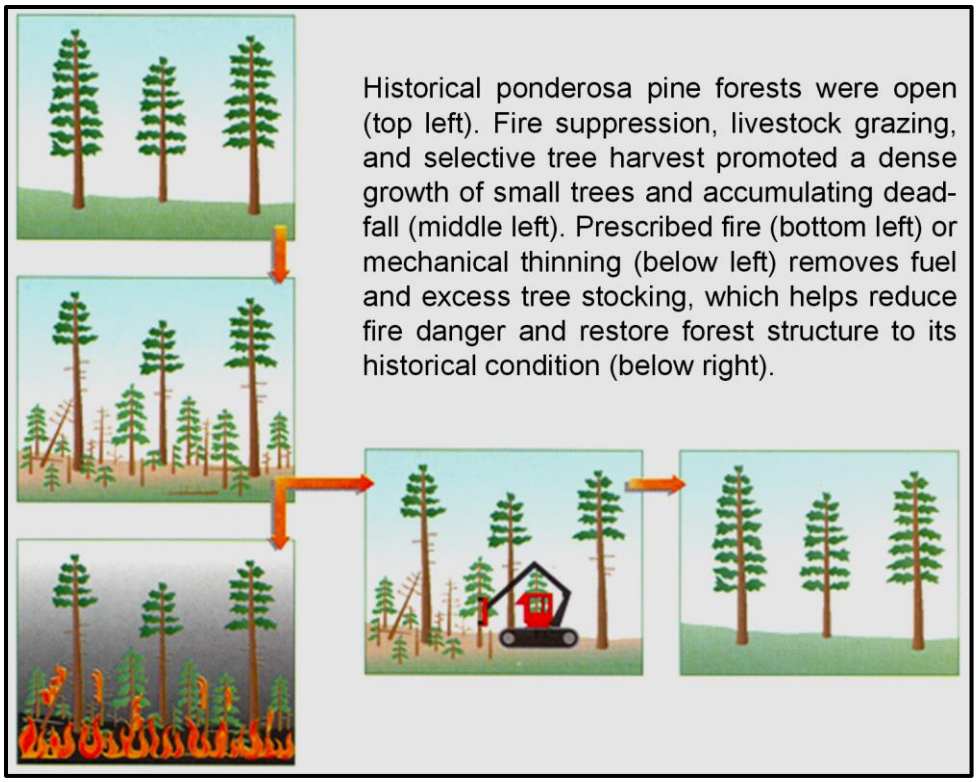


Figure 24 – Correcting a history of fire exclusion (adapted from Phillips 1995). Thinning and prescribed fire are examples of stand-maintaining disturbances that kill from the bottom up (Smith et al. 1997). These treatments need to become more common as one way to address forest health issues resulting from changes in species composition and forest structure caused by fire exclusion, livestock grazing, and selective tree harvest (Johnson et al. 2011, McIver et al. 2013, Stephens et al. 2009, Youngblood 2010).

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 or biochar (Barbour and Skog 1997).

On forest sites in eastern Washington, residual trees increased growth following surface fires that killed trees in intermediate and suppressed crown classes, but growth increases were greater when thinning was used to reduce overall stand density. Unlike fire, manual thinning did not damage fine roots, so residual trees occupied increased growing space quickly. After overstory trees claimed additional growing space provided by 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 concentrate near the soil surface, especially on sites with shallow soils, heat generated by prescribed fire has 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).

Stewardship Tree Harvest

Stewardship tree harvest, depending on techniques used and woody debris left behind, may reduce fuels and wildfire hazard in the near term, or it may not. Harvest alone, without also thinning small unmerchantable trees, treating woody debris produced by 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 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 a treatment. When fire occurred in a thinned stand in Arizona, with woody debris having been removed prior to the fire, fire improved residual-tree resin production as compared to an unthinned control (Feeney et al. 1998). Improved resin production promotes defensive chemical compounds enhancing bark beetle resistance (Kolb et al. 1998); without thinning first, fire could benefit bark beetles more than old trees (Perrakis et al. 2011).

More Use Of Prescribed Fire?

In early 1990s, Bob Mutch and other fire scientists recommended that prescribed fire use (fig. 25) 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 Blue Mountains forest health concerns (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 25 – A prescribed fire burning at night. Using prescribed fire on dry sites is intended to emulate an historical fire regime that sustained open, resilient forests of ponderosa pine. For the Blue Mountains, surface fires with short flame lengths (less than 3 feet) tended to occur at an interval of 5 to 20 years, a fire frequency favoring thick-barked ponderosa pines and western larches while discriminating against 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. Widely used models of dead-tree (snag) and down-log dynamics developed more than 35 years ago for the Blue Mountains (Maser et al. 1979, Thomas et al. 1979) are not fully compatible with ecology of dry-forest disturbance regimes.

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 a snag falls. Fallen snags then become downed logs, which go through another series of five classes of decomposition and decay (Maser et al. 1979) (fig. 26).

Frequent fires on dry-forest sites tended to burn snags before they could progress through all 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 a 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. 26).

For a dry-forest climatic zone of the interior Pacific Northwest, a short-interval fire regime functioned as a 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 nutrient cycling differences between dry and moist forest sites).

When evaluated through a 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 biomass that did accumulate in the most persistent ecosystem component – large, old trees (see figs. 44-46). Due to evolutionary adaptations of dominant trees (thick bark and an elevated canopy), resilience of these ecosystems was high, even when considering the high frequency of low-severity surface fire as a disturbance process.

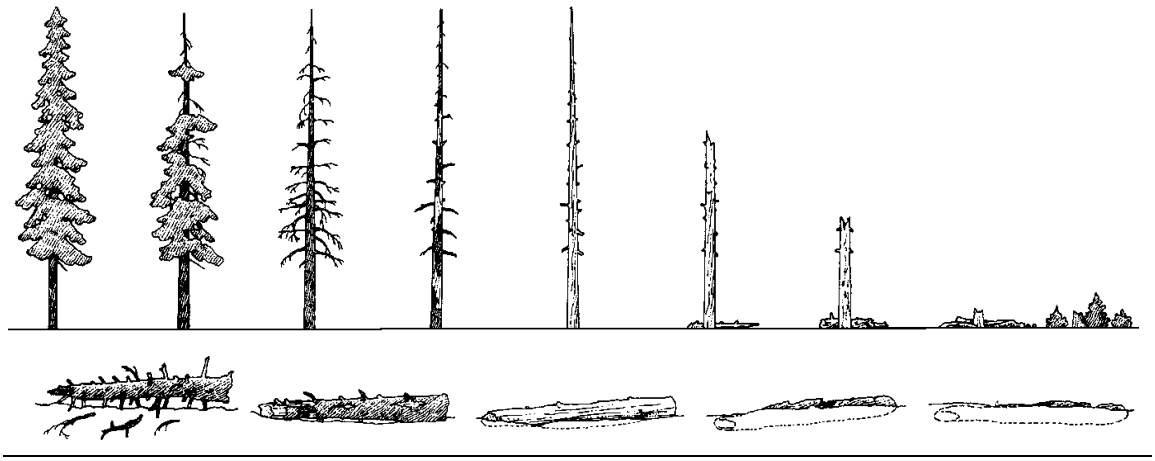


Figure 26 – Diagrams illustrating succession and evolution of snags (top) and down logs (bottom) through time (from Maser et al. 1979 and Thomas et al. 1979). These models of snag and down wood succession are most appropriate for ecological environments where decomposition is primarily accomplished by microbes (e.g., moist and cold upland forests; see fig. 8). They are probably less appropriate for dry-forest environments where woody detritus was cycled mainly by frequent surface fire (Passovoy and Fulé 2006, Stephens and Moghaddas 2005). Note that for dry sites, surface 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. 27), allowing a low-intensity surface fire to climb into upper canopy layers and kill dominant trees, including fire-resistant species (Arno et al. 1997, Steele et al. 1986).

When large quantities of standing dead trees are present following lethal fire on dry sites, salvage harvest is appropriate to remove some portion of this uncharacteristic fuel loading (Harvey et al. 1999, Mutch et al. 1993). [Despite controversy surrounding post-fire salvage harvest (Beschta et al. 2004), I assert that if a dry forest's live-tree density is uncharacteristically high, and if it burns with uncharacteristic fire severity, then resulting dead-tree density is also uncharacteristic, and salvage harvest could be used to reduce the number of dead trees to characteristic levels by retaining 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. "An over-abundance of green trees before a fire becomes an over-abundance of burned logs and snags ready to fuel the next fire" (Everett et al. 1996, p. 272).]

7.6 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. 28; 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).

Figure 27 – Shade-tolerant trees can get established on dry sites in an absence of surface fire (from Powell 1994). Grand firs and Douglas-firs are clustered around the base of a 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 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). A mound of bark flakes at the base of this old pine is an indicator of long-term fire exclusion; fire can smolder there and kill fine roots (Ryan and Frandsen 1991, Swezy and Agee 1991). Note: Experience in American southwest suggests it requires at least 100 years for ponderosa pine to develop a characteristic orange, platy bark shown here (White 1985).

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

Land managers can emulate a natural competition process by intentionally reducing number of trees on a site, a practice called thinning. Thinning has been used to describe activities 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), a reference to thinning is assumed to be "understory thinning, thinning from below, or low thinning," which refer to cutting or removal of subordinate trees (fig. 29; Smith et al. 1997). To capture maximum restoration benefit from thinning, post-thinning stand density should be reduced to a lower limit of the management zone stocking level (figs. 30-31).

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 management*. One contemporary objective is to create fire-safe forest conditions, particularly for developed areas containing wildland-urban interface or other values-at-risk, and thinning addresses three of four fire-safe principles (table 6).

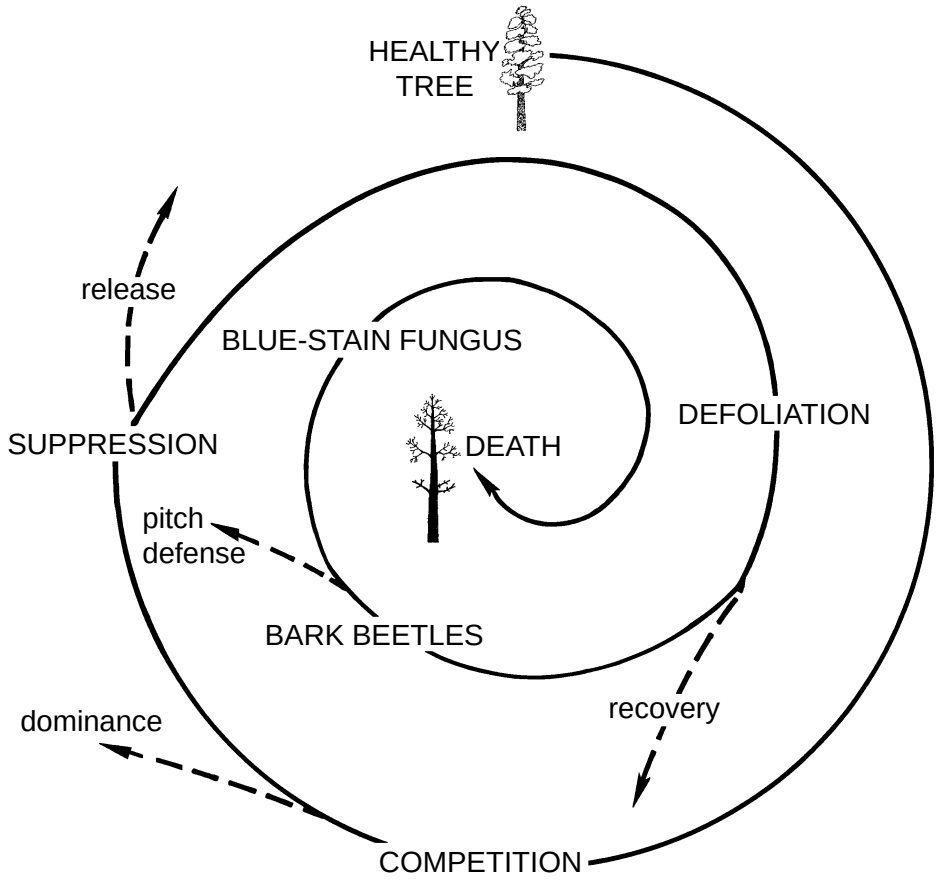


Figure 28 – Death spiral for a Douglas-fir tree in Blue Mountains (adapted from Franklin et al. 1987). In this death spiral, a slightly taller tree suppresses a shorter but otherwise healthy tree. If not released from competition, a suppressed tree is predisposed to attack by defoliators. Once partially defoliated, a weakened tree is attractive to bark beetles, including Douglas-fir beetles (Wickman 1978) carrying blue-stain fungus. Blue-stain 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).

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 these activities implemented in different ways, but selective cutting was directed at short-term (economic) objectives (Ames 1931), while thinning is designed to meet silvicultural objectives. These differences demonstrate that all mechanical treatments are not the same – low thinning is an ideal restoration activity for dry forests, whereas selective cutting contributed to deterioration in the first place.

By removing some trees and increasing space around those that remain, thinning provides more sunlight, water, and nutrients for residual trees. Reducing tree density quickly improves physiological vigor of 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; Perrakis et al. 2011; Stoszek 1988; Vité 1961; Waring and Pitman 1985).

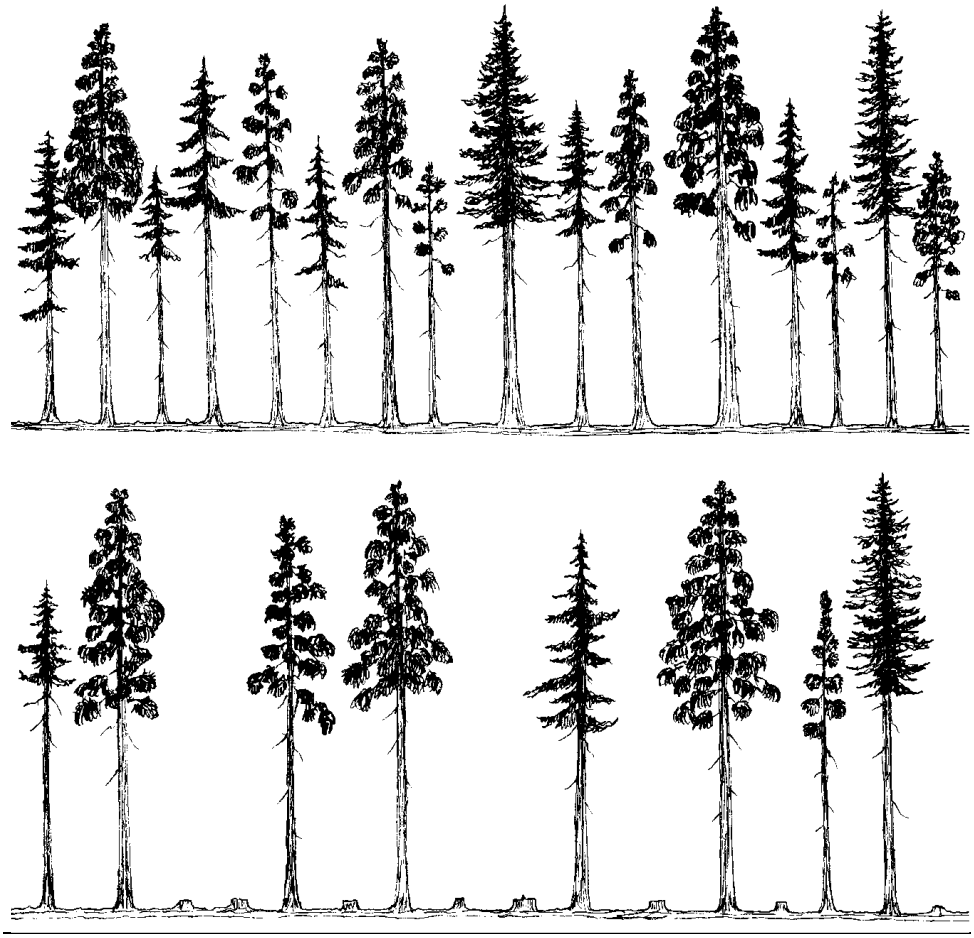


Figure 29 – Example of low thinning in a mixed-conifer forest (from Powell 1999b). Low thinning is defined as removal of trees from lower crown classes or canopy layers in order to favor trees in upper crown classes or layers. Low thinning is also referred to as 'thinning from below.' Note how smaller trees were removed in every instance but one: a western larch at center of top panel was infected with dwarf mistletoe to an extent threatening its continued survival. Because of its canopy position, the larch would not have been removed except for insect or disease reasons.

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).

"The silvicultural practices designed to maintain forest health will be different than those used to produce timber as a primary objective. Smaller material will be removed. There will be more use of thinnings, salvage, and other silvicultural treatments that involve removal of only a portion of the trees on a site. Wood product values will be lower and logging costs higher" (MacCleery 1995). A similar conclusion was reached during an assessment of timber availability from forest restoration in the Blue Mountains of Oregon, when it was noted that thinning might be difficult to accomplish economically due to small tree size (Rainville et al. 2008, p. 63).

Table 6: 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; 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 a landscape scale

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

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 residual trees;
(2) It provides more control over amounts and distribution of standing and down wood as wildlife habitat (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 restoration treatment costs (Barbour et al. 2007).

Guidelines have been developed to identify and describe site-specific levels of intertree competition (stocking), and to relate them to various categories of insect or disease susceptibility (Cochran et al. 1994; Hessburg et al. 1994, 1999a; Lehmkuhl et al. 1994; Powell 1999b; Schmitt and Powell 2005, 2012). These guidelines are commonly used to prepare silvicultural prescriptions for commercial thinnings and other density management treatments in dry forests. A basic density-management concept is this: maintain stands within an ecologically appropriate 'management zone' (fig. 30) to ensure reasonable stand development, high tree vigor, and improved resilience to a wide variety of insect and disease organisms.

Management Note: Unless management objectives dictate a different density management regime, *I suggest that a thinning treatment be initiated when stand density approaches an upper limit of a management zone (which is about 60% of maximum density or 75% of full stocking), and that thinning reduce stand density to a lower limit of a management zone (which is about 35% of maximum density or 50% of full stocking).* Figure 32 provides stocking charts portraying four density-management thresholds (fig. 32a) and two crown-fire susceptibility thresholds (fig. 32b), along with an example of how a stocking chart can be used to compare pre-treatment and post-treatment conditions (fig. 32c).

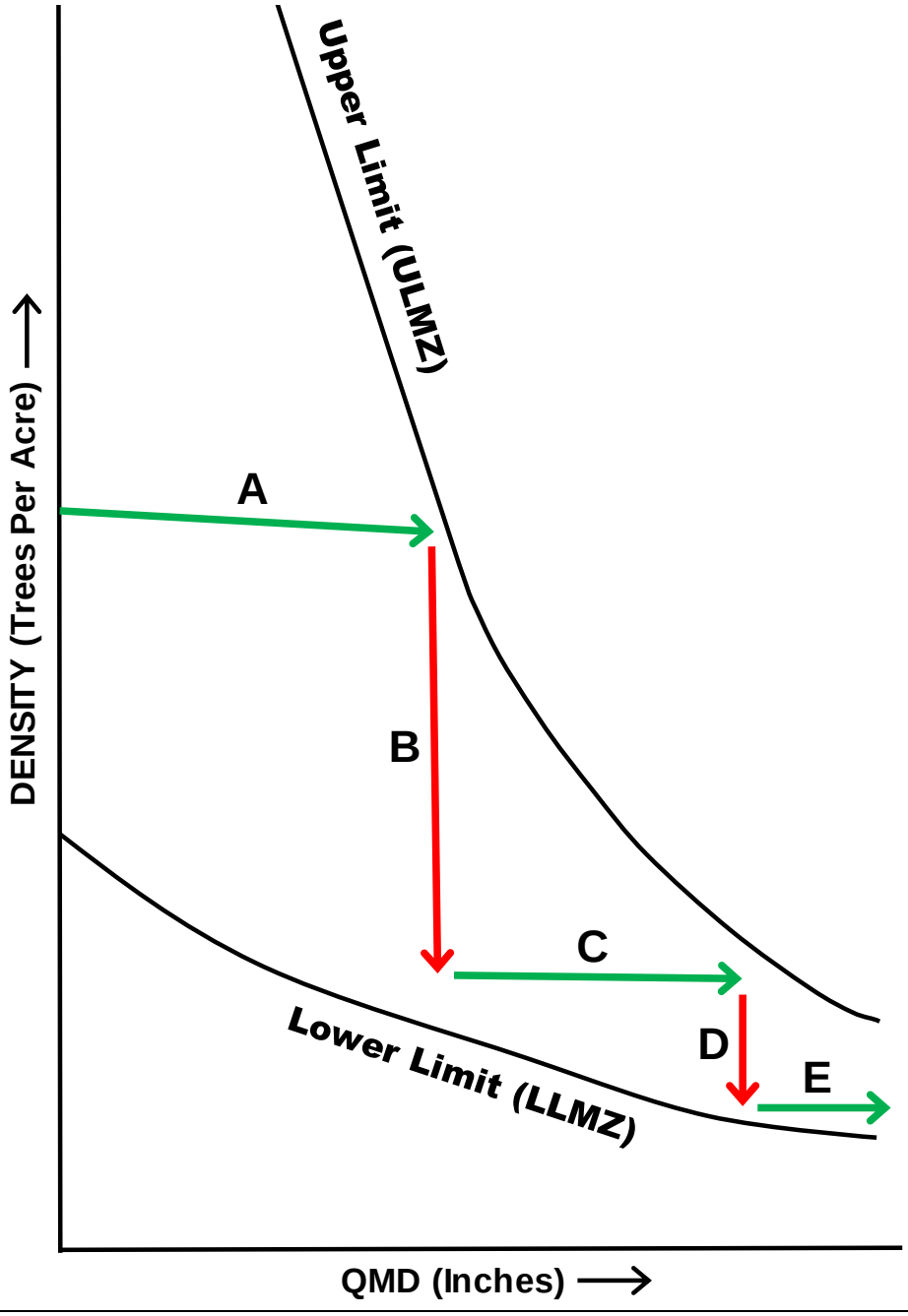


Figure 30 - Hypothetical thinning regime utilizing upper and lower limits of a management zone as stocking curves (curving black lines). This figure shows how a stocking-level chart could be used to prepare a thinning regime. In this example, initial stocking begins within a management zone and stand growth causes QMD to increase toward an upper limit (this is segment A; green segments show growth; red segments show thinnings). When this example trajectory approaches an upper limit, thinning is completed and stocking is reduced until it approaches a lower limit (segment B). Post-thinning growth causes the stand to approach an upper limit again (segment C), at which point a second thinning is scheduled to reduce stand density toward a lower limit again (segment D). For this example regime, stand density would ostensibly be low enough to stay within a management zone after completing the second thinning (segment E).

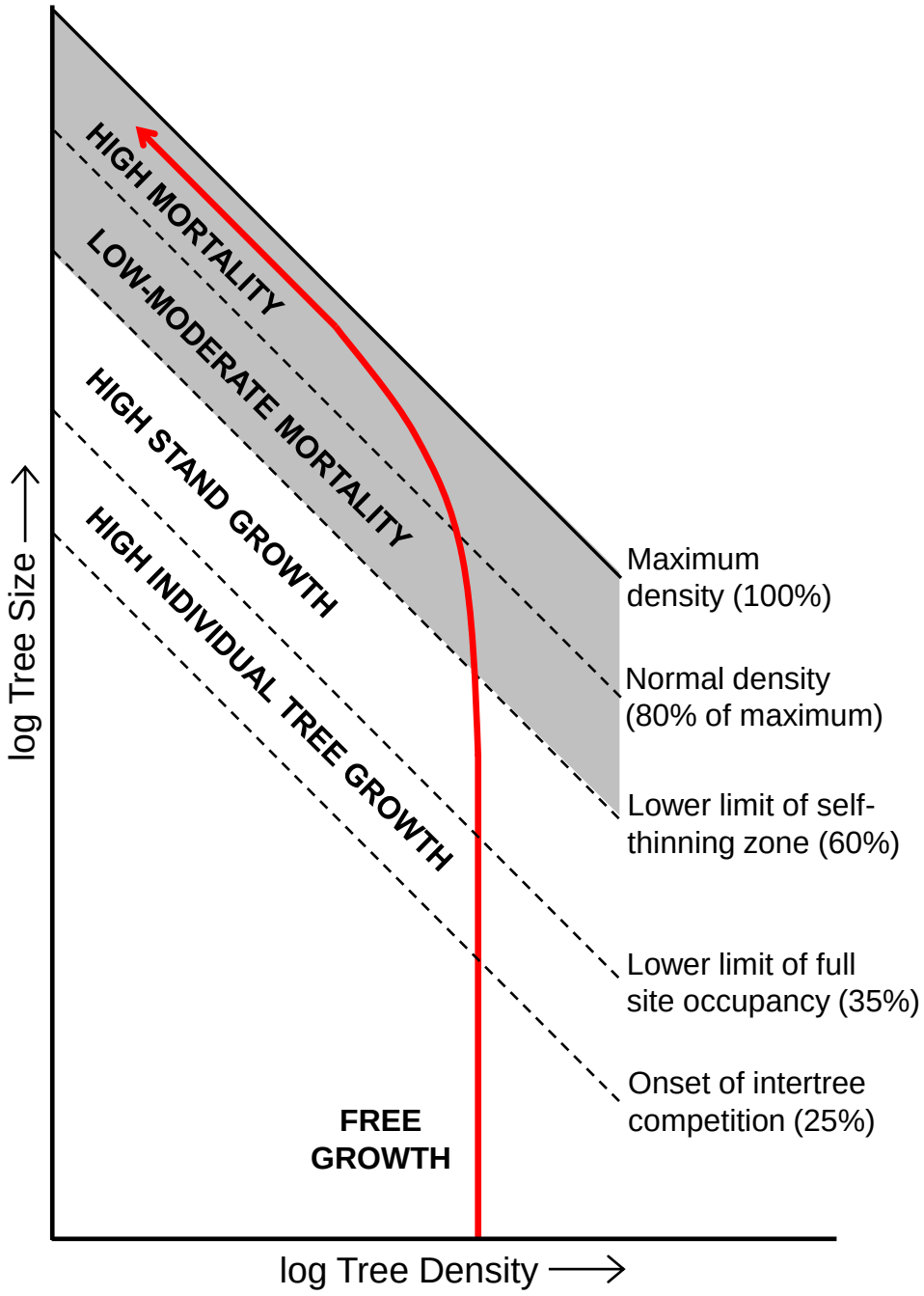


Figure 31 – Stand development indexed to maximum density. Initially, trees are too small to use all of a site's resources, and they experience a period of free growth (no intertree competition is occurring). Eventually, roots and crowns begin to interact and an 'onset of intertree competition' threshold is reached. As a stand continues growing through a zone of high individual tree growth, trees capture growing space and a 'lower limit of full site occupancy' threshold is breached. This next zone features high stand growth. As competition intensifies, stands eventually enter a self-thinning zone by crossing a 'lower limit of self-thinning zone' threshold. In a self-thinning zone (gray area), a tree can only increase in size if neighboring trees relinquish their growing space by dying. The pace of tree mortality quickens as a stand passes a 'normal density' threshold and approaches maximum density. Maximum density, shown as a solid line because it is an absolute threshold, is a reference level (100%) for the stocking system described here.

Selective cutting. A system in which groups of trees, or individual trees, are periodically removed from a 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) or stocking control.
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). [Note: selective and selection cutting are not the same practice!]
Thinning. A treatment designed to reduce tree density and thereby improve growth of residual trees, enhance forest health, or recover potential mortality resulting from intertree competition. Two types of thinning are recognized – commercial thinning (trees being removed have economic value), and noncommercial thinning (trees are too small to have economic value, so they are 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 fire to a predetermined area, and to produce a fireline intensity and rate of spread meeting land management objectives (Powell et al. 2001).

Explanatory Notes for Figures 31-32, and Table 7. Figure 32 provides stocking-level tools for active management of dry upland forest. Figure 32 has three parts – conventional stocking levels expressed by using four stand density thresholds (fig. 32a), 'special-purpose' stocking levels expressed by using two levels of crown-fire susceptibility (fig. 32b), and an example of how stocking-level charts can be used to assess treatment effectiveness (fig. 32c). Figures 32a-32c assume an even-aged stand structure (e.g., SDI was not reduced to account for irregular or other structures). Notes about threshold levels shown on the stocking charts (figs. 32a and 32b):

Maximum density: Although rarely observed in nature, maximum density represents a useful upper limit, so it is often used when establishing stocking levels.

Full stocking (80% of max): Full stocking is also referred to as normal density. Full stocking pertains to single-cohort (even-aged) stands where intertree competition results in crown-class differentiation – dominant, codominant, intermediate, and subcanopy trees are evident in differentiated stands. As shown in figure 31, normal density/full-stocking occurs in a self-thinning zone where stand density is high enough to cause tree mortality.

Upper limit of a management zone (60% of max; Upper Limit in fig. 32a): This stocking level corresponds with a 'lower limit of self-thinning zone' threshold shown in figure 31. It is often used whenever land managers wish to avoid density levels high enough to cause self-thinning and competition-induced tree mortality.

Lower limit of a management zone (35% of max; Lower Limit in fig. 32a): This stocking level corresponds with a 'lower limit of full site occupancy' threshold shown in figure 31. This threshold functions well as a lower limit because a site is fully occupied at stocking levels above it – growing space is not being underutilized ('wasted') at these stocking levels.

High susceptibility to crown fire (High Susceptibility in fig. 32b): This stocking level pertains to stand densities where crown fire is easily sustained – namely, canopy bulk density (CBD) values of 0.10 kg/m^3 or more (Agee 1996c).

Low susceptibility to crown fire (Low Susceptibility in fig. 32b): This stocking level pertains to stand densities where crown fire is either impossible or highly unlikely – namely, canopy bulk density (CBD) values of 0.05 kg/m^3 or less (Alexander 1988, Van Wagner 1977).

CBD values (0.10 and 0.05 kg/m^3) for crown-fire susceptibility were translated into their corresponding forestry metrics (Powell 2010) in order to prepare a stocking chart (fig. 32b).

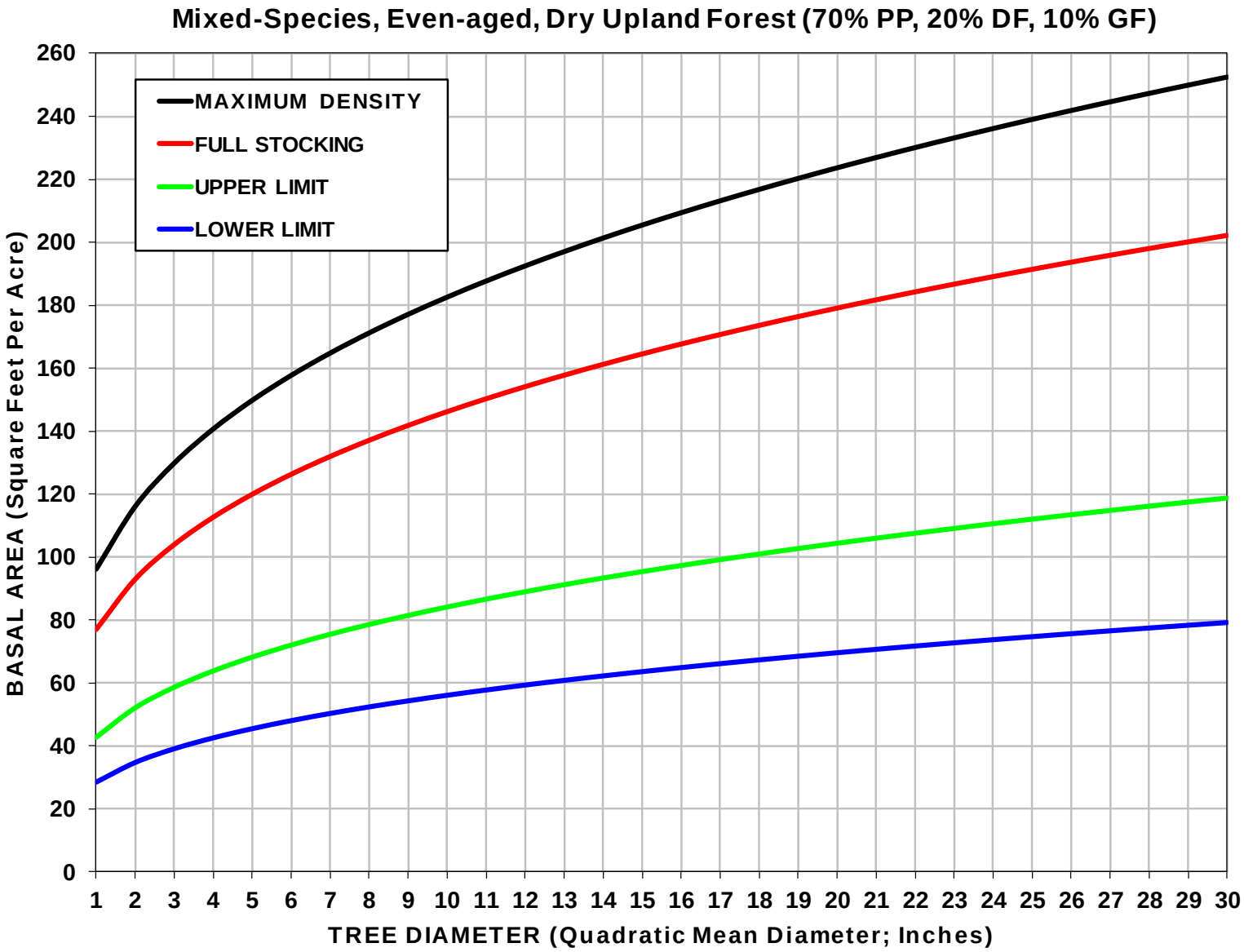


Figure 32a – Stocking chart for dry forests, expressing four stand density thresholds (color lines) by using basal area and QMD values.

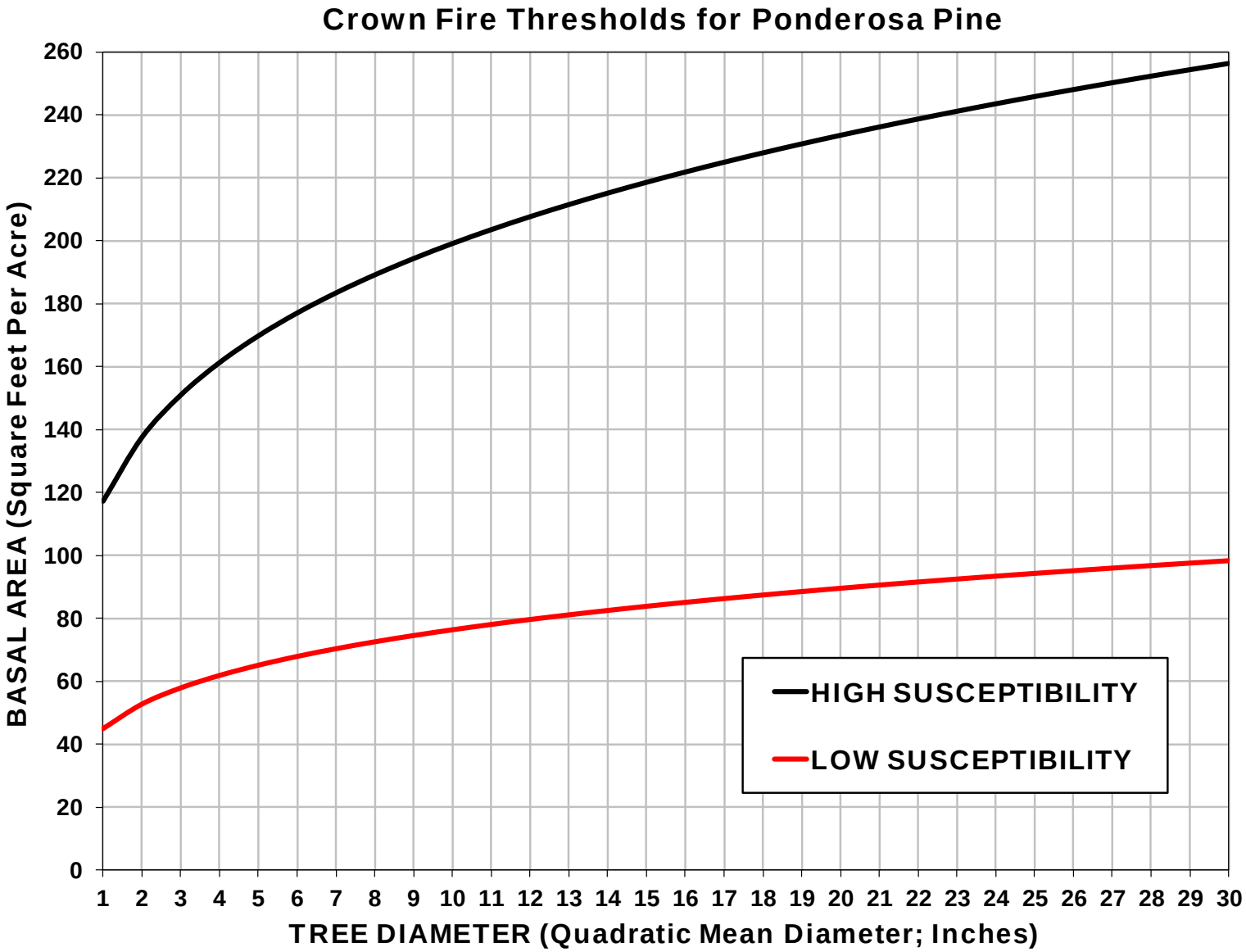


Figure 32b – Stocking chart for dry forests, showing two crown-fire susceptibility thresholds (color lines) by using basal area and QMD values.

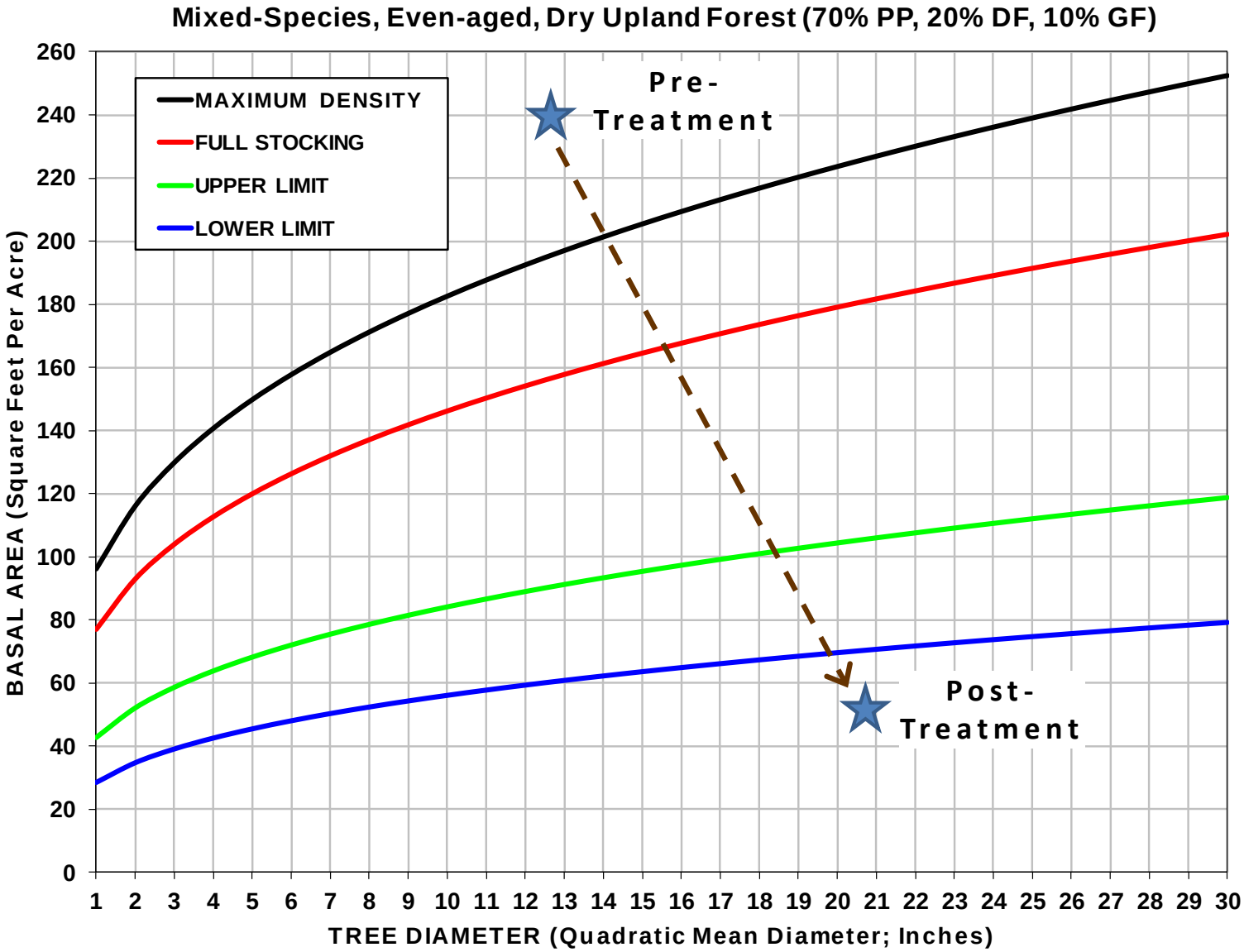


Figure 32c – Did a density-management treatment successfully move stand density within a management zone? In this example, treatment reduced stand density below a Lower Limit (pre-treatment = 238 ft^2/acre and 12.6" QMD; post-treatment = 50 ft^2/acre and 20.7" QMD).

Table 7: Recommended stocking levels for the dry-forest PVG, as expressed by using the stand development zones depicted in figure 30.

	CLIMATE CHANGE (WARMER, DRYER)		HIGH TREE GROWTH		HIGH STAND GROWTH		LOW-MODERATE MORTALITY		HIGH MORTALITY	
	0-25% OF MAX SDI		25-35% OF MAX SDI		35-60% OF MAX SDI		60-80% OF MAX SDI		80-100% OF MAX SDI	
	TPA	BAA	TPA	BAA	TPA	BAA	TPA	BAA	TPA	BAA
Ponderosa pine[1]	0-76	0-41	76	41	76-114	41-62	114-241	62-131	241-301	131-164
Douglas-fir	0-84	0-46	84-135	46-74	135-202	74-110	202-270	110-147	270-337	147-184
Western larch	0-80	0-44	80-128	44-70	128-192	70-105	192-256	105-140	256-320	140-175
Grand fir	0-142	0-77	142-226	77-123	226-340	123-185	340-453	185-247	453-566	247-309
Mixed composition[1]	0-84	0-46	84-103	46-56	103-154	56-84	154-268	84-146	268-335	146-183

Notes: Stocking levels are means for 24 plant associations assigned to a dry-forest potential vegetation group (PVG). They are expressed as percentages of maximum stand density index (SDI). TPA is trees per acre, and BAA is basal area (square feet) per acre; both metrics pertain to even-aged stands. Stocking levels presented in this table should be reduced by 7% for an irregular structure, and by 13% for an uneven-aged structure (and note that either of these structures is more common on dry-forest sites than an even-aged structure). TPA and BAA stocking levels pertain to a 10-inch quadratic mean diameter (QMD) – they will differ for a QMD other than 10 inches.

[1] For ponderosa pine, upper limits of 'high tree growth' and 'high stand growth' zones are calculated by using a process accounting for mountain pine beetle susceptibility (see Cochran et al. 1994), so they are not 35% and 60% of the mean maximum density SDI value for ponderosa pine (301). For mixed composition, stocking levels represent weighted averages (70% ponderosa pine, 20% Douglas-fir, and 10% grand fir).

Stocking-level categories, as depicted by using differing colors for column headings, have the following interpretations.

- 'Climate change' category is equivalent to 'free growth' zone in figure 31. Much climate change research suggests the western United States could become significantly warmer and dryer as climate changes continue, so climate-change stocking levels are lower than conventional levels shown to the right of them. (White paper #45, "Climate change and carbon sequestration," discusses climate change in more detail.)
- 'High tree growth' zone spans 25 to 35 percent of maximum density. A 25% value corresponds to 'onset of intertree competition' stocking threshold; a 35% value is 'lower limit of full site occupancy' threshold (fig. 31). For the Blue Mountains, 'lower limit of full site occupancy' threshold is used as a 'lower limit of a management zone' (see fig. 30) (Cochran et al. 1994, Powell 1999b).
- 'High stand growth' zone spans 35 to 60 percent of maximum density. A 60% value corresponds to 'lower limit of self-thinning zone' stocking threshold shown in figure 31. It is used as an 'upper limit of a management zone' (see fig. 30) (Cochran et al. 1994, Powell 1999b).
- 'Low-moderate mortality' zone spans 60 to 80 percent of maximum density. An 80% value corresponds to 'normal density' stocking threshold shown in figure 31.
- 'High mortality' zone spans 80 to 100 percent of maximum density. A 100% value corresponds to maximum density shown in figure 31.

Regardless of whether mechanical thinning or prescribed fire is used as a forest restoration activity, fuel treatments need to account for wildlife needs. For example, research found that treated stands provide better elk forage during spring, whereas untreated stands provide better summer forage, suggesting that a mosaic of treated and untreated areas may provide better elk foraging habitat than treating a large proportion of a landscape (Long et al. 2008).

White-headed woodpecker, however, prefers lower- and mid-elevation ponderosa pine forests on flat or gently sloping terrain. Two features of dry forest are important for this species: availability of snags or cavity trees for nesting, and abundant ponderosa pine cones to provide seeds as a food source during winter (see Box 1) (Buchanan et al. 2003).

Restoration Considerations

Although contentious debate about 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 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 dry-forest 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.** 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 extent to which environmental regulations are inhibiting forest restoration.** Legislation such as Endangered Species Act and 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 wildland-urban interface (WUI).** This is more a political issue than a forest health issue, but presence of WUI (and other values at risk) 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 combined effect of domestic and wild ungulates 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.** Computer models, decision support systems, and visualization systems can be used to help balance public expectations for forest uses with a need to reestablish landscapes having characteristic levels of fire, insect, or disease hazard. We must identify landscapes with highest priority for restoration treatments, and then seek to create a vegetation mosaic that functions within its range of variation.

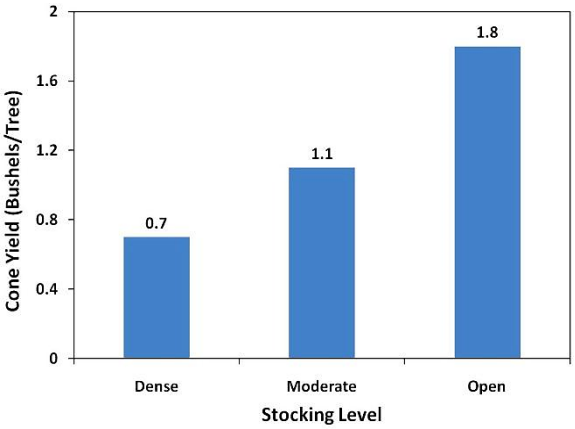
Box 1: Stand Density and White-Headed Woodpecker

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 a 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 is the currently uncommon white-headed woodpecker (*Picoides albolarvatus*). In Oregon's portion of this woodpecker's west-wide range, ponderosa pine cones are believed to provide a primary food source during winter, non-breeding periods (Buchanan et al. 2003, Garrett et al. 1996).

Dry-forest restoration activities in Washington focusing 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 winter 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 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 cone production benefits of active management on dry sites: when wildfire occurred in a thinned stand in Arizona, and 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, a 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.



7.7 Restoring Old Forest On Dry Sites

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 differing disturbance regimes on these biophysical environments (Camp et al. 1997, Everett et al. 1994, Habeck 1990, O'Hara et al. 1996, Oliver and Larson 1996).

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 frequent surface fire, multi-layer stands with large trees in an 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 a process of enabling a system to resume acting, or continue to act, following disturbance as if disturbance had not occurred (Powell et al. 2001).

On dry mixed-conifer sites, frequent surface fires historically interrupted plant succession toward a climatic climax, thereby preventing eventual domination by Douglas-fir or grand fir. This short-interval fire regime maintained an early-seral species composition consisting of ponderosa pine (fig. 33); 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). An old forest structure produced by frequent fire is termed old forest single stratum (table 8).

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 1998b). Fire frequency must be maintained at an appropriate periodicity if ponderosa pine is to persist, and this is a 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. 34). This park-like structure did not occupy an 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).

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 (skips); and (3) creating small gaps (up to ½ acre in size) in some 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).

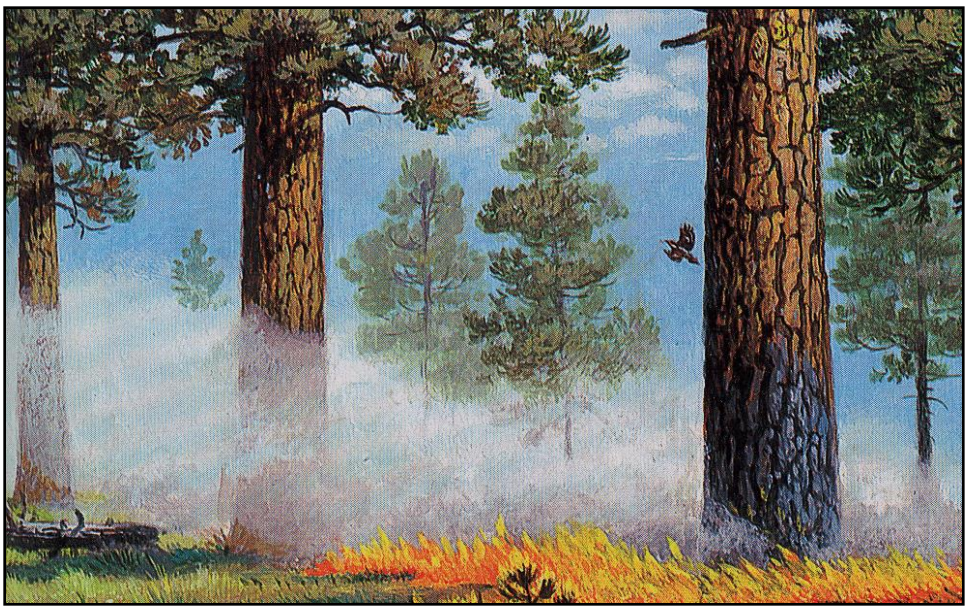


Figure 33 – Low severity surface fire in ponderosa pine forest (from Powell et al. 2001). In eastern Oregon, a presettlement fire regime created stable old forest referred to as 'park-like pine forest.' These ecosystems featured big, widely spaced ponderosa pines above a dense herb layer (also see fig. 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 34 – 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 replacement tree species provide habitat for these insects (Mason and Wickman 1994, Wickman 1992).

Table 8: Description of forest structural stages.

	**Stand Initiation**. Following stand-replacing disturbance, growing space is occupied rapidly by vegetation that either survives a disturbance or colonizes an area. Survivors survive a disturbance above ground, or initiate new growth from underground organs or seeds present onsite. Colonizers disperse seed into disturbed areas, it germinates, and new seedlings establish. One stratum of tree seedlings and saplings is present in this stage.
	**Stem Exclusion**. Trees initially grow fast and occupy their growing space, competing strongly for sunlight and moisture. Because trees are tall and reduce light, understory plants 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, creating 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 multi-layer stands with large trees in an uppermost stratum (old forest multi strata).

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

Many land managers agree that fire exclusion was a policy with good intentions, but it failed to consider ecological implications of a major shift in species composition. Grand firs and Douglas-firs can get established under ponderosa pines when fire is absent, but they may not have enough resilience to make it over the long run, let alone survive the next drought. This means that many 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)

Principles Of Old Forest Restoration

I believe an old-forest restoration program for dry upland forests of the Blue Mountains should incorporate these concepts relating to 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.
- An individual old-forest patch has limited landscape contribution, so additional old-forest stands must be continually created to maintain a dynamic old-forest equilibrium with time.
- Efforts to conserve old forest should not sacrifice contributions from other limited structures or components in a landscape.
- Conserving disturbance processes influencing 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 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).
- When managing old forests, a landscape perspective is needed that coordinates wildlife species requirements with ecological processes and other functional ecosystem attributes (see Box 1) (Hessburg et al. 2005).
- Forest ecosystems of interior Pacific Northwest exist in a constant cycle of change; it should be acknowledged that successional pathways for a certain proportion of forest stands will be interrupted by fire, windthrow, insect attack, or disease before they reach an old-forest condition.

Strategies And Tactics For Restoring Old Forest

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

- Conservation of remaining old-forest patches is a 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, such as tree 'defect' as shown in figure 35, can partially function as old forest for any attributes that are present.
- Dry old-forest differs dramatically from west-side Douglas-fir/hemlock old-growth (see Franklin et al. 1981), and it should not be evaluated by using west-side criteria. Managers should not use moist or cold wildlife species (such as marten) as dry, old-forest indicators.
- In some parts of a 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 regarding 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 a large-tree component of old forest (Sullivan et al. 2001, Tappeiner et al. 1997).
- When identifying candidates for future 'old forest multi strata' stands (tables 8, 10) in 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 function as semi-stable environmental settings (Camp et al. 1997).
- Although mid- to late-seral stands are 'in a pipeline' to replace old forests lost to disturbance, we still do not know an appropriate ratio of late-seral to old forest patches to ensure that desired levels of old forest are maintained in perpetuity (but see section 7.9, RV).
- Evaluating historical amounts of old forest (as is done when analyzing a range of variation for forest structural stages: table 10) 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 spatial and temporal disturbance regime characteristics have been accounted for.
- A successful strategy would allow flexibility in specific on-the-ground locations over time. A '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 a foundation of an old-forest strategy – an ecologically sustainable representation of old forest structure in a 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 treatment across a portion of a landscape provides considerable reduction in overall landscape fire risk, although it may not lower 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 adjoining stands apparently provides minimal reductions in fire risk, although reducing surface fuels by using a combination of thinning and prescribed fire might provide at least a modicum of fire protection (Wilson and Baker 1998).
- Low thinning and prescribed fire in an old-forest stand will lower its fire risk substantially (Johnson et al. 2011, McIver et al. 2013); however, some of a stand's old-forest characteristics (such as multi-layered canopies) are altered by such practices (Wilson and Baker 1998).
- Thinning and prescribed fire can be especially valuable for sustaining high vigor levels for old ponderosa pines, and high vigor translates into increased resin production and chemical defenses against western pine beetles (fig. 36) and other bark beetles (Perrakis et al. 2011).

Figure 35 – Ponderosa pine 'character' tree on Pomeroy Ranger District of Umatilla National Forest. One focus of dry-forest restoration is to retain these character trees in recognition of their value to wildlife. Old character trees have distinctive ecological characteristics, including unusual shapes formed in response to both physical and biotic damage (wind, mechanical abrasion, disease, parasites, and insects). Note how an old fire scar at this tree's base is now decayed, providing habitat for cavity-dependent wildlife species. Contorted, upper-crown branches, horizontal crown branching, and multiple ('bayonet') tops are indicative of this tree's wildlife value (Van Pelt 2008).

Figure 36 – Old ponderosa pine killed by western pine beetle. Journal articles emphasize the importance of restoring historically appropriate intrastand structure (Larson and Churchill 2012, Larson et al. 2012). A feature of historical, dry-forest structure is clumps of mature trees. Large trees in closely-spaced clumps are under enough stress to function as focus trees (Eckberg et al. 1994) for western pine beetle. Historically, large-tree clumps occurred in a vegetation mosaic where openings were common; a clump embedded in an herbaceous opening (savanna condition, as described in Munger 1917) experienced a much different competitive environment than a clump surrounded by trees.

Scale Considerations For Dry Forest

Scale is fundamentally important, and scale considerations permeate all aspects of active management (Cumming et al. 2013). For example, many recent journal papers from landscape ecology and forest management literature emphasize the importance of spatial heterogeneity (Franklin et al. 2008, 2013; Hessburg et al. 1999b, 1999c, 2000, 2004, 2007; Turner et al. 1989, 1994, 2001; and others in References section).

When evaluating spatial heterogeneity for dry forests, it is important to consider a sub-stand level because scale is fine-grained and intricate for dry forests – ponderosa pine stands historically featured a groupy or clumpy structure at a sub-stand scale (Harrod et al. 1999; also see Powell 2019b). A stand functions as an 'aggregating' level because a consistent but repeating pattern of groups or clumps could be collected (aggregated) within a common stand boundary. In this context, sub-stand clumps function as a fine-scale, base-level unit, reflecting ecosystem pattern and process, but a stand functions as an aggregating unit (e.g., stands are a mid-scale unit representing aggregations of sub-stand clumps). [I define 'base-level' as a scale at which ecosystem processes result in tree regeneration sufficient to perpetuate a forest type.]

Note about clumps: Tree clusters are a feature of many forest ecosystem types. For dry forests, tree clusters can be created by a disturbance regime – variable thinning provided by surface fire, or pockets of tree regeneration after western pine beetle attack (see figures 44-46 later in this paper), in which case clusters function well as an indicator of ecosystem function and process. But we need to remember that dry-forest clusters can also be caused by seed caching activity of deer mice, chipmunks, and other small mammals (Keyes et al. 2007).

Practitioners should be able to interpret spatial pattern in order to understand if it should be emulated by proposed treatments. Does pattern reflect inherent ecosystem process, in which case it would be repeatable across a landscape? Or, is it simply a product of random historical circumstances that may not be repeated again? Answers to these questions are important because contemporary science emphasizes provision of spatial heterogeneity, but:
It is most important to provide heterogeneity for biophysical environments where heterogeneity was a normal 'byproduct' of a properly functioning disturbance regime.

Scale's fundamental importance also provides a useful context for evaluating existing conditions of species composition, forest structure, and stand density. Compositional or structural changes need to be evaluated at a sub-stand scale for dry forest (e.g., at the scale of a tree clump or cluster) – Is a characteristic clumpy structure still evident for a dry forest stand? If so, does composition of any particular dry-forest stand feature a majority of ponderosa pine rather than fire-sensitive (late-seral) species such as grand fir?

Dry-forest composition could be evaluated this way: up to 70% of tree clumps in a dry-forest stand should have a predominance of ponderosa pines, rather than a majority of Douglas-fir or grand fir (because properly functioning surface fire produced high percentages of ponderosa).

Dry-forest conditions have changed dramatically as a result of fire exclusion, livestock grazing, and selective timber harvest – and these changes are overtly expressed in existing conditions, including at a clump scale, so when entering most dry-forest stands, it quickly becomes

apparent when they are substantially departed from reference conditions (see page 44). *Unfortunately, we've taken an ecosystem sustained by fire, and converted it to one destroyed by fire*.

7.8 Active Restoration Of Dry Forests: Wildlife Considerations

A primary purpose of this white paper is to examine silvicultural considerations related to active management of dry-forest ecosystems. Dry-forest management is also influenced by wildlife concerns, and some of them are discussed in this section (Box 1 in section 7.6 also provides a wildlife discussion). An informative synthesis of fuel-reduction and fire-surrogate treatments was recently published, and because it provides an excellent summary of wildlife-silviculture interactions, I decided to include it here (Stephens et al. 2012b, p. 553-557). Note that any literature citations in quoted material below are included in this white paper's References section. [A recent wildlife synthesis journal paper was published for moist, mixed-conifer forests (Irwin et al. 2018), and it also includes useful insights.]

"In addition to its use in managing wildfire hazards, the application of prescribed-fire and fire-surrogate treatments is frequently motivated by wildlife-habitat objectives (Yager et al. 2007, Kennedy and Fontaine 2009, Roberts et al. 2010). Research on fire and its effects on terrestrial vertebrates (wildlife) has been conducted since the early 1900s, beginning with research showing the negative effects of fire exclusion in longleaf pine (*Pinus palustris*) forests on northern bobwhite (*Colinus virginianus*; Stoddard 1931). Since then, a large body of work has been developed, particularly in the last 10-15 years (Kennedy and Fontaine 2009), which has shown that many wildlife species depend on fire-maintained habitats or pyrogenic structures, such as the snags, shrubs, and bare ground created by fires of varying severity (Hutto 2008).

Increased applications of fuel-reduction treatments, public scrutiny of land management agencies, and a growing scientific literature on the topic motivated a recent comprehensive review and meta-analysis of the fire-wildlife literature from forests dominated by low- to moderate-intensity fire regimes (Kennedy and Fontaine 2009, Fontaine and Kennedy 2012). On the basis of the characteristics of the available literature, fuel-reduction treatments and high-severity fire were considered at 0-4 years post-treatment. A lack of published longer-term (more than 5 years) studies precluded any analyses of longer-term effects. Importantly, the only thinning treatments included in this analysis were those conducted for fuel reduction, which is generally a lower-intensity treatment (e.g., the median reduction in basal area for the FFS Study was 30%; Schwilk et al. 2009) than those implemented for other silvicultural objectives (see Vanderwel et al. 2007 for a detailed meta-analysis of avian responses to a broad range of thinning intensities).

Data from low- and moderate-severity fires were pooled, because neither of these treatments resulted in a large canopy loss (less than 50% canopy mortality, less than 25% in almost all cases), and there are insufficient studies of mixed-severity fire to warrant separation. These categories allowed for a comparison of vertebrate responses (mean abundance, density, and vital rate in treated and reference conditions) to fire surrogates combined with fire, as well as differing levels of fire severity (measured by overstory tree mortality). Data were more abundant for birds than for any other taxon (fig. 37), which underscores a need for further work on other wildlife taxa – particularly herpetofauna, which reside primarily on the forest floor.

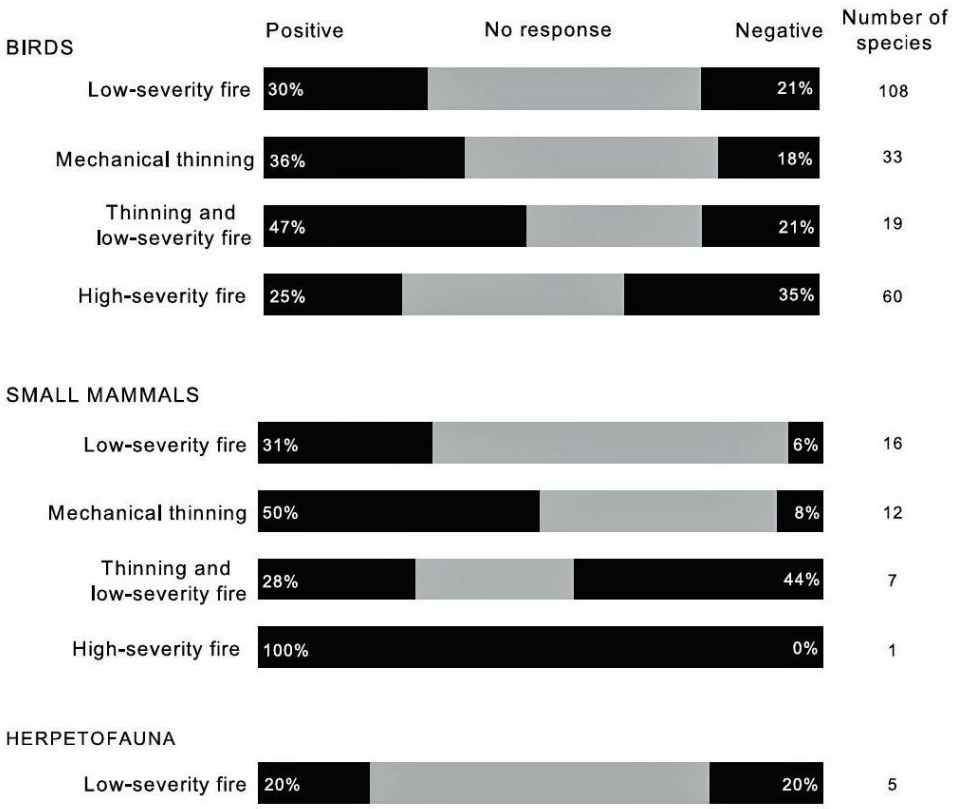


Figure 37 – The responses (positive, neutral, and negative; number of species with sufficient data) of birds, small mammals, and herpetofauna to fire and fire-surrogate treatments 0-4 years after fire treatment in seasonally dry forests of the United States (this is fig. 3 from Stephens et al. 2012b). The response classification was based on a meta-analysis of existing literature and generation of cumulative effect-size estimates and their 95% confidence intervals with overlap (neutral) or not (positive, negative) with zero.

This similarity in the responses of birds and small mammals to thinning and low-severity prescribed fire suggests that, at the stand scale and in the short term (0-4 years), thinning may adequately mimic low-severity fire in terms of its effects on these taxa. The levels of regeneration of vegetation, fuel dynamics, and nutrient cycling following prescribed fire and following thinning differed substantially (Boerner et al. 2009, Schwilk et al. 2009), but thinning or low-severity prescribed fire have the potential, in the short term, to create forests with similar structure and with habitat conditions favored by many wildlife species. Therefore, the results suggest that the use of thinning in lieu of prescribed fire may be warranted for birds and small mammals, particularly in areas in which the implementation of prescribed fire is problematic. However, the long-term effects of these two treatments on wildlife require further investigation before these results can be fully integrated into management.

Research illustrates that these fuel treatments do not create conditions suitable for all species (see negative responses in fig. 37). Additional analyses demonstrate that low- to moderate-severity surface fire (and presumably its thinning surrogate) does not mimic the early successional habitat conditions created by high-intensity, patchy, stand-replacing fires. When it is feasible, managers may aim for patchy high-intensity prescribed fire to mimic the effects of wildfire

(Fulé et al. 2004a). In short, there is no one-size-fits-all prescription when it comes to incorporating disturbances into land management (i.e., there is a need for the presence of all successional stages within a forested landscape to maximize wildlife diversity; Fontaine et al. 2009).

The wildlife literature, which is dominated by studies on birds and small mammals, demonstrates that in the short term and at the stand scale, fire-surrogate forest-thinning treatments effectively mimic low-severity fire, whereas low-severity fire is not a substitute for high-severity fire (Kennedy and Fontaine 2009)."

7.9 Range Of Variation As A Restoration Framework

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

Range of variation (RV) is an analytical technique to characterize inherent variation in ecosystem composition, structure, and function, reflecting recent evolutionary history and dynamic interplay of biotic and abiotic factors (fig. 38). "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 from major influence by Euro-American humans, thereby providing an insight into ecosystem resilience (Kaufmann et al. 1994). It helps us understand what an ecosystem is capable of, how historical disturbance regimes functioned, and underlying variation in ecosystem processes and functions – patterns, connectivity, seral stages, and cover types produced by ecological processes operating at a landscape scale (USDA Forest Service 1997).

Perhaps an effective yardstick for evaluating 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 by using a metric, like RV, that explicitly accounts for change. Resilient forests not only tolerate periodic disturbance, they depend on it for rejuvenation and renewal (Johnson et al. 1994). Significant changes in magnitude (extent), intensity, or pattern of disturbance, however, may be warning signals of impaired ecosystem integrity (Sampson and Adams 1994).

Range of variation concept has been proposed as a way to identify restoration needs and opportunities. Using reference conditions to guide restoration programs will continue into the future because this approach is explicitly required by certain laws governing dry-forest management, such as 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://www.gpo.gov/fdsys/pkg/PLAW-108publ148/pdf/PLAW-108publ148.pdf).

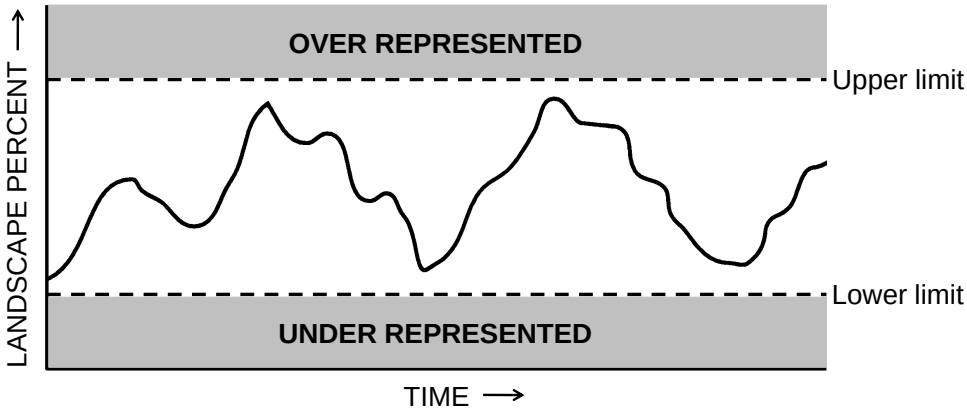


Figure 38 – A range of variation (RV) helps us decide whether existing amounts of vegetation composition, structure, and density, when summarized for a landscape-scale analysis area, are occurring within a characteristic range (Aplet and Keeton 1999, Morgan et al. 1994, Swanson et al. 1994). This diagram shows an ecological trajectory for an ecosystem component (the solid line); it varies through time because the phrase 'range of variation' is meant to encompass more than just extreme values (e.g., upper and lower limits, shown as dashed lines) (diagram modified from Morgan et al. 1994).

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

Both now and in the future, a desirable landscape condition for Blue Mountains province is a diverse, heterogeneous vegetation mosaic more consistent with a 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 an ultimate objective being maintenance of dynamic patterns and processes integral to resilient ecosystems (Aplet and Keeton 1999).

Dry-forest RV information for species composition, structural stage, tree density, and insect and disease susceptibility is provided in tables 9-12. Information in table 9 expresses percentages of a dry-forest landscape (preferably at least 15,000-35,000 acres in size) occupied by various vegetation cover types (ponderosa pine, grand fir, etc.). A cover-type patch (stand) may have a majority of one species – if grand fir comprises more than 50% of stocking, then cover type is coded as ABGR. If less than 50% of a species is predominant, however, then a cover type is named for a species comprising a plurality of stocking – if grand fir is less than 50% of stocking but it is predominant, then cover type is coded as mix-ABGR.

I must emphasize that cover type information in table 9 does NOT reflect species composition of an individual stand or polygon. In other words, species composition of a typical dry-forest stand would not be expected to consist of 50-80% ponderosa pine, 5-20% Douglas-fir, 1-10% grand fir, and so forth – these ranges, taken from table 9, refer to percentages of a dry-forest landscape supporting ponderosa pine stands, Douglas-fir stands, grand fir stands, etc.

Table 9: RV information for species composition (vegetation cover type) for 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	NA
Western white pine	0-5
Grand fir	1-10
Whitebark pine	NA
Subalpine fir and spruce	NA

Sources/Notes: Derived from disturbance process modeling based on Vegetation Dynamics Development Tool (VDDT) (Powell 2019c). NA is Not Applicable.

Cover types reflect vegetation composition of a polygon (Eyre 1980, Shiflet 1994); they are described in Powell (2013a). 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 10: RV information for forest structural stage for dry upland forest PVG.

Forest Structural Stage	Range of Variation (Percent)
Stand initiation	15-30
Stem exclusion	10-20
Understory reinitiation	0-5
Old forest single stratum	40-65
Old forest multi strata	1-15

Sources/Notes: Derived from disturbance process modeling based on Vegetation Dynamics Development Tool (VDDT) (Powell 2019c). Forest structural stages are illustrated and described in table 8.

Table 11: RV information for tree density for dry upland forest PVG.

Tree Density Class (mixed species composition at a quadratic mean diameter of 10")	Range of Variation (Percent)
Low (<47% canopy cover; <55 ft^2/ac basal area; <103 tpa or sdi)	40-85
Moderate (47-55% canopy cover; 55-85 ft^2/ac basal area; 103-154 tpa or sdi)	15-30
High (>55% canopy cover; >85 ft^2/ac basal area; >154 tpa or sdi)	5-15

Sources/Notes: Tree density class values derived from table 7 and Powell (2013b); range of variation values derived from Schmitt and Powell (2012). Note that tpa refers to trees per acre; sdi refers to stand density index. All 'tree density class' values pertain to mixed-species, even-aged stands (a species mix of 70% ponderosa pine, 20% Douglas-fir, and 10% grand fir). Tpa and sdi values are the same because sdi uses a 10" quadratic mean diameter (QMD) as a reference tree size; if QMD had been any value other than 10", tpa and sdi values would not have been identical.

Table 12: RV information for insect and disease susceptibility for 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
Douglas-fir beetle	
Low susceptibility	35-75
Moderate susceptibility	15-30
High susceptibility	10-25
Fir engraver	
Low susceptibility	45-95
Moderate susceptibility	10-25
High susceptibility	5-10
Bark beetles in ponderosa pine	
Low susceptibility	35-75
Moderate susceptibility	15-35
High susceptibility	10-20
Mountain pine beetle in lodgepole pine	
Low susceptibility	55-90
Moderate susceptibility	5-35
High susceptibility	0-5
Douglas-fir dwarf mistletoe	
Low susceptibility	30-60
Moderate susceptibility	10-35
High susceptibility	20-35
Western larch dwarf mistletoe	
Low susceptibility	55-95
Moderate susceptibility	5-30
High susceptibility	0-5

Insect and Disease Agents[1]	Range of Variation (Percent)
Root diseases	
Low susceptibility	35-75
Moderate susceptibility	20-35
High susceptibility	5-20

Sources/Notes: Derived from Schmitt and Powell (2012). 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.

7.10 Climate Change Considerations

A pressing environmental matter of critical concern is a long-term and ongoing increase in surface temperature of the earth. This threat goes under several names – climate change and global warming are probably most common. Global warming exacerbates a natural process called the 'greenhouse effect,' referring to a principle of a greenhouse in that an enclosing shell allows passage of incoming sunlight but traps a portion of reflected infrared radiation, warming a greenhouse's interior above outside temperatures.

Greenhouse gases in earth's atmosphere play a similar role to a greenhouse's shell – they function to raise temperature of the earth and make it habitable. Without greenhouse gases, surface of the earth would be about 30 °C (54 °F) cooler than it is today, rendering human life impossible.

Since beginning of what is termed an 'industrial era' (mid 1700s), combustion of fossil fuels, together with permanent deforestation and a few other anthropogenic activities, has led to an increase in 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 middle of 21st century, depending on rates of fossil fuel burning over next few decades.
[After excluding water vapor, 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 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 particularly 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 this amount of historical change was sufficient to cause major dislocations for human agrarian societies (Mann 2006).

Climate change effects are not uniform – in the northern hemisphere, polar regions are

warming faster than equatorial zones, and centers of continental landmasses are becoming drier than their peripheries. In ice ages of the past, weather changed gradually enough to allow plants and animals to migrate and survive; rapid pace of change occurring now is likely too quick to allow many organisms to adjust to modified habitats. For this reason, some of the most concerning human impacts of climate change could involve agriculture and forestry and, of the two, forestry has fewer mitigation or adaptation options than agriculture (narrative to this point in section 7.10 is based primarily on Karl et al. 2009).

Much concern about climate change relates to how it will affect baseline climate conditions. But will climate change effects be additive, subtractive, or neutral on baseline temperature and moisture relationships, and will their magnitude be great enough to exceed environmental tolerances of existing plant species (table 13)? If an answer to the second question is yes, then one likely effect of climate change will be extirpation of certain plant species, and their related fauna and ecosystem services, from portions of the Blue Mountains (Kerns et al. 2017, 2018).

When considering precipitation patterns, it's not just potential for more and longer future droughts that is problematic (Adams et al. 2009, Hanson and Weltzin 2000, Voelker et al. 2019, Vose et al. 2016) – it's the projected change in precipitation form, with less being received as snow and more as rain (fig. 39). This trend might actually improve forest growth by lengthening the growing season into early spring, when soil moisture is at a maximum.

Because the Blue Mountains have a summer-dry, Mediterranean climate where soil-based snowmelt storage is crucial for sustaining tree growth across a relatively long growing season, a change in precipitation from snow to rain is much more likely to induce earlier summer plant dormancy, lengthen the fire season, shorten the wetland saturation period, and affect many other ecosystem goods and services (van Mantgem et al. 2009). In addition to lengthening the fire season (Hamilton et al. 2016), higher temperatures contribute to extensive fuel drying, making dry forests more flammable (Abatzoglou and Williams 2016).

Certain life history traits in table 13, such as 'tolerance to frost,' might seem unrelated to climate change. But climate change has apparently influenced the cold hardiness of trees, 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 onset of climate change, frost damage in mid-May was unusual because boreal trees had not lost cold hardiness at that point in a year.

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 1998b, Turner and Krannitz 2001), have put dry-forest ecosystems on precisely a wrong trajectory when considering the warm, fire-favoring climate expected for the 21st century (fig. 40) (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).

An important bottom-line is: "Designing more fire-resistant stands and landscapes will likely create forests that are more resistant and resilient to the changes imposed on them by climate change" (Stephens et al. 2012b).

Table 13: Selected life history traits for five primary conifers of dry forests.

	Western juniper	Ponderosa pine	Western larch	Douglas-fir	Grand fir
Tolerance to shading	L	L	L	M	H
Tolerance to full sunlight	H	H	H	M	L
Seral status	Early	Early	Early	Mid	Late
Tolerance to frost	L	L	L	L	M
Tolerance to drought	H	H	M	M	M
Rooting habit (depth)	S/M	D	D	D	S
Fire resistance	L/M	H	H	M/H	L/M
Evolutionary mode	NR	Inter	Inter	Spec	NR
Seed germination on charred or ashy soil	NR	IN	NE	IN	IN
Maximum seed dispersal distance (feet)	NR	120	150	330	200
Potential for regeneration in the open	H	H	H	H	L
Overall reproductive capacity	M	H	H	H	M
Potential initial growth rate (first 5 years)	L	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 an amount of genetic differentiation; it indicates how well a species could adapt to future climates (Inter is intermediate; Spec is specialist; NR is not rated; source = Rehfeldt 1994).

"A century of fire suppression and timber extraction has led to denser forests, with higher numbers of smaller diameter trees and larger fuel loads supporting larger, more intense fires (Hurteau and North 2010). However there is disagreement about whether these forests store more carbon now in comparison to the past – some researchers argue that because these forests have fewer mature, large trees they have lower carbon storage compared with historical levels (North and others 2009). Others suggest that current levels of carbon storage are higher than historical levels because of fire suppression (Harmon and Marks 2002; Reinhardt and Holsinger 2010)" (Ellenwood et al. 2012, p. 962).

These carbon accounting concerns are important because Collins and others (2011) found that a better approximation of historical structure and composition was produced when old forests were burned, as a restoration treatment, at moderate rather than low severity (with moderate severity ostensibly reducing carbon storage to a greater extent than low severity). This result occurred because higher fire intensity was needed to kill a sufficient number of intermediate-sized trees resulting from decades of fire exclusion. ["we now have millions of acres where 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" (page 45).]

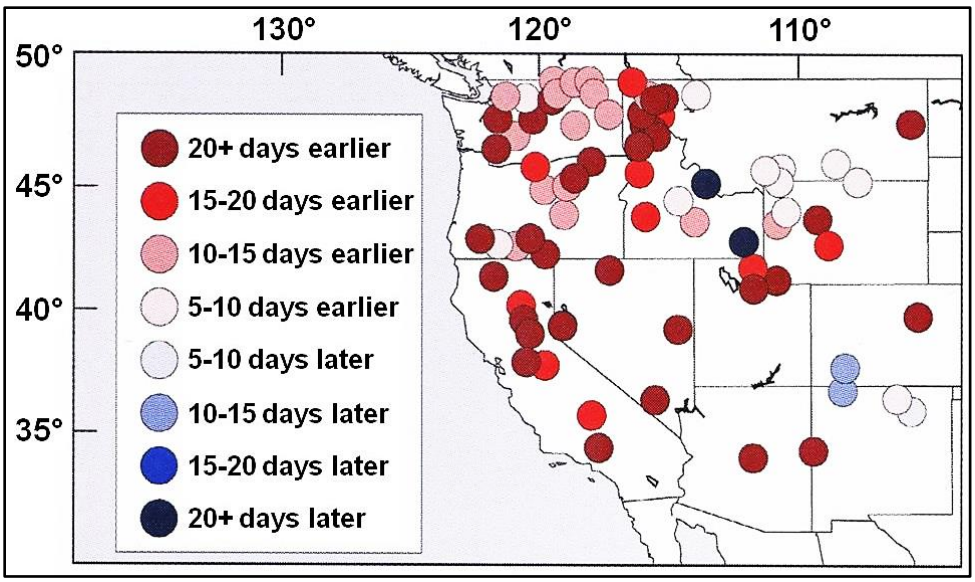


Figure 39 – Recent changes in spring snowmelt timing for western United States (from Karl et al. 2009, p. 33). This chart shows trends in streamflow runoff timing for 1948-2000, as a number of days runoff occurs earlier. According to this analysis, northern Blue Mountains river basins occur in a zone where runoff occurred 10-20+ days earlier for a 1948-2000 period than it did previously. Future climate change is expected to continue and exacerbate this trend (Furniss et al. 2010, Stewart et al. 2004).

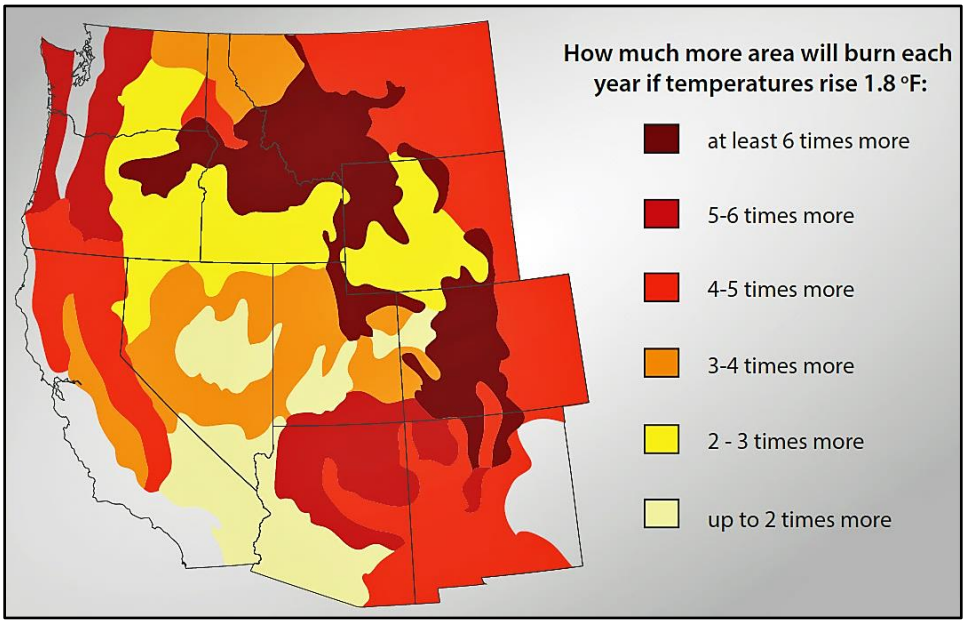


Figure 40 – Predicted increase in area burned by wildfire as associated with a mean annual temperature increase of 1 °C (1.8 °F), shown as percentage change relative to median annual area burned during 1950-2003 (source: Climate Central 2012). Results are aggregated to ecoprovinces (Bailey 1995) of western US. Climate-fire models were derived from National Climatic Data Center records and observed burned-area data following methods described in Littell et al. (2009). Prediction shown here is similar to several reports from National Research Council showing at least a quadrupling of area burned in western US with each 1 °C (1.8 °F) of temperature increase (this figure adapted from figure 5.8 in National Research Council 2011).

Predicted increases displayed in figure 40 are alarming because when examining a century-long period from 1970-99 to 2070-99, increases in average annual temperature of 3.3 to 9.7 °F are projected, depending largely on whether global emissions eventually decline (B1 greenhouse gas emissions scenario) or continue to rise (A1B, A2 emission scenarios), and temperature increases are projected to be largest in summer when they would coincide with fire season.

If dry mixed-conifer forests are to have a reasonable opportunity for persistence under future climate regimes, restoring conditions more similar to historical characteristics of frequently burned, open forests of the past is likely to function as a useful start point (Fiedler 2000b, Harrod et al. 1999, Munger 1917). Although debate continues about how, where, and to what extent reference conditions derived from historical ecology should be used for land management (Millar and Woolfenden 1999), consensus is growing that it is useful to know and understand the past to properly manage future ecosystems (Swetnam et al. 1999).

Sustainable, dry-forest conditions can be achieved by reintroducing surface fire, and by implementing thinning treatments, to change fire-free intervals from centuries to decades, to reduce surface fuels, and to reduce canopy and ladder fuels to counteract a compositional trend toward increased representation of fire-sensitive trees (fig. 41). These treatments could help reestablish spatial heterogeneity (fig. 42). "A lack of treatment or passive management (Stephens and Ruth 2005) perpetuates the potential for extensive high fire severity in forests that once burned frequently with low- to moderate-intensity fire regimes" (Stephens et al. 2012b).

"Thinning is thought to reduce the risk of stand-replacing fires and the corresponding sudden release of large amounts of carbon to the atmosphere (Hurteau and others 2008; Dore and others 2010; Reinhardt and Holsinger 2010). However, the effects of thinning and fuels treatment on overall carbon balance are complex. The ultimate effect of thinning and fuels treatment on carbon stocks is affected by the initial state of the forest, the types of treatments conducted (e.g., mechanical thinning versus prescribed burning), and the time period over which one compares the carbon balance (North and others 2009; Hurteau and North 2010). Some researchers suggest that while thinning and other mechanisms to reduce fire risk reduce the overall carbon stocks in the forest by a moderate amount in the short run, if the treated forest subsequently supports the growth of larger mature trees it may end up storing as much carbon as it did before treatment, in a landscape that is less susceptible to large stand-replacing fires (North and others 2009; Reinhardt and Holsinger 2010)."

"Because the carbon balance continues to change as the forest recovers long after the fire event, there is much uncertainty about how long it might take for the carbon balance to be back in equilibrium (Dore and others 2010; North and others 2009; Kashian and others 2006). In addition, the fate of thinned material influences the overall carbon impact, e.g., by sequestering carbon if the material goes into long-lived products such as timber or displacing fossil fuel emissions if used to produce biomass energy (Harmon and Marks 2002)" (quoted material in two paragraphs above from Ellenwood et al. 2012, p. 962).

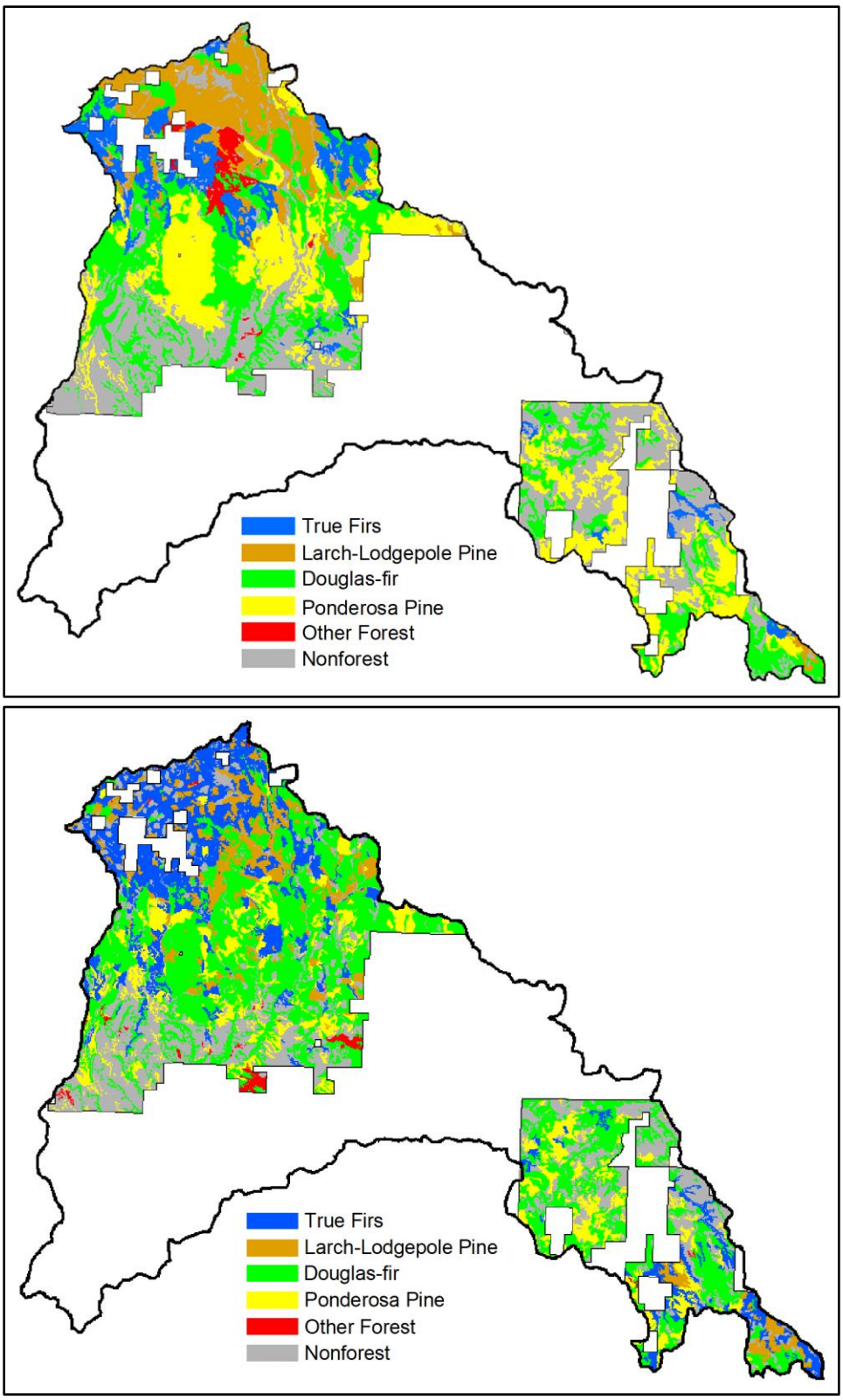


Figure 41 – Historical (upper; 1939 conditions) and existing (lower) vegetation cover types for Potamus watershed, Umatilla NF. When comparing these two maps, ponderosa pine type declined, and Douglas-fir and true-firs types increased.

Figure 42 – Reintroduction of spatial heterogeneity into Wild Horse prescribed fire area. Blue Mountains experience over past 20 years suggests that adopting a conservative approach to restoration of dry forests is not a choice of 'no action' because a passive strategy accepts risk of high-severity wildfire and other uncharacteristic disturbance (see fig. 19). Upon recognizing that 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 (Agee and Skinner 2005; Allen et al. 2002; Hessburg et al. 2000, 2007; Wright and Agee 2004) (and, see figs. 44-45 later in this paper).

Restoration prescriptions that lack prescribed fire are incomplete because thinning alone is not sufficient to renew nutrient cycling processes for dry sites. And, if scale of management activity does not emulate 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). And, scale includes temporal considerations as well – seasonality of historical fires, for example, follows a latitudinal gradient for the Blue Mountains, with predominantly early-season fires in southern Blues and late-season fires in northern Blues (Heyerdahl et al. 2001).

Forest trees in an upper canopy layer have better access to sunlight, nutrients, and moisture than trees in subordinate positions. Since dominant trees use a disproportionate share of site resources, it seems logical 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 is a reciprocal process; removing subordinate trees, by using a low thinning to remove many intermediate and subcanopy/suppressed trees, results in dramatic vigor increases for dominant ponderosa pines (Woodall et al. 2003), particularly for 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 Dry Upland Forest (UF) PVG might increase at

expense of Moist UF or Cold UF PVGs is speculative at this point, several climate change scenarios examined for the interior Pacific Northwest suggest that this is a likely outcome (Dello and Mote 2010, Kerns et al. 2017, 2018).

There is also no assurance that current amounts 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 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 circumstances under which it has occurred elsewhere, a savanna outcome is certainly plausible for some proportion of dry-forest acreage located within a Blue Mountains ecoregion (Kerns et al. 2018).

Many policy proposals being considered to address climate change are based on mitigation – reducing greenhouse gas emissions from fossil fuels and land-use changes to minimize pace and magnitude of climate change. While mitigation is important, adaptation to climate change is increasingly viewed as a necessary and complementary strategy to mitigation (Joyce et al. 2009). Table 14 provides adaptation strategies proposed for National Forest System lands, and pertaining to upland forest vegetation. Table 14 also describes predicted compatibility of active management treatments with climate change adaptation strategies.
Note: some sources frame a mitigation/adaptation couplet as resistance/resilience – near-term resistance measures, such as thinning, need to be fully coordinated with far-term resilience strategies (e.g., creating climate-adapted genotypes by establishing new tree regeneration).

Information in table 14 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).

Potential for uncharacteristically severe wildfire is particularly high – of 47 million acres of federal land in the Pacific Northwest, approximately 47 percent (22.6 million acres) was historically affected by short interval fire (these are dry sites once dominated by ponderosa pine, shrubs, or bunchgrasses). A majority of these lands are located east of 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 existed historically (fig. 43).

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 13 for five primary tree 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 would improve adaptive capacity (Olsson et al. 2004) of dry mixed-conifer forests in the Blue Mountains, particularly by alleviating chronic stress associated with high tree density levels, and by reestablishing an historically appropriate structural condition (fig. 44). A recently developed option for reestablishing a groupy or clumpy condition in dry forest is to apply an ICO approach (Churchill et al. 2013a, 2013b; Franklin et al. 2013) (fig. 45).

Table 14: Estimated compatibility of climate change adaptation strategies and active management of dry upland forests.

Climate Change Adaptation Strategies	Compatibility With Dry-Forest Management
Improve ecosystem capability to withstand uncharacteristically severe drought, wildfire, and insect infestation at landscape scales.	Thinning and similar active management practices might be necessary to improve 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 13).
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 compatible with 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 13).
Use historical ecological information to identify environments buffered against climate change, and that 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 uncertainty about which tree species will prosper in the future.	Small-diameter trees will be removed frequently 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 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 13) 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 plant materials.	This recommendation is similar to 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. Rapid response to mortality helps address increased fire and insect risk related to climate change.
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 future climate warms and dries.

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

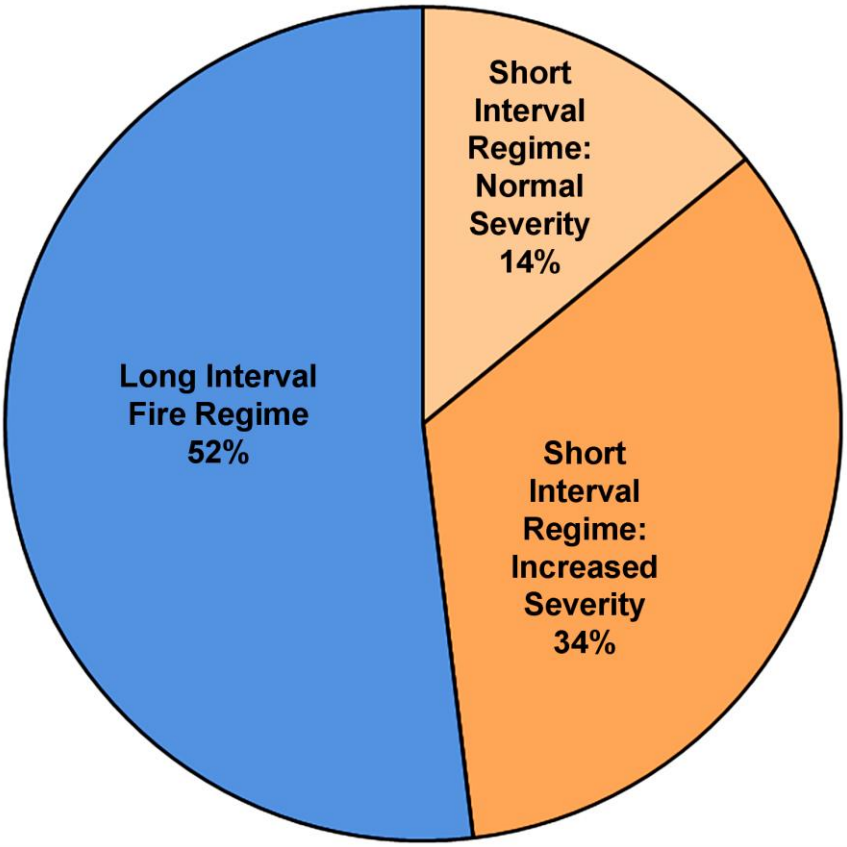


Figure 43 – Trend toward increasing fire susceptibility for short-interval fire regimes of Pacific Northwest.[1] Many sources show the scale of predicted change in fire severity for fire regime 1 sites to be enormous (Hessburg et al. 2005, Hann et al. 1997, Quigley et al. 1996). When considering 47 million acres of federal lands in Pacific Northwest administered by Bureau of Indian Affairs, Bureau of Land Management, National Park Service, U.S. Fish and Wildlife Service, and U.S. Forest Service, about 48% can be assigned to a short-interval fire regime (fire regimes 1 and 2). Of this dry-site acreage, 71% (16 million acres, or 34% of total acreage) currently has a higher predicted fire severity (stand-replacing) than would have existed historically (stand-maintaining). When General Accounting Office evaluated catastrophic fire risk for 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).

Many dry-forest studies suggest that a large proportion of remaining 200+ year-old ponderosa pines shown in figures 44-45 are likely to die in next few decades, particularly as a result of western pine beetle attack (fig. 36), unless major restoration treatments are completed (Lynch et al. 2000). Many of these veteran trees could survive for another century or more if dry-forest composition, structure, and density is returned to historical conditions (fig. 46).

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

Figure 44 – Many dry forests still have remnant historical structure such as this clump of mature ponderosa pine. Historically, dry forests tended to be uneven-aged at a stand level, with stands consisting of small even-aged tree clumps, each differing in age from others (Cooper 1960, 1961a; Munger 1917; White 1985). Historical clump size 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). So for dry forests, patch size (e.g., an opening large enough to sustain regeneration) is quite small – 0.6 acres or less (Agee 1998). Restoration treatments can be used to remove late-seral species (Douglas-fir and grand fir) from this clump, improving its resistance and resilience to climate change.

Figure 45 – Tree clumps in ponderosa pine forest. When evaluating spatial heterogeneity for dry forests, it is important to consider a sub-stand level because scale is fine-grained and intricate for dry forests – ponderosa pine stands historically featured a groupy or clumpy structure at a sub-stand scale (Harrod et al. 1999). What are sources of these clumps? As described in section 4, variable thinning effects caused by frequent surface fire was certainly an important factor. Another factor relates to seed caching by deer mice, golden-mantled ground squirrels, chipmunks, shrews, and other small mammals (Keyes et al. 2007; Saigo 1969; Vander Wall 2002, 2003) because unutilized seed caches also result in tree clusters. Pine clumps can contain few (above) or many trees (below); an intertree distance of 6 meters (app. 20 feet) is sometimes used to identify clump 'membership' (trees < 6m apart are in a clump, whereas trees > 6m apart are out). A recent approach for reestablishing spatial heterogeneity in dry forests is an ICO (Individuals, Clumps, Openings) methodology (Churchill et al. 2013a, 2013b; Franklin et al. 2013).

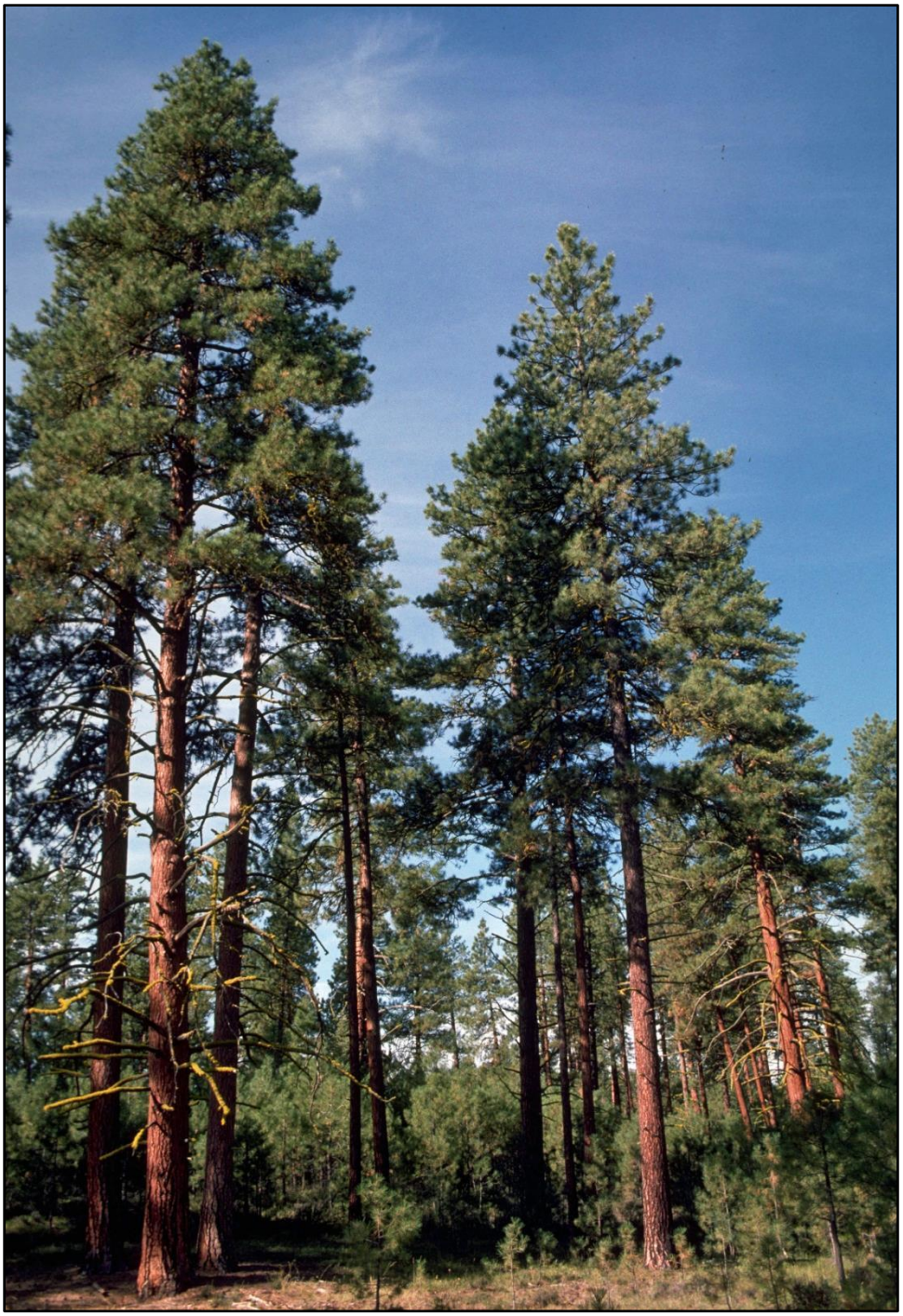


Figure 46 – Clumps of old ponderosa pines surrounded by ponderosa pine regeneration, southern Blue Mountains (Malheur National Forest). Other images in this white paper show old pines surrounded by regeneration of mid- or late-successional trees such as Douglas-fir or grand fir (see figs. 27 and 44). However, dense regeneration of ponderosa pine intermixed with, or adjacent to, old ponderosa pines is problematic, even though the regenerating species is ecologically appropriate for dry sites. Completing an 'understory removal' treatment to remove pine regeneration, within at least 2 driplines of mature trees, is beneficial for reestablishing an historically appropriate stand structure, and it responds to our long-term suppression of nature's thinning agent – frequent, recurring surface fire.

A common goal of dry-forest restoration is to develop more open structures consistent with historical disturbance regimes (Arno et al. 1995), an outcome also considered to be compatible with a warmer and dryer future (Brown 2008, Stephens et al. 2013). This goal agrees with studies reconstructing composition and structure for dry sites because they report much lower stem densities than today, larger trees, and a strong clumping pattern for overstory trees (see table 2; Churchill et al. 2017; Harrod et al. 1999; Johnston et al. 2016, 2017, 2018).

Climatic drought is projected to be more common in the future (Hanson and Weltzin 2000) because mid-summer temperatures are expected to be higher, and summer precipitation lower, than at present. But, dense stands exist in a sort of perpetual 'physiological' drought because intertree competition results in a situation where not enough soil moisture is available to meet water needs of all trees (regardless of rainfall amounts); silvicultural treatments are used to alleviate this moisture stress and allow residual trees to survive and continue growing.

Since climate change will amplify effects of density-caused stress, the need for future thinning is expected to be much greater than at present, particularly because thinning improves physiological vigor, and trees with improved vigor produce more 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).

Direct effects of climate change on temperature and precipitation, in conjunction with indirect effects from wildfires, insect outbreaks, and other disturbances that will continue ramping up as climate changes, could detrimentally affect future provision of ecosystem goods and services (Krieger 2001), including old forest, properly functioning soil and water services, wildlife habitat, animal and plant diversity, recreational opportunities, and carbon storage (fig. 47).

Climate modeling suggests that western larch could be extirpated from Blue Mountains by early 22nd century (Rehfeldt et al. 2006). And although studies were not specific to the Blue Mountains, recent developments across western U.S. suggest that quaking aspen is also quite sensitive to climate change (Rehfeldt et al. 2009, Rogers et al. 2007, Worrall et al. 2010).

Although aspen is generally perceived as being associated with moist or mesic conditions (fig. 47), aspen is also a dry-site species – a Dry Upland Forest potential vegetation group includes 8 potential vegetation types dominated by aspen (see appendix 1).

Effective resistance treatments must consider the landscape context in which they occur – Finney and others (2007) compared effectiveness of different rates of fuels treatment over several decades for western U.S., and they found that treatment rates beyond 2% of a landscape per year, based on optimized treatment placement (such as 'strategic placement of landscape area treatments' or SPLATs), yielded little additional benefit (Stephens et al. 2012b).

Treatment timing is also important. Although cut-burn treatment combinations are consistently effective at restoring dry forests, it has also been found that burning too soon after thinning can result in delayed tree mortality (Fajardo et al. 2007). [A consistent management implication from many dry-forest restoration studies is that direct reduction of overstory density (thinning), in combination with renewal of nutrient cycling mechanisms (burning), yields the highest increases in undergrowth plant and overstory tree vigor.]

Figure 47 – Fenced clone of quaking aspen located along 5316 road on North Fork John Day Ranger District (aspen is short and has yellow foliage). Aspen reproduces almost exclusively from root suckers, resulting in a clonal life history where its root system functions as a genet, producing successive generations of suckers called ramets. Ramets develop into mature trees. Although aspen ramets are relatively short-lived (60 to 100 years is common), an underground genet may be thousands of years old. Some clones in the intermountain West 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. Genetic testing indicates that an ancient aspen clone has existed for thousands of years in Morsay Creek drainage, approximately 15 miles west of Ukiah, Oregon (Shirley and Erickson 2001).

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

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

Aspen has a surprising affinity for dry-forest environments (e.g., it occupies moist microsites within a broader warm dry biophysical setting), as illustrated by a recent classification of quaking aspen types – of 15 aspen plant community types identified for the Blue Mountains, 8 of them occur in a Dry Upland Forest potential vegetation group (see appendix 1; Swanson et al. 2010).

It has been suggested that increasing the time interval between thinning and burning treatments could reduce amounts of delayed mortality by allowing more time for residual trees to increase their post-thinning vigor, and by allowing thinning-created surface fuel to decompose before completing a prescribed fire (fig. 48).

Another potential benefit of increasing time interval between thinning and burning is that it lengthens the total period for which treated areas have reduced fire hazard (i.e., burning quickly after thinning will result in an area returning to pretreatment fuel hazard levels more quickly than by waiting a little longer between treatments) (Stephens et al. 2012b). And, as climate continues to warm and dry, treating dry-forest areas to reduce fire hazard allows them to be maintained in a low-hazard state by using a 'managed wildfire' approach (North et al. 2012), which is an increasingly attractive option as wildfire acreage (fig. 40) and fire suppression costs continue to increase (Haughian et al. 2012, Littell et al. 2009).

Obviously, changes of the magnitude described in this section would cause 'ripple effects' across many biological webs and trophic levels (Perry et al. 2008). If climate change precludes us from sustaining desired levels of ecosystem composition, structure, and density, then how can we sustain the ecosystem goods and services contingent on these components? I believe the best answer to this question, as hopefully demonstrated by this white paper, is to apply correct dry-forest restoration practices, in correct places, at proper times, and for correct reasons.

SUMMARY: ESSENTIAL TENETS OF DRY-FOREST MANAGEMENT

This white paper discusses how three primary human influences affected dry forests – fire exclusion, livestock grazing, and selective cutting. Other factors also contributed to changes:

- Dry forests often have low ecological integrity, particularly in response to suppression of surface fire, a keystone ecosystem process for this biophysical environment.
- Low-severity surface fire has now been replaced, in many areas, by high-severity crown fire.
- Forests comprised primarily of late-seral structural stages have declined, especially for an old forest single stratum stage dominated by large-diameter ponderosa pines.
- Mid-seral structures have increased, and this change contributed to landscapes that are much more homogeneous than they were historically.
- Contemporary dry forest has markedly higher tree density and understory fuel loading, while simultaneously exhibiting much lower undergrowth productivity and diversity.

- Existing stand density is often inconsistent with an historical disturbance regime – uncharacteristically high density levels are common across dry landscapes of the federal estate.
- Forest canopies became more complex and layered due to loss of thinning agents; resulting multi-layered structure is now ladder fuel and provides a budworm feeding ladder.
- Wildfires and defoliating insects (mainly western spruce budworm and Douglas-fir tussock moth) now occur with uncharacteristic, stand-replacing severity on many dry landscapes.
- Largely in response to fire exclusion and livestock grazing, dry forest expanded onto sites historically supporting woodland, shrubland, or grassland vegetation.
- High livestock grazing levels in early 1900s affected tree regeneration by reducing herbaceous competition with seedlings, and by exposing mineral soil for tree-seed germination.
- Historical disturbance regimes provide a blueprint for active management to maintain ecological function; treatments must emulate native disturbance and succession processes.
- Stand density changes led to a decline in individual-tree vigor, increased probability of bark beetle outbreaks, and contributed to a higher probability of uncharacteristic wildfire. Stand density changes also influence rates and types of surface fuel accumulation.
- Thinning and prescribed fire, applied in proper places and at appropriate times, is needed to help recover integrity and resilience of dry-forest ecosystems.
- Thinning and prescribed fire are effective for sustaining high vigor levels for old ponderosa pines; high vigor translates into increased resin production and chemical defenses against western pine beetle and other insects and diseases.
- Thinning to low stocking levels will be especially important in a climate-changed future because wildfire may convert dry forests to shrublands or herblands, whereas trees in thinned areas would survive and promote a fire-compatible savanna structure.
- Going back to early 1990s, fire scientists recommended that prescribed fire use be increased tenfold for Blue Mountains national forests to address forest health concerns; unfortunately, and for many and varied reasons, an increase in prescribed-fire use did not occur.
- To support increased utilization of prescribed fire, dry forest should be managed to sustain coarse woody debris (CWD) levels ranging between 5 and 20 tons per acre (defined as dead standing and downed pieces larger than 3 inches in diameter). Between 4 and 7 tons per acre of a 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 snags per acre should be at least 15" in diameter).
- A lack of low-severity disturbance means that composition (including nonforest) and structure is less diverse now than historically, but more tree species may now be found on dry sites because ponderosa pine has been joined by Douglas-fir, grand fir, and western juniper.
- Historical timber harvest removed many large, fire-resistant ponderosa pines. Future harvest could use uneven-aged management to restore intra-stand heterogeneity (fig. 49).
- Small trees got established abundantly (appendix 3), and many of them are fire-sensitive species (Douglas-fir, grand fir). Small trees act as ladder fuel during wildfires, causing individual tree clumps to torch, or contributing to stand-level crown-fire behavior.
- A dynamic ecosystems philosophy should form a foundation for dry-forest management – *an ecologically sustainable representation of composition, structure, and density is more important than perpetuation of a particular condition in a specific location*.

Figure 48 – Shelterwood seed cut in dry upland forest on Manitou Experimental Forest, Pike National Forest, southern Front Range (Rampart Range), south-central Colorado.

An important objective of dry-forest restoration programs is to reestablish a stand structure similar to an old forest single stratum structural stage (see table 8), which predominated during historical (presettlement) eras (see figs. 34, 44, 45).

Table 5 summarizes these restoration concepts by noting that sustainable dry-forest ecosystems (characterized as an 'ecosystem maintenance stage') would classify as Condition Class 1 when evaluated by using a Fire Regime Condition Class framework.

Table 5 describes composition and structure, tree density, vigor, fire regime, fuel dynamics, and resilience and risk characteristics associated with condition class 1 dry forests.

Although this image shows a regeneration cutting method (shelterwood seed cut), it is illustrative because it shows what a post-treatment, tree-structure outcome might look like for restoration treatments designed to recreate an historical stand structure by emphasizing retention of larger-diameter, older-age ponderosa pine trees.

And, note that this treatment obviously addressed ladder-fuels, but a paucity of smaller trees results in an unbalanced size-class distribution, which may not bode well for the future (especially if bark beetles, pine butterfly, or another disturbance agent affects a high proportion of the overstory-tree cohort).

Note, however, that this image is not necessarily a good example of a comprehensive restoration treatment. Why is this statement true? Because, there is an obvious lack of dead wood, both as down logs and as standing dead trees (snags). A lack of dead wood, and an extreme reduction in surface fuels, was completed for research purposes, not in response to restoration goals and objectives.

Reductions in surface fuel can be important, however, for ensuring overstory-tree survival during subsequent prescribed fire treatments (Fajardo et al. 2007, Hood 2010, Stephens et al. 2012b).

Another example of dry-forest restoration, with an emphasis on restoration of historically appropriate 'old forest single stratum' structure, is the cover image used for this white paper. It shows post-treatment conditions for a restoration project completed in Swan Valley of western Montana.

Note that the Swan Valley image still features an impressive reduction in surface and ladder fuels, but that a minimal component of dead wood was retained in the form of down logs and stem wood pieces containing attached branches.

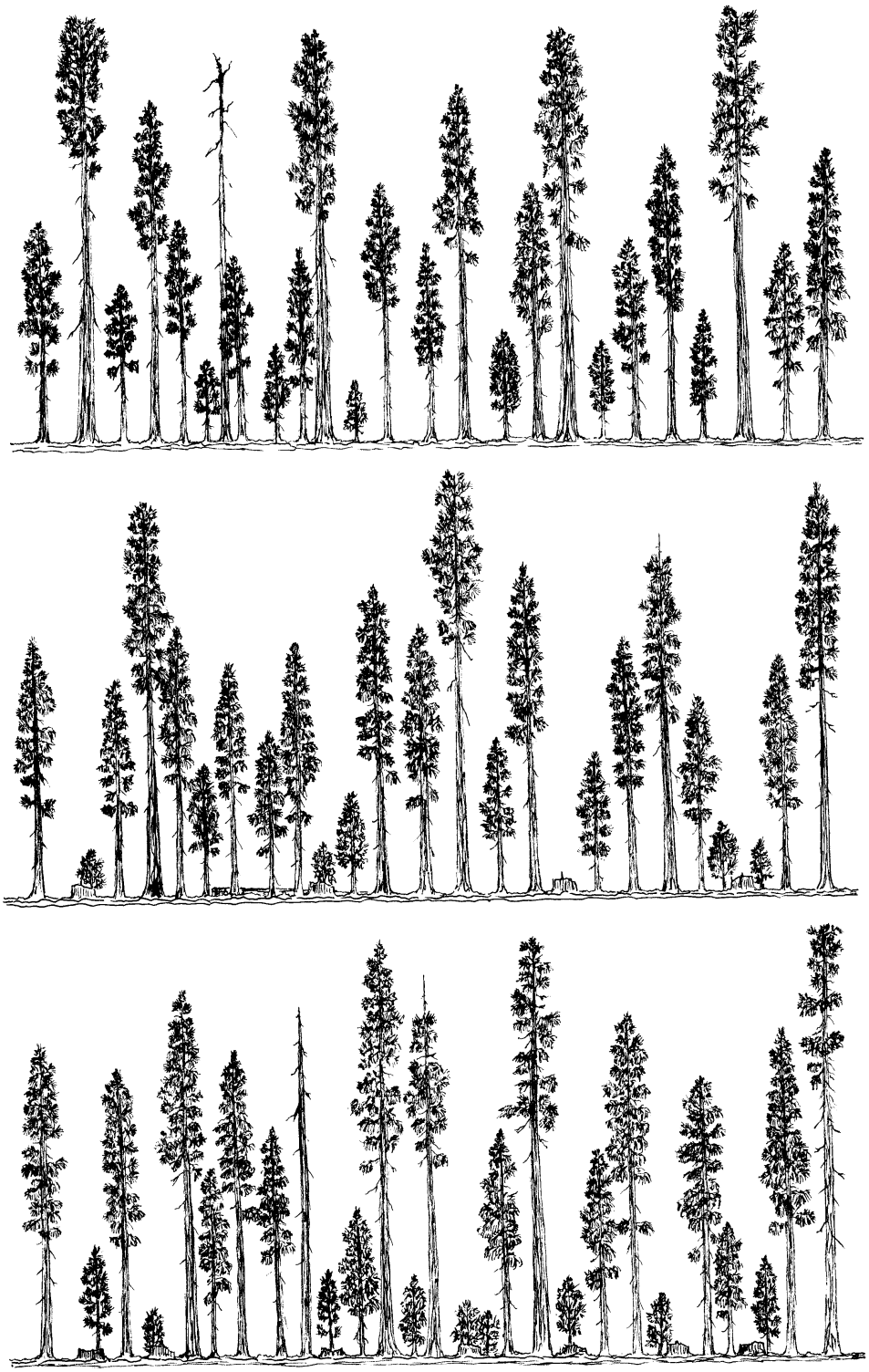


Figure 49 – Application of individual-tree selection in a ponderosa pine forest. An untreated stand (top) has a range of tree sizes. In the first entry (middle), note how four mature trees were removed. A second entry (bottom) continues this cutting intensity. In many respects, *uneven-aged management is ideally suited as a silvicultural system for perpetuating and sustaining dry-forest ecosystems*, while also ensuring that heterogeneous stand structures are provided through time (Franklin et al. 2013).

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Don Justice, data analyst with Natural Resources group in Umatilla NF Supervisor's Office, compiled maps and data used for figure 2 and table 1. An initial version of this white paper benefited greatly from technical reviews provided by Rick Brown (Defenders of Wildlife), Craig Schmitt (USDA Forest Service, Blue Mountains Service Center, La Grande), and Carrie Spradlin (USDA Forest Service, Umatilla National Forest, Heppner Ranger District).

IMAGE CREDITS

Unless noted otherwise in a figure caption, color photographs used in this white paper were acquired by David C. Powell. A white-headed woodpecker photograph used in Box 1 was acquired by Al and Elaine Wilson (www.BirdsISaw.com). A panoramic image on page 104 shows a mixed-conifer forest with complex forest structure created by frequent use of prescribed fire in the Illilouette Basin, Yosemite National Park (image taken from North 2012, page 166). Historical, black-and-white photographs are acknowledged in figure captions. Unless noted otherwise, non-photographic charts, diagrams, or graphics were prepared by David C. Powell.

APPENDIX 1: POTENTIAL VEGETATION COMPOSITION

A dry upland forest PVG includes dry mixed-conifer forests occurring in a lower montane vegetation zone (see fig. 3). Portions of three potential vegetation series (see fig. 4) are represented in a dry upland forest PVG – grand fir, ponderosa pine, and Douglas-fir. Note that quaking aspen plant community types, 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 a dry upland forest PVG (two 'sodgrass' types: elk sedge and pinegrass associations, and a birchleaf spiraea type), but they occupy substantial acreage in 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 (snowberry, birchleaf spiraea, and ninebark associations).

Although ponderosa pine is ubiquitous in the Blue Mountains, it is the climax species on a surprisingly small percentage of this area (certainly less than 10% for northern Blue Mountains, but a higher percentage than that for 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 environmental tolerances of ponderosa pine do not allow it to predominate on cold or moist forest sites.

Table 15: Potential vegetation type (PVT) codes and names, and plant association group (PAG) assignments, for a 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
ABGR/SPBE	grand fir/birchleaf spiraea	warm dry

PVT Code	PVT Name	PAG
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, are taken from Powell et al. (2007) except for aspen community types, which are from Swanson et al. (2010).

APPENDIX 2: EARLY 1940S TIMBER HARVEST

This white paper has a section describing selective cutting effects on dry-forest conditions. Light selective cutting occurred in the Blue Mountains as early as late 1800s, and most early harvests involved relatively low timber volumes distributed across fairly small areas. Early mills were small and could easily be moved to a new site (portable circular saws); their locations changed frequently as available timber was depleted. Many early mills were located along streams because they depended on water power to run a circular saw.

By 1920s, high-volume sales covering large areas were awarded. Oregon Lumber Company built a sawmill at Bates (a town-site located near Oregon Highway 7 about 15 miles northeast of Prairie City) after it was awarded a 124 million board foot timber sale in Middle Fork of the John Day River drainage (USDA Forest Service 1916). Another example is a Camas Creek sale containing 221.3 million board feet of national forest timber on Umatilla National Forest. It covered an area of about 69,645 acres in the Camas and Meadow Creek watersheds (North Fork John Day Ranger District); western boundary of this unit was located app. 10 miles east of Ukiah, Oregon.

Timber on the Camas unit consisted of a mixed stand of ponderosa pine, Douglas-fir, western larch, lodgepole pine, and other species (Matz 1932). All advertised volume was ponderosa pine; cutting of other species was optional at discretion of a purchaser (Stevenson 1937). Calculations in a timber appraisal showed that sustained yield for Camas Creek block was 13,780,000 board feet per year, so the Camas Creek timber sale represented an 'over-cut' of 71,100,000 board feet for a five-year period (USDA Forest Service 1938).

Blue Mountain harvest levels escalated in 1928 when Edward Hines Lumber Company was awarded a long-term contract for 890 million board feet in Seneca area. This enormous timber sale, called the Bear Valley Unit, is a good example of the concept that timber was not only a commodity, but also a federal government tool for community development (Fedkiw 1999). As of late 1920s, Bear Valley timber sale was the largest ever offered in Pacific Northwest, and it was probably more widely advertised throughout the United States than any other sale of national forest timber up to that point in time.

Bear Valley timber sale was designed to extend Oregon Short Line Railroad from Crane to Burns (30 miles of standard-gauge track), extend the railroad farther by running lines from Burns to Seneca (50 miles), and then develop short branch lines from Seneca into timber sale areas in headwaters of Silvies River. Railroad work associated with Bear Valley sale had an influence on the broader Blue Mountains area. Construction of Oregon Short Line connecting Oregon-Washington Railroad and Navigation Company with Union Pacific Railroad allowed eastern Oregon to enter national lumber markets for the first time. Prior to this railroad development, all pine lumber produced in the Blue Mountains was used for local or regional consumption.

Photographs in this appendix were taken on Bear Valley Unit as it was being operated by Hines Lumber Company. Photographer was Russell Lee, who worked for the federal government as an employee of Farm Security Administration. The photographs were taken from this website: http://photogrammar.yale.edu/

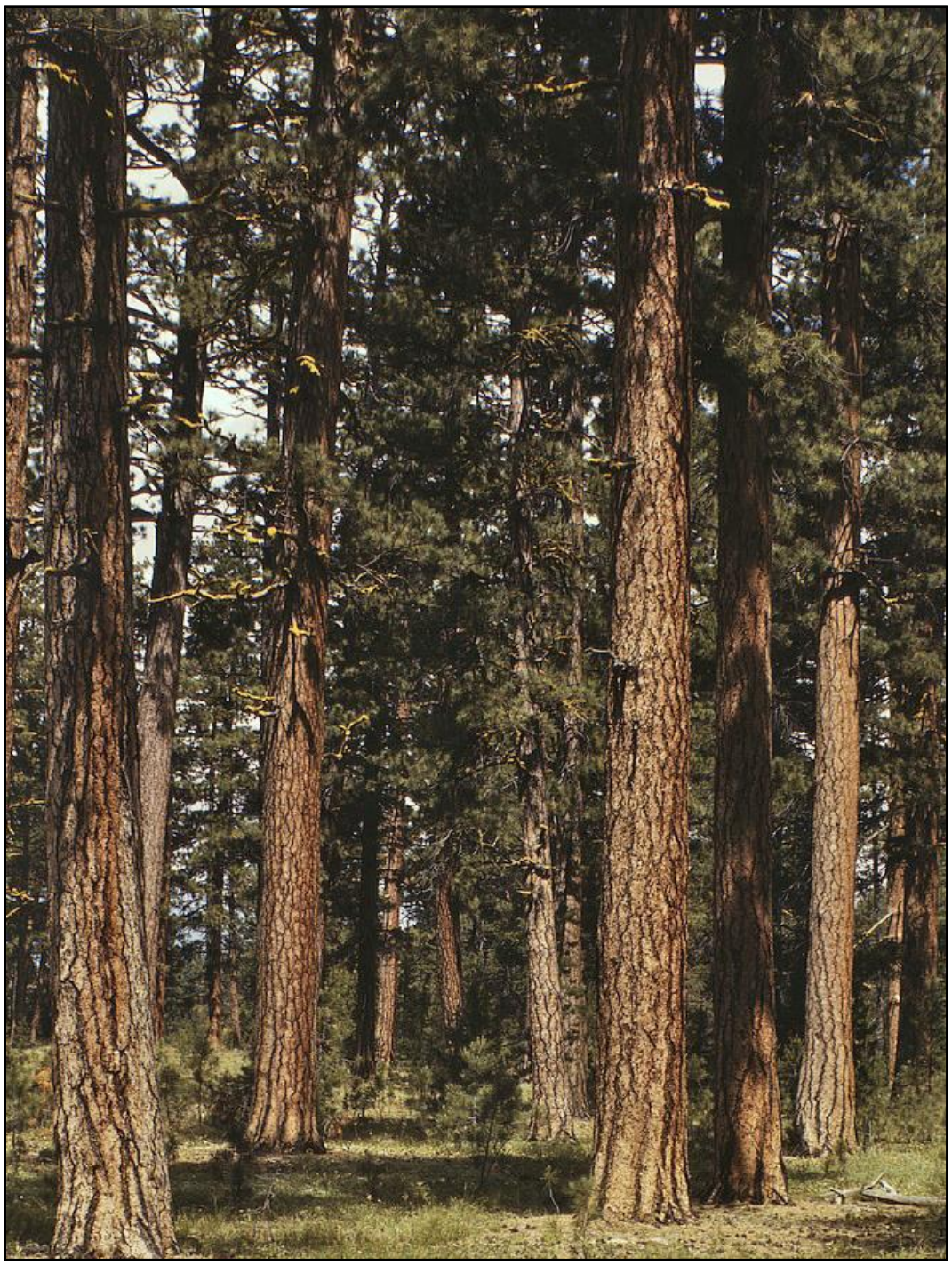

Figure 50 – Virgin (unmanaged) ponderosa pine forest in Grant County, Oregon (photo by Russell Lee, July 1942). [This image depicts better-than-average conditions in terms of stocking levels; it is known that photographers generally selected the best examples.] This image shows a mature stand of ponderosa pine, with closely-spaced groups or clumps of large-diameter trees. Note that spindly, suppressed ponderosa pine seedlings are also present as an understory. A paradigm of this era was to remove mature and overmature ponderosa pines before they were attacked by western pine beetle. Maturity selection methods were used to evaluate tree vigor and insect susceptibility, including methods developed by F.P. Keen (1936, 1950), and thereby identify old pines with high susceptibility to western pine beetle attack.

Figure 51 – Forest Service official designating (marking) a tree for removal in Grant County, Oregon (photo by Russell Lee, July 1942). [Note a soft, fedora-style 'hardhat' worn by this Forest Service official.] Even though a high-volume timber sale occurring across a wide area had been awarded to Edward Hines Lumber Company, Forest Service still needed to designate trees for removal before they could be felled. In this era and for these large sales, it was common to have a Forest Service official assigned to a sale operation full-time, and he might mark trees in the morning that were felled later the same day by a purchaser's timber fallers. This strategy differs from contemporary practice because Forest Service now completes timber designation activities before awarding a timber sale to a successful purchaser.

Here is a description of how early marking occurred: "The mechanical part of marking consists simply of striking a clean-surfaced bark blaze at breast or shoulder height with a keen-bladed special hatchet and stamping on this with the head of the hatchet the letters 'U.S.' This operation is repeated on the base of the tree below stump height. Whether a man is fast at this job, which may be a big and time-consuming job on a large sale, depends upon several things. He will be slow unless his judgment is such as to enable him to weigh all the factors and make his decision as fast as he can get to the trees, if he is either lazy or physically unfit for hard work, or if he does not avoid lost motion in getting around to his timber" (Perry 1999, p. 97). Walt Perry went on to describe how he was able to mark more than 1,000 trees in an 8-hour workday by using this method, and he bragged that on one day, he marked 1,685 trees in only 7 hours and 50 minutes (suggesting that on some days, at least, quantity was placed at a higher premium than quality when it came to tree marking).

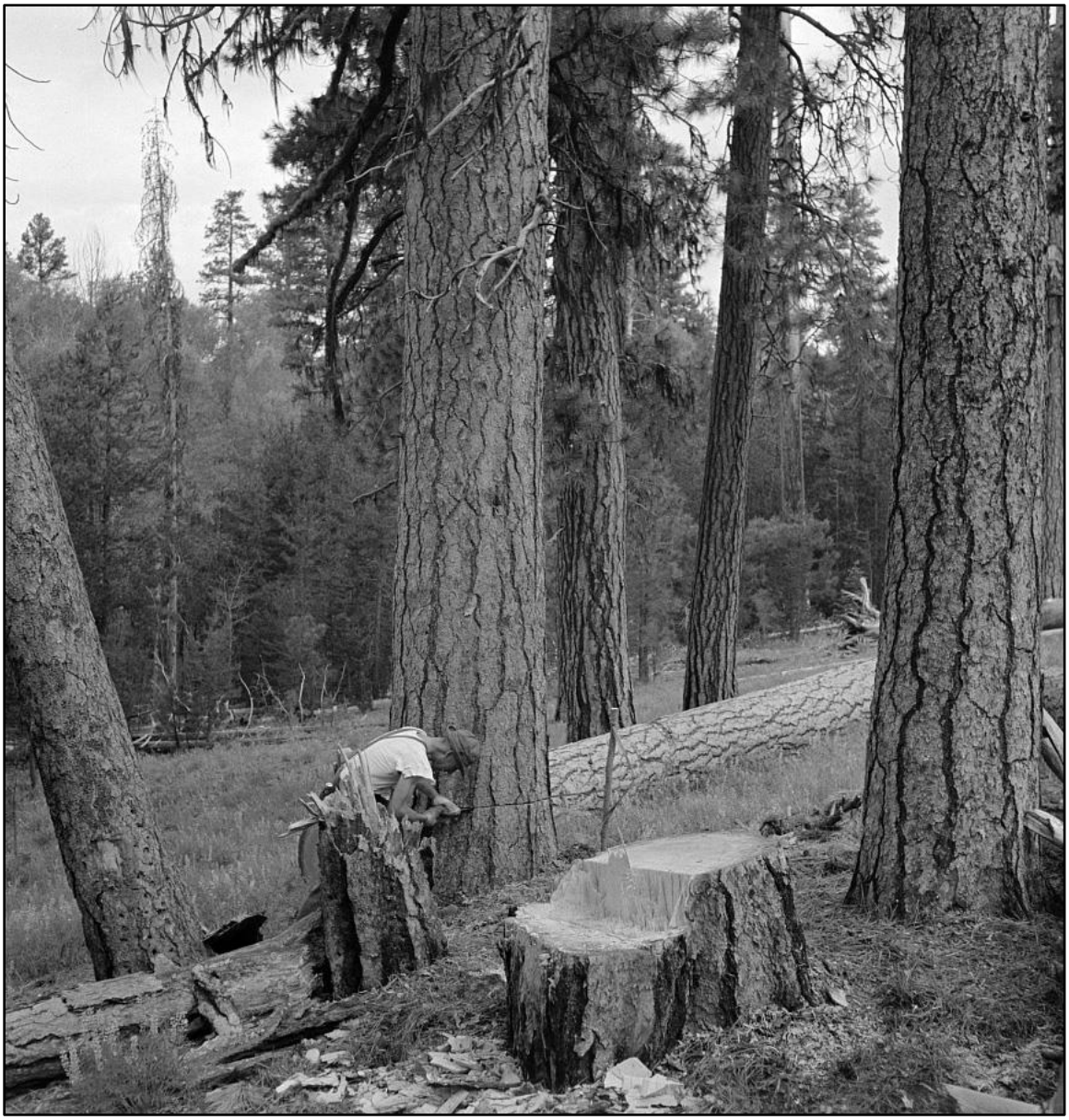

Figure 52 – Falling a large ponderosa pine with a crosscut saw in Grant County, Oregon (photo by Russell Lee, July 1942). When these photographs were taken, all tree falling operations were accomplished by using crosscut saws, commonly referred to as 'misery whips' because of the human labor and effort involved in their use. Optimum productivity was obtained by having fallers work in pairs so they could operate a crosscut saw in tandem.

Figure 53 – Falling a large ponderosa pine by using a crosscut saw and a 'rubber-man' setup in Grant County, Oregon (photo by Russell Lee, July 1942). In this situation, a single faller could work a tree by driving a stout stake in the ground, and then attaching strips of rubber inner tube to the stake and one end of a saw. A faller would pull the saw, and a 'rubber-man' would pull it back.

This setup was often used when there weren't enough fallers to form only two-man teams, or when a faller wanted to work alone to collect extra wages or avoid back-talk from a partner. Often, fallers who used rubber-man setups frequently, because they preferred to work alone, would use an iron stake instead of a wood stake, as it tended to be more rigid and durable.

Figure 54 – Setting a choker cable on a large ponderosa pine log in Grant County, Oregon (photo by Russell Lee, July 1942). This image shows very large size associated with centuries-old ponderosa pines being removed from Bear Valley Unit by Edward Hines Lumber Company. When considering the high wood quality shown here, it's a little sad to realize that many of these trees were used to make shipping boxes for apples, cherries, and other fruit from Yakima, Walla Walla, and Hood River valleys.

Figure 55 – Yarding ponderosa pine logs after they were felled and bucked into merchantable lengths; Grant County, Oregon (photo by Russell Lee, July 1942). During this era, crawler tractors were commonly used to haul bucked logs to a collection point where a larger tractor would then skid them to a loader (fig. 56). It was subsequently learned, however, that crawler tractors often caused unacceptable levels of soil compaction, depending on soil type, soil moisture, and other factors.

Note that the surrounding stand contains some tree species other than ponderosa pine, although sale contracts of this era typically did not require removal of tree species other than ponderosa pine, or removal of these 'other species' was at discretion of a purchaser.

Subsequent forest stand dynamics research showed that leaving what was commonly referred to as 'inferior species' on dry-forest sites (Neff 1928, Starker 1915) contributed to compositional shifts that eventually resulted in substantial defoliation impacts during western spruce budworm and Douglas-fir tussock moth outbreaks, two insect defoliators that historically played minor roles on dry-forest sites when they were dominated by ponderosa pine instead of Douglas-fir and grand fir (Powell 1994, Williams 1978, Williams et al. 1980).

Figure 56 – Skidding logs to a loader in Grant County, Oregon (photo by Russell Lee, July 1942). As noted for figure 55, this photo depicts a surrounding stand containing tree species other than ponderosa pine. In fact, the tractor is approaching a western larch on its right side with dwarf mistletoe infection in its lower branches. Note high levels of soil disturbance associated with these major skid trail routes.

Figure 57 – Loading ponderosa pine logs on a truck for transport to a mill yard in Grant County, Oregon (photo by Russell Lee, July 1942). After a large crawler tractor delivers its load of logs to a loading area (fig. 56), a boom-type loader lifts them up and swings them over to a log trailer, where they are added to a load in preparation for transport to a mill yard in town.

Once again, note a mixed-species composition for the background forest stand, and a high level of soil disturbance in the foreground scene.

Figure 58 – A full load of large ponderosa pine logs before being transported to a mill yard in Grant County, Oregon (photo by Russell Lee, July 1942). Mills of this era used large circular saws specifically designed to process large-diameter logs, and logs shown in this image were typical for timber sales up through the late 1980s.

These logs were an important raw material for lumber mills operating in small communities throughout the Blue Mountains. In mid to late 1950s, almost every community in the Blue Mountains supported at least one sawmill, including communities as small as Troy, Oregon.

It may be hard to believe when considering contemporary conditions, but in eastern Oregon, 49 communities had timber-processing mills in mid to late 1950s – 33 communities had one mill, 10 communities had two mills, and 6 communities had three or more mills (Gedney 1963).

Figure 59 – Loading large ponderosa pine logs onto railroad cars in Baker County (photo by Russell Lee, 1941). As noted at the beginning of this appendix, railroad logging was not uncommon in the Blue Mountains, particularly in association with high-volume timber sales such as Bear Valley Unit.

Most logging railroads (and they were plentiful) were narrow-gauge, although track and ties have now been removed and little evidence remains of their existence, other than an abandoned rail bed (although some old rail beds have been incorporated into a 'rails to trails' recreation system).

APPENDIX 3: REGENERATION MONITORING RESULTS

This appendix provides regeneration monitoring results for dry upland forests of Umatilla National Forest (table 16). It summarizes tree density (stems per acre), by species as grouped by seral status, for 76 plots established in plantations located on a Dry Upland Forest potential vegetation group (71 plots are Managed Stand Survey installations; 5 plots were established during a Forest Plan review of regeneration results). Sources/Notes section at end of table 16 provides additional information about origin of this data.

Plots are grouped hierarchically – first by plant association (mean values are provided for each association), and second by potential vegetation group (mean values are provided for a Dry Upland Forest PVG overall).

This table provides monitoring information to inform dry-forest prescriptions for tree planting (reforestation). Planting is a high-cost activity in a vegetation management realm – when considering both internal costs (contract administration, seed procurement, etc.) and external costs (service contract for out-planting, seedling procurement, etc.), total planting cost often runs from $300 to $500 per acre.

As budgetary resources continue to decline, land managers must consider reforestation options that could be implemented at lower cost. Regeneration monitoring data presented in this appendix demonstrates that dry-forest sites tend to support abundant amounts of natural regeneration, and that much of this regeneration has relatively high levels of species diversity.

When considering a Dry Upland Forest PVG in its entirety (see “Mean: Dry Upland Forest PVG” row at bottom of table 16), pines have high average density levels (ponderosa pine averages 243 stems per acre and lodgepole pine averages 138 stems per acre). Relatively high amounts of Douglas-fir and grand fir regeneration are also present (180 stems per acre for grand fir and 143 stems per acre for Douglas-fir), which is not surprising because seed rain and seedling frequency for these species are known to be up to an order of magnitude higher than for pines, larch, and other early-seral tree species (Zald et al. 2008). Therefore, ‘fir be gone’ prescriptions designed to specifically reduce (but not eliminate) representation of Douglas-fir and grand fir are justified somewhat as a counterbalance to their regeneration proficiency.

National Forest Management Act of 1976 (P.L. 94-588) (NFMA) states that when trees are cut to achieve timber production objectives, cuttings shall be made in such a way that “there is assurance that such lands can be adequately restocked within 5 years after harvest” (sec. 6, (g), (3), (E), (ii)). This statement has been interpreted in various ways, but it does <u>not</u> mean that reforestation (tree planting) must occur within 5 years of timber harvest (Watrud et al. 2012).

A NFMA passage quoted above uses the word ‘assurance’ when describing a 5-year regeneration requirement. The Washington Office has interpreted ‘assurance’ to mean that we must use our best efforts and best judgment to assure that restocking occurs within five years. Other interpretations of NFMA language suggest that regeneration cutting should not be prescribed for areas where previous experience suggests that restocking will likely not occur in 5 years, regardless of whether stocking is derived from natural regeneration, tree planting, or both.

An interest in prompt reforestation following harvest is also expressed in other language from NFMA: “Sec. 3 (d) (1) It is the policy of the Congress that all forested lands in the National Forest System be maintained in appropriate forest cover with species of trees, degree of stocking, rate of growth, and conditions of stand designed to secure the maximum benefits of multiple use sustained yield management in accordance with land management plans.”

USDA Forest Service has defined appropriate forest cover as “vegetation composed of plant communities, which would occur naturally on similar sites depending upon the stage of plant succession. Forbs, grasses, and shrubs in their proper ratios are also elements of forest cover” (FSM 2470, section 2472.05 – Definitions). This interpretation of appropriate forest cover is well aligned with recent science findings highlighting the ecological importance of early-successional stages (Swanson et al. 2011).

I recommend that tree planting be considered as a post-harvest activity for dry upland forests receiving a regeneration cutting treatment, and I also recommend that a planting decision be informed by regeneration monitoring results presented in table 16. Table 16 data suggests that natural regeneration is often abundant for dry upland forest sites, but much of it is comprised of mid- or late-seral tree species. Therefore, tree planting could be prescribed to establish an ecologically appropriate forest cover, including an appropriate mix of early-seral tree species in the context of an early stage of plant succession (Swanson et al. 2011).

Forest Vegetation Simulator (FVS) Regeration Modeling Considerations

None of the Pacific Northwest variants of FVS include a regeneration establishment model. If they did, FVS would periodically interject ‘background’ levels of natural regeneration (e.g., ingrowth), and composition and amounts of regeneration would vary with a stand’s plant association code (e.g., assumptions about periodic ingrowth would vary by plant association).

For most areas of Pacific Northwest, relatively high levels of background ingrowth are a fact of life, and they should be reflected in growth-and-yield simulations. Since the Blue Mountains variant of FVS is not interjecting ingrowth automatically, users need to add it manually by invoking either ‘natural’ or ‘plant’ keywords. Regeneration monitoring results presented in this appendix provide a reasonable basis for formulating credible ingrowth scenarios for dry upland forests of northern Blue Mountains (Umatilla National Forest).

[Note: A Managed Stand Survey (MSS) inventory process was initiated in late 1980s. Initial installations (1-acre plots) were installed in young, managed stands throughout the Pacific Northwest Region. One reason for initiating an MSS program was to obtain long-term information about ingrowth and young-stand development, and then use it to calibrate FVS and develop variant-specific regeneration establishment models. Unfortunately, MSS plots were never remeasured, so they could not provide long-term trend data suitable for making FVS revisions.]

Table 16: Regeneration monitoring results for dry upland forests of the Umatilla National Forest.

Plot	Plant Association	PAG	Early Seral					Mid Seral			Late Seral				Other Spp	Total
			WJ	PP	LP	WL		DF	WP		ES	GF	SF			
			<——— All values in these columns are Trees per Acre ———>													
2012	DF/CAGE	WD		84			84	80		80						164
2259	DF/CAGE	WD		147			147	217		217		100		100		464
2508	DF/CAGE	WD		105			105	8		8						113
2513	DF/CAGE	WD		144			144	64		64		44		44		252
2515	DF/CAGE	WD		189			189	72		72						261
2527	DF/CAGE	WD		88	425		513	61		61		4		4		578
2531	DF/CAGE	WD		27			27	44		44		20		20		91
2780	DF/CAGE	WD		108			108	68		68		4		4		180
	Mean: DF/CAGE			112	425		165	77		77		34		34		263
2003	DF/CARU	WD		36		4	40	131		131		584		584		755
2807	DF/CARU	WD		99			99	475		475		8		8		582
2809	DF/CARU	WD		4		40	44	111		111		61		61		216
2815	DF/CARU	WD		291		8	299	163		163		80		80		542
	Mean: DF/CARU			108		17	121	220		220		183		183		524
2808	DF/PHMA	WD		8		7	15	311		311		4		4		330
UMA15	DF/PHMA	WD		170			170	10		10						180
	Mean: DF/PHMA			89		7	93	161		161		4		4		255
2020	DF/SYAL	WD	93	51			144	108		108						252
2806	DF/SYAL	WD		15		7	22	167		167		80		80		269
2836	DF/SYAL	WD		4		8	12	200		200		60		60	20	292
UMA3	DF/SYAL	WD		155		10	165	10		10						175
UMA8	DF/SYAL	WD		840		30	870	60		60						930
	Mean: DF/SYAL		93	213		14	243	109		109		70		70	20	384
2514	DF/VAME	WD		1623			1623									1623
2005	GF/CAGE	WD			100	60	160	367		367		20		20		547

Plot	Plant Association	PAG	WJ	PP	LP	WL	Early Seral	DF	WP	Mid Seral	ES	GF	SF	Late Seral	Other Spp	Total
			<——— All values in these columns are Trees per Acre ———>													
2009	GF/CARU	WD	20	569		420	1009	20		20		504		504		1533
2015	GF/CARU	WD		147	20	4	171	52		52		251		251		474
2016	GF/CARU	WD		36	20		56	156		156		164		164		376
2250	GF/CARU	WD		465			465	179		179		104		104		748
2251	GF/CARU	WD		63		35	98	121		121		315		315		534
2260	GF/CARU	WD		335		4	339	12		12	20	100		120		471
2503	GF/CARU	WD		360	235	32	627	1180		1180		553	40	593		2400
2528	GF/CARU	WD		628	165	701	1494	373		373		1069	100	1169		3036
2813	GF/CARU	WD		117		4	121	111		111		317		317		549
2823	GF/CARU	WD		519		4	523	439		439		160		160		1122
UMA4	GF/CARU	WD		195			195	145		145		80		80		420
	Mean: GF/CARU		20	312	110	151	463	253		253	20	329	70	343		1060
2772	GF/SPBE	WD		139			139	120		120		28		28		287
2810	GF/SPBE	WD		11		7	18	240		240		1037		1037		1295
2814	GF/SPBE	WD		31		4	35	229		229		87		87		351
2831	GF/SPBE	WD						69		69		92		92		161
UMA5	GF/SPBE	WD		80		190	270	390		390		170		170		830
	Mean: GF/SPBE			65		67	116	210		210		283		283		585
2023	PP/AGSP	HD	100	139			239									239
2779	PP/AGSP	HD		159			159	12		12						171
	Mean: PP/AGSP		100	149			199	12		12						205
2000	PP/CAGE	WD	20	99			119	24		24	143	167	167	477		620
2002	PP/CAGE	WD	228	203			431	11		11						442
2013	PP/CAGE	WD	104	191			295									295
2022	PP/CAGE	WD	80	600			680									680
2502	PP/CAGE	WD		225			225	92		92						317
2505	PP/CAGE	WD		140			140	100		100						240
2509	PP/CAGE	WD		163			163	4		4		20		20		187

Plot	Plant Association	PAG	WJ	PP	LP	WL	Early Seral	DF	WP	Mid Seral	ES	GF	SF	Late Seral	Other Spp	Total
			<———— All values in these columns are Trees per Acre ————>													
2511	PP/CAGE	WD		1072			1072	67		67		24		24		1163
2520	PP/CAGE	WD		120			120	4		4						124
2523	PP/CAGE	WD		117			117									117
2525	PP/CAGE	WD		1025		4	1029	20		20		51		51		1100
2536	PP/CAGE	WD		765		100	865	44		44						909
	Mean: PP/CAGE		108	393		52	438	41		41	143	66	167	143		516
2519	PP/CARU	WD		233			233									233
2522	PP/CARU	WD		205	4		209	8		8						217
2534	PP/CARU	WD		172			172									172
	Mean: PP/CARU			203	4		205	8		8						207
2521	PP/CELE/CAGE	WD		173			173	20		20						193
2510	PP/CELE/FEID-AGSP	HD		1020			1020	68		68		20		20		1108
2524	PP/CELE/FEID-AGSP	HD		40			40									40
	Mean: PP/CELE/FEID-AGSP			530			530	68		68		20		20		574
2001	PP/FEID	HD	64	301			365									365
2006	PP/FEID	HD	160	71			231	40		40						271
2007	PP/FEID	HD	20	188			208									208
2014	PP/FEID	HD	40	307			347	120		120						467
2021	PP/FEID	HD	44	199			243									243
2252	PP/FEID	HD		147			147	7		7						154
2253	PP/FEID	HD		12			12	4		4						16
2500	PP/FEID	HD		139			139	101		101						240
2501	PP/FEID	HD	4	64			68	4		4						72
2506	PP/FEID	HD	44	75			119									119
2526	PP/FEID	HD		304			304	40		40						344
2535	PP/FEID	HD		92			92									92
	Mean: PP/FEID		54	158			190	45		45						216

Plot	Plant Association	PAG	WJ	PP	LP	WL	Early Seral	DF	WP	Mid Seral	ES	GF	SF	Late Seral	Other Spp	Total
			<——————— All values in these columns are Trees per Acre ———————>													
2008	PP/SYAL	WD	60	253			313	1041		1041		60		60		1414
2018	PP/SYAL	WD	100	168			268									268
2255	PP/SYAL	WD		359			359	440		440						799
2804	PP/SYAL	WD		109			109	16		16						125
2805	PP/SYAL	WD		185			185	7		7		20		20		212
2811	PP/SYAL	WD		132			132	8		8						140
2812	PP/SYAL	WD		116			116	8		8						124
	Mean: PP/SYAL		80	189			212	253		253		40		40		440
Mean: Dry Upland Forest PVG			74	243	138	77	291	143		143	82	180	102	193	20	495

Sources/Notes: **Plot** includes two types of plots: numbers refer to plots from a Managed Stand Survey (MSS), a plot-based system (5-point plot cluster covering about 1 acre) installed in 1990 in young, managed stands with an average stand diameter of 3 inches or more. Plots beginning with UMA were part of a Forest-wide reforestation monitoring effort completed in 1994 (16 plots installed in randomly selected reforestation units across the Umatilla National Forest; see Powell 1995). **Plant association** is an acronym formed from a 2-digit tree species (DF = Douglas-fir; GF = grand fir; PP = ponderosa pine) and a 4- or 5-digit understory species code (AGSP = bluebunch wheatgrass; CAGE = elk sedge; CARU = pinegrass; CELE = mountain mahogany; FEID = Idaho fescue; PHMA = mallow ninebark; SPBE = birchleaf spiraea; SYAL = common snowberry; VAME = big huckleberry). **PAG** refers to plant association group (WD is warm dry; HD is hot dry; see Powell et al. 2007). Columns are provided for individual tree species (in addition to species codes already mentioned, WJ = western juniper; PP = ponderosa pine; LP = lodgepole pine; WL = western larch; DF = Douglas-fir; WP = western white pine; ES = Engelmann spruce). **Early Seral** is a sum of preceding four species columns; **Mid Seral** is a sum of preceding two columns; **Late Seral** is a sum of preceding three columns; other species (Spp) includes Pacific yew, hawthorn, willow, and paper birch. **Total** is a summed tree density value, as trees per acre, for all individual species columns.

Note: tree density values include total tree stocking, including three categories of trees: (1) trees established by out-planting (e.g., trees originating as nursery-produced seedlings), (2) trees established as natural regeneration (trees originating from natural seeding occurring after timber harvest or another disturbance), and (3) trees present before a disturbance process (including 'advance' regeneration and mature trees from a previous stand that survived the disturbance process).

Mean values are presented for each plant association, and at the bottom of this table for Dry Upland Forest PVG as a whole. They were calculated in such a way that plots where a tree species or seral stage did not occur (there is a blank in the species or seral stage column) were not included in the calculation (in other words, blanks were not treated as zero values when calculating Means).

DRY-FOREST REFERENCES AND LITERATURE CITED

This section includes literature cited in text, along with other references having relevance to ecology and management of dry forests in the Blue Mountains of northeastern Oregon, southeastern Washington, and west-central Idaho.

Cautionary note about dry-forest references: users of literature in this section should consider that concerns about dry forest exist for all of western North America, so research involving dry-forest ecosystems spans a broad geographical area ranging from southern Okanogan valley of British Columbia to Black Hills of South Dakota, Colorado's Front Range, Mogollon Rim in Arizona, and Sierra Nevada Mountains of California.

I believe it is useful for practitioners to be aware of a wide breadth of dry-forest research, and I have attempted to provide a relatively diverse array of sources in this section.

It is also important to recognize that dry-forest studies pertaining to Blue Mountains and adjacent portions of interior Pacific Northwest qualify as primary, place-based research, findings from northern Rockies (particularly for areas located west of Continental Divide) qualify as valuable secondary sources, and works from the Southwest or Sierra Nevada are tertiary sources.

Note: I believe Sierra Nevada research is more appropriate for summer-dry Mediterranean climates of the Blue Mountains than sources derived from monsoon climates of the Southwestern U.S. or the southern Front Range of Colorado.

This cautionary note is particularly germane to literature describing dry-forest reference conditions. Excellent insights about historical conditions for ponderosa pine ecosystems of the southwestern U.S. are provided by Gus Pearson (1923), Gil Schubert (1974), Charles Avery (Avery et al. 1976), Charles Cooper (1960, 1961a), and others, but in my opinion, their characterizations of presettlement stand structure (stocking levels, etc.) should not be extrapolated to dry-forest ecosystems of eastern Oregon, *or extrapolation should be attempted very carefully*, due to climatic and environmental differences between these two regions.

With few exceptions, sources contained in this References section are available from World Wide Web in digital form, and a Digital Object Identifier (doi) is included for these items whenever possible.

[Digital object identifier is an international system used to uniquely identify, and link to, electronic versions of scientific information, primarily journal articles. A doi can be thought of as a 'catalog number' for journal articles and other non-book sources.]

All doi links pertain to formally published sources only; local analysis protocols, white papers (like this one), monitoring reports, and similar items will not have a doi.

For recent USDA Forest Service research reports (general technical reports, research papers, research notes, conference proceedings, etc.), a doi may also be available. But most reports do not yet have a doi, so a doi is not included for reports in this References section.

For FS research items, however, this section provides a weblink for the online Treesearch system, because most FS research reports are available for download from Treesearch.

When preparing a white paper, one of my objectives is to help users locate its references

and literature citations. For journal articles or books, I provide a doi or isbn number whenever one is available. For other reference materials, a weblink is provided, although I realize that weblinks have not been stable (except for USDA Forest Service Treesearch links, which have been quite stable thus far).

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APPENDIX 4: SILVICULTURE WHITE PAPERS

White papers are internal reports, and they are produced with a consistent formatting and numbering scheme – all papers dealing with Silviculture, for example, are placed in a silviculture series (Silv) and numbered sequentially. Generally, white papers receive only limited review and, in some instances pertaining to highly technical or narrowly focused topics, the papers may receive no technical peer review at all. For papers that receive no review, the viewpoints and perspectives expressed in the paper are those of the author only, and do not necessarily represent agency positions of the Umatilla National Forest or the USDA Forest Service.

Large or important papers, such as two papers discussing active management considerations for dry and moist forests (white papers Silv-4 and Silv-7, respectively), receive extensive review comparable to what would occur for a research station general technical report (but they don't receive blind peer review, a process often used for journal articles).

White papers are designed to address a variety of objectives:

(1) They guide how a methodology, model, or procedure is used by practitioners on the Umatilla National Forest (to ensure consistency from one unit, or project, to another).
(2) Papers are often prepared to address ongoing and recurring needs; some papers have existed for more than 20 years and still receive high use, indicating that the need (or issue) has long standing – an example is white paper #1 describing the Forest's big-tree program, which has operated continuously for 25 years.
(3) Papers are sometimes prepared to address emerging or controversial issues, such as management of moist forests, elk thermal cover, or aspen forest in the Blue Mountains. These papers help establish a foundation of relevant literature, concepts, and principles that continuously evolve as an issue matures, and hence they may experience many iterations through time. [But also note that some papers have not changed since their initial development, in which case they reflect historical concepts or procedures.]
(4) Papers synthesize science viewed as particularly relevant to geographical and management contexts for the Umatilla National Forest. This is considered to be the Forest's self-selected 'best available science' (BAS), realizing that non-agency commenters would generally have a different conception of what constitutes BAS – like beauty, BAS is in the eye of the beholder.
(5) The objective of some papers is to locate and summarize the science germane to a particular topic or issue, including obscure sources such as master's theses or Ph.D. dissertations. In other instances, a paper may be designed to wade through an overwhelming amount of published science (dry-forest management), and then synthesize sources viewed as being most relevant to a local context.
(6) White papers function as a citable literature source for methodologies, models, and procedures used during environmental analysis – by citing a white paper, specialist reports can include less verbiage describing analytical databases, techniques, and so forth, some of which change little (if at all) from one planning effort to another.
(7) White papers are often used to describe how a map, database, or other product was developed. In this situation, the white paper functions as a 'user's guide' for the new product. Ex-

amples include papers dealing with historical products: (a) historical fire extents for the Tucannon watershed (WP Silv-21); (b) an 1880s map developed from General Land Office survey notes (WP Silv-41); and (c) a description of historical mapping sources (24 separate items) available from the Forest's history website (WP Silv-23).

The following white papers are available from the Forest's website: Silviculture White Papers

Paper #	Title
1	Big tree program
2	Description of composite vegetation database
3	Range of variation recommendations for dry, moist, and cold forests
4	Active management of Blue Mountains dry forests: Silvicultural considerations
5	Site productivity estimates for upland forest plant associations of Blue and Ochoco Mountains
6	Blue Mountains fire regimes
7	Active management of Blue Mountains moist forests: Silvicultural considerations
8	Keys for identifying forest series and plant associations of Blue and Ochoco Mountains
9	Is elk thermal cover ecologically sustainable?
10	A stage is a stage is a stage…or is it? Successional stages, structural stages, seral stages
11	Blue Mountains vegetation chronology
12	Calculated values of basal area and board-foot timber volume for existing (known) values of canopy cover
13	Created opening, minimum stocking, and reforestation standards from Umatilla National Forest Land and Resource Management Plan
14	Description of EVG-PI database
15	Determining green-tree replacements for snags: A process paper
16	Douglas-fir tussock moth: A briefing paper
17	Fact sheet: Forest Service trust funds
18	Fire regime condition class queries
19	Forest health notes for an Interior Columbia Basin Ecosystem Management Project field trip on July 30, 1998 (handout)
20	Height-diameter equations for tree species of Blue and Wallowa Mountains
21	Historical wildfires in headwaters portion of Tucannon River watershed
22	Range of variation recommendations for insect and disease susceptibility
23	Historical vegetation mapping
24	How to measure a big tree
25	Important Blue Mountains insects and diseases
26	Is this stand overstocked? An environmental education activity
27	Mechanized timber harvest: Some ecosystem management considerations
28	Common plants of south-central Blue Mountains (Malheur National Forest)
29	Potential natural vegetation of Umatilla National Forest

Paper #	Title
30	Potential vegetation mapping chronology
31	Probability of tree mortality as related to fire-caused crown scorch
32	Review of “Integrated scientific assessment for ecosystem management in the interior Columbia basin, and portions of the Klamath and Great basins” – Forest vegetation
33	Silviculture facts
34	Silvicultural activities: Description and terminology
35	Site potential tree height estimates for Pomeroy and Walla Walla Ranger Districts
36	Stand density protocol for mid-scale assessments
37	Stand density thresholds as related to crown-fire susceptibility
38	Umatilla National Forest Land and Resource Management Plan: Forestry direction
39	Updates of maximum stand density index and site index for Blue Mountains variant of Forest Vegetation Simulator
40	Competing vegetation analysis for southern portion of Tower Fire area
41	Using General Land Office survey notes to characterize historical vegetation conditions for Umatilla National Forest
42	Life history traits for common Blue Mountains conifer trees
43	Timber volume reductions associated with green-tree snag replacements
44	Density management field exercise
45	Climate change and carbon sequestration: Vegetation management considerations
46	Knutson-Vandenberg (K-V) program
47	Active management of quaking aspen plant communities in northern Blue Mountains: Regeneration ecology and silvicultural considerations
48	Tower Fire…then and now. Using camera points to monitor postfire recovery
49	How to prepare a silvicultural prescription for uneven-aged management
50	Stand density conditions for Umatilla National Forest: A range of variation analysis
51	Restoration opportunities for upland forest environments of Umatilla National Forest
52	New perspectives in riparian management: Why might we want to consider active management for certain portions of riparian habitat conservation areas?
53	Eastside Screens chronology
54	Using mathematics in forestry: An environmental education activity
55	Silviculture certification: Tips, tools, and trip-ups
56	Vegetation polygon mapping and classification standards: Malheur, Umatilla, and Wallowa-Whitman National Forests
57	State of vegetation databases for Malheur, Umatilla, and Wallowa-Whitman National Forests
58	Seral status for tree species of Blue and Ochoco Mountains

REVISION HISTORY

July 2011: initial version of this white paper was circulated for technical peer review by Rick Brown (Defenders of Wildlife), Craig Schmitt (USDA Forest Service, Blue Mountains Service Center, La Grande), and Carrie Spradlin (USDA Forest Service, Umatilla National Forest, Heppner Ranger District). Their reviews contributed significantly to this white paper!

July 2012: minor formatting and editing changes were made (renumbered section headings and figure captions); a new section (7.11) providing selected wildlife considerations was added, along with additional literature citations.

November 2012: minor formatting and editing changes were made throughout the document; the References section was corrected; appendix 2 was added describing the silviculture white paper system, including a list of available white papers.

April 2013: minor formatting and editing changes were made throughout the document; new text (a quote) was added to section 3.1; additional text and a new figure were added to the ungulate herbivory discussion (section 5); a new figure was added to the climate change discussion (section 7.13).

January 2014: this was a significant update: formatting and editing changes were made throughout the document, including a renumbering of all subsections within section 7; additional photographs were added (livestock grazing, selective cutting, and restoration sections); new material was added about dry-forest stocking levels and how they could be applied (figs. 30-32 and table 6); a new figure was added describing western juniper expansion on dry sites; and a Summary section was added before the References section.

December 2014: minor formatting and editing changes were made, including the addition of new photographs as figures; a new appendix 2 was added describing early 1940s timber harvest practices in the Blue Mountains by using a series of photographs from Grant County and the Malheur National Forest; a new appendix 3 was added providing regeneration monitoring results for 76 plots established on dry upland forest sites on the Umatilla National Forest; and additional references were added to the References section, along with weblinks and digital object identifiers (doi) to improve access to more dry-forest references.