WHITE PAPER

USDA Forest Service Pacific Northwest Region Umatilla National Forest

WHITE PAPER F14-SO-WP-SILV-7

Active Management of Blue Mountains Moist Forests: Silvicultural Considerations

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

Initial Version: **JANUARY 2013**
Most Recent Revision: **DECEMBER 2019**

Moist forest field trip in northern Blue Mountains (Aug. 2012)

CONTENTS

1. INTRODUCTION

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

This white paper discusses silvicultural considerations associated with active management of moist upland forests[1], a biophysical environment found most commonly in the northern Blue Mountains, and to a lesser extent in the central and southern Blue Mountains (table 1, fig. 2). Appendix 1 provides a list of the potential vegetation types (plant associations, plant community types, plant communities) occurring in the moist upland forest potential vegetation group (PVG).

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

Potential Vegetation Group	North Half	South Half	Total
Cold Upland Forest	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 the Umatilla National Forest geographical information system.

Although moist upland forests of the Blue Mountains have been actively managed since the 1950s (see section 5.13), some stakeholders and publics began to raise concerns about moist-forest management proposals in the late 2000s. Comments received in response to environmental analysis documents assert that the Forest Service lacks a credible scientific rationale for proposing treatments on moist-forest sites, that moist-forest management is an area of scientific controversy, and that best available science does not support active management of moist forests.

---

[1] A companion white paper (F14-SO-WP-Silv-4) discusses silvicultural considerations for active management of dry upland forests (Powell 2014a).

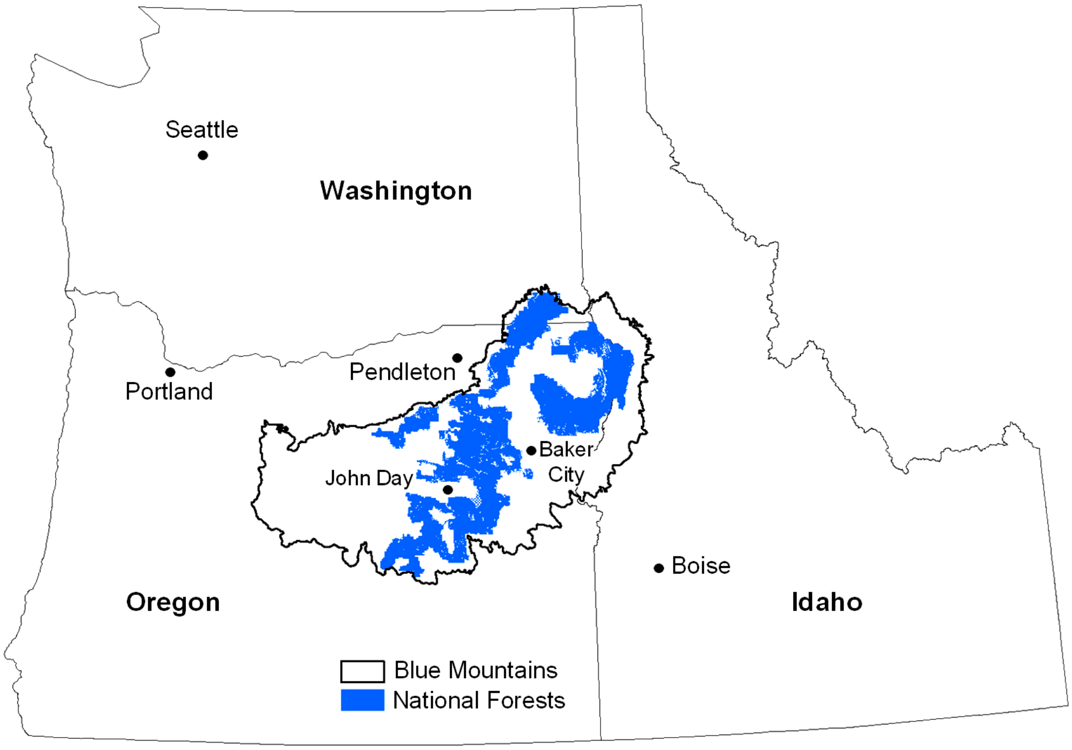


Figure 1 – The Blue Mountains ecoregion of northeastern Oregon, southeastern Washington, and west-central Idaho consists of a series of mountain ranges in a southwest to northeast orientation, extending from the Ochoco Mountains in central Oregon to the western edge of the Seven Devils Mountains in west-central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

This white paper is one response to concerns raised about moist-forest management since the late 2000s. This document's intent is not to promote any particular perspective, as in a 'correct' interpretation of moist-forest management, but to provide concepts, principles, and historical context supporting further dialogue about active management of Blue Mountains moist forest. Since it provides background information and context only, it is not intended to be a 'cookbook' or 'how-to' guide.

Why develop a white paper about moist-forest management? The basic concept behind this approach is that project planning processes tend to be similar across Blue Mountain national forests (such as environmental analyses required by NEPA, the National Environmental Policy Act), so little justification exists for unnecessarily providing the same background information in every NEPA document.

Some background information used for project-level NEPA, including material related to biophysical characterization, potential vegetation classification, disturbance ecology, and forest structural stages, is largely unchanged from one analysis to another. An example is assignment of potential vegetation types (PVTs) to potential vegetation groups (PVGs) (i.e., which PVTs qualify as moist forest?); these assignments seldom vary from one planning area to another, so the methodology could be described in a report or white paper and then cited (Powell et al. 2007).

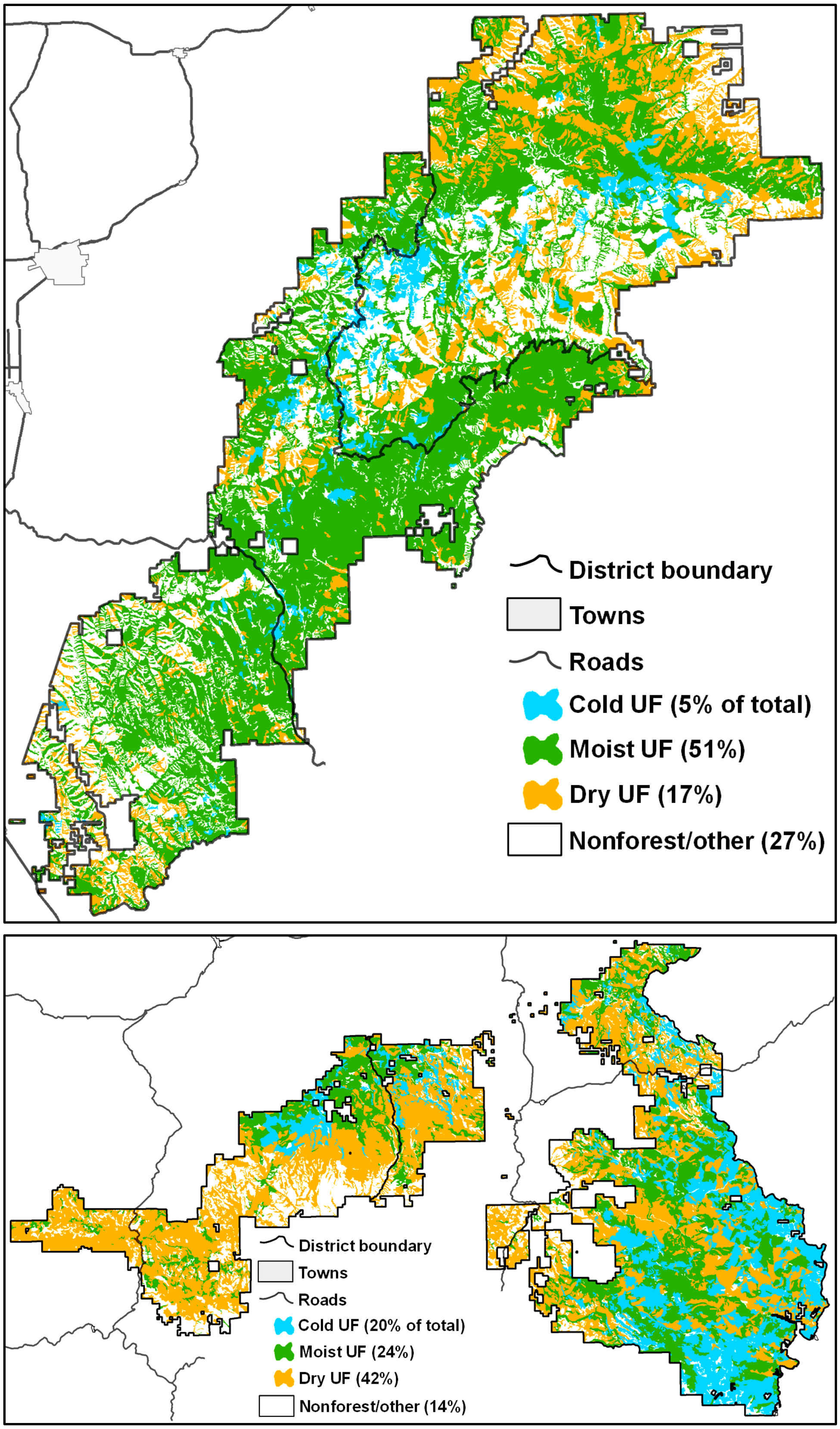


Figure 2 – Distribution of upland forest (UF) potential vegetation groups on the Umatilla National Forest (north-end districts above; south-end below).

During project planning, USDA Forest Service sometimes assumes that background information sources are not readily available to readers of NEPA documents (by presuming a lack of internet access, or similar constraints). Under this assumption, document preparers may decide to include quite a bit of background material in the NEPA document itself (instead of citing white papers, general technical reports, and other sources).

In actuality, I believe it is more efficient to direct a NEPA document reader to a white paper or general technical report for background information, thereby allowing the document itself to address specifics of a planning area and a project proposal. By minimizing unnecessary background information, this approach focuses a NEPA document on treatment locations and specifications, design features, environmental effects, and other details about a project and its proposed actions.

My objective for this white paper about active management of moist upland forests in the Blue Mountains is to:

1. Help establish a foundation of relevant literature ('best available science'), concepts, principles, and context relating to moist-forest management. Note that science is synthesized here from my frame-of-reference as a practitioner.
2. Synthesize and summarize background information in a citable reference supporting project planning (NEPA) processes for Blue Mountains national forests (e.g., Malheur, Umatilla, and Wallowa-Whitman national forests).
3. Define and characterize moist-forest environments of the Blue Mountains.
    a. Describe an ecological setting for moist forests of the Blue Mountains: how are they defined in a potential vegetation context (section 2 and appendix 1)?
    b. Provide a narrative describing historical application of management activities in moist-forest ecosystems (section 3): how did historical management practices contribute to existing conditions for moist forests?
    c. Describe disturbance ecology concepts, including a discussion about how active management can be designed to emulate natural disturbance processes (provided by section 4 primarily, but also by portions of section 6).
    d. Describe important disturbance agents and processes affecting moist forests (section 5): defoliating insects (western spruce budworm and Douglas-fir tussock moth), dwarf mistletoes, stem decays, root diseases, bark beetles (mountain pine beetle, Douglas-fir beetle, fir engraver), wildfire, windstorms, timber harvest, invasive species (white pine blister rust, balsam woolly adelgid), and climate change.
    e. Conclude with a discussion about active vegetation management considerations, including recommendations for moist forests (section 6).

Formatting notes: Unlike the dry-forest white paper (F14-SO-WP-Silv-4), which disperses glossary terms throughout the text, this white paper includes a stand-alone glossary section. Eight short topics of special interest are provided as consecutively numbered Boxes. At the beginning of long sections (sections 4-6), a short summary for the section is provided in a gray box.

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 vegetation diversity within each zone (fig. 3) and borders that are poorly defined: herbaceous communities and stands of deciduous trees are scattered throughout the coniferous forest, and the species of dominant conifer changes from one site to another (Powell 2000).

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

Potential vegetation is defined as the community of plants that would become established if all successional sequences were completed, without interference by humans, under existing environmental conditions (Hall et al. 1995). It also implies that over the course of time and in the absence of future 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 ideal for strategic assessments (Johnson et al. 1999, REO 1995; fig. 4).

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

Potential vegetation (PV) is used to classify biophysical environments because it has an important influence on ecosystem processes. It is the ecological engine that powers vegetation change – it controls the speed at which shade-tolerant species get established beneath shade-intolerant trees, the rate at which forests produce and accumulate biomass, and the impact that fire, insects, pathogens, and other disturbance agents have on species composition, forest structure, and tree density.

The implications of these ecological and successional processes are predictable (within limits) because they can be related to PV, and sites with similar PV tend to behave in a similar way (Cook 1996, Daubenmire 1961).

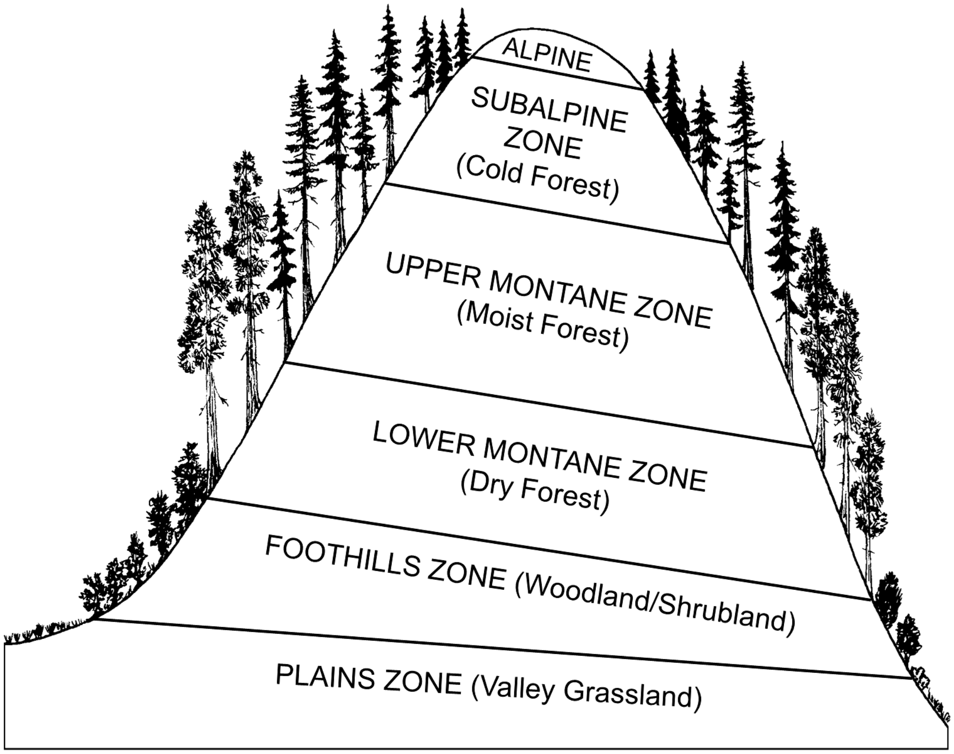


Figure 3 – Vegetation zones of the Blue Mountains (adapted from Powell 2000). In the northern hemisphere, a south-facing slope receives more solar radiation than a flat surface, and a north-facing slope receives less (south slope is to the left, and north is to the right). These solar radiation patterns result in the vegetation zones or bands shown here – they are arranged vertically in response to elevation (moisture), and sloping downward from south to north (left to right) in response to slope direction or aspect (temperature).

Plains zone contains grasslands and shrublands because moisture is too low to support forests except along waterways. Foothills zone is usually dominated by western juniper, although for the maritime zone of the northern Blues, it supports tall shrubs. Located above foothills environments is a lower montane zone, which contains dry mixed-conifer forests in ponderosa pine, grand fir, and Douglas-fir potential vegetation series. An upper montane zone includes moist mixed-conifer forests in Douglas-fir, grand fir, and subalpine fir series (moist upland forests are the subject of this white paper). This moist mixed-conifer zone consists of 13 grand fir types, 10 subalpine fir or fir-spruce types, and 3 Douglas-fir types (plus miscellaneous aspen and lodgepole pine community types; see appendix 1 for specifics). High elevations in the Blues support a subalpine zone with Engelmann spruce and subalpine fir, or an alpine zone near mountain summits where trees are absent.

Because of its predictive power, PV is extremely useful for estimating the impact of disturbance processes on differing ecological environments (also see section 6.5). 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 PV sites where the overstory trees have thick bark (ponderosa pine, Douglas-fir, larch). The same burn has dramatically different results (near-complete tree mortality) on cold PV sites dominated by thin-barked subalpine firs and lodgepole pines.

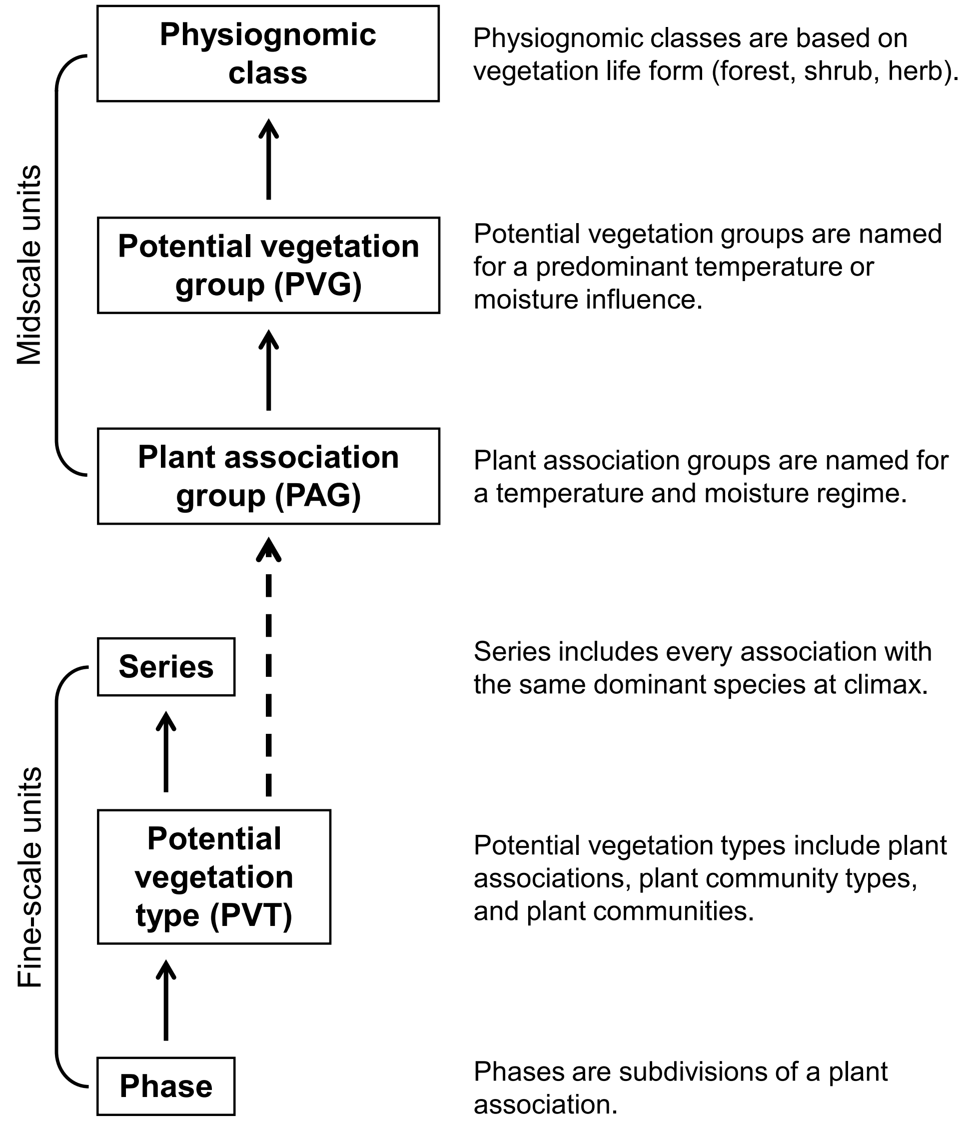


Figure 4 – Hierarchy of potential vegetation (PV) for the Blue Mountains (from Powell et al. 2007). PV taxonomic units have been organized as two integrated portions of a hierarchy. Fine-scale hierarchical units are described in PV classification reports (Crowe and Clausnitzer 1997, Johnson 2004, Johnson and Clausnitzer 1992, Johnson and Simon 1987, Johnson and Swanson 2005, and Wells 2006); midscale units are primarily described in reports from an Interior Columbia Basin Ecosystem Management Project (such as Jensen et al. 1997). 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.

2.1 Moist Upland Forest Potential Vegetation Group

Moist upland forests tend to occur at moderate elevations of the montane vegetation zone and at low elevations of the subalpine zone. Late-seral stands are dominated by subalpine fir, grand fir, or Douglas-fir as the climax tree dominants, while lodgepole pine or western larch often occur as early-seral species in this biophysical environment. Douglas-fir and western white pine function as mid-seral species (except on sites where Douglas-fir is climax). Moist forests are adjoined by cold upland forests at their upper edge, and by dry upland forests at their lower edge (fig. 3).

As described earlier, potential vegetation represents the underlying foundation on which the biological landscape is constructed. It functions as a biophysical template (geology, soils, climate) controlling which tree species, and the proportions of each, that can exist for any particular suite of physical site factors – each unique combination of site factors results in a slightly different temperature and moisture regime. For the Blue Mountains, temperature and moisture regimes are used to establish plant association groups and potential vegetation groups (Powell et al. 2007).

For the Blue Mountains, the Moist Upland Forest PVG consists of five plant association groups (PAG) – three in the cool temperature regime (Cool Wet, Cool Very Moist, and Cool Moist PAGs), and two in the warm temperature regime (Warm Very Moist and Warm Moist PAGs). The Cool Moist PAG is by far and away the most common member of the Moist UF potential vegetation group (appendix 1). Cool moist forests are moister than warm dry forests at lower elevations, and warmer than cold moist forests at higher elevations.

Cool, moist forests (defined as potential vegetation types assigned to the cool moist upland forest plant association group) tend to occupy the most productive forested environments of the Blue Mountains because moisture is usually not limiting – the temperate nature of this PAG is reflected in high species diversity and a closed forest structure. The high floristic species diversity pertains to both the overstory (forest) composition (when considering all of the early-, mid-, and late-seral tree species associated with this PAG), and to the undergrowth plant union.

Moist-forest undergrowths are dominated by forbs, some mid-height shrubs, and a few tall shrubs in warmer environments. Moist-site plants such as queencup beadlily, twinflower, false bugbane, swordfern, and ginger occur in this zone, but the most common mesic environments within the Moist Upland Forest PVG have big huckleberry as the undergrowth dominant. Moist forests at the warm end of the temperature spectrum feature mid or tall shrubs such as Rocky Mountain maple, ninebark, and oceanspray – these occur in the Warm Very Moist and Warm Moist plant association groups.

Moist Forest Classification Concepts

Some readers are confused by the fact that moist forest also includes subalpine fir plant associations. Traditionally, sites with a subalpine fir climax potential have been assigned to the subalpine (cold forest) zone depicted in figure 3. This traditional classification of subalpine fir associations tends to focus on the series level (see fig. 4) – this approach places all plant associations (habitat types) with the same climax tree species in the same series bin (the climax tree species is used to name a forest series, such as the subalpine fir series).

The 'one size fits all' assumption exemplified by the series approach can be useful for broad-scale assessments because the dominant (climax overstory) tree species (the series) reflects the macroclimate (regional climate) of an area. At a broad or regional scale, vegetation patterns are assumed to reflect macroclimatic trends, and these trends are best represented by the series level. This is in contrast to the subordinate indicator plants of a plant association, which are assumed to represent an area's microclimate and soils. For the *Abies grandis/Clintonia uniflora* plant association (grand fir/queencup beadlily), *Abies grandis* (the grand fir series) is assumed to reflect the macroclimatic regime, whereas *Clintonia uniflora* (the subordinate indicator plant) is responding to microclimatic and site conditions (Powell et al. 2007).

Plant associations are distinct entities varying in response to landform, topography, geology, soils, and other biophysical site factors; they are not just random groupings of plants brought together by chance (Westveld 1951). Since the Blue Mountains temperature-moisture classification system (Powell et al. 2007) assigns some subalpine fir associations to moist forest, and others to cold forest, it explicitly assumes that potential vegetation is a good indicator of biophysical relationships expressed as temperature and moisture gradients.

When assigning plant associations to temperature-moisture classes in the Blue Mountains system, the subordinate indicator plants had more influence than the dominant (climax) tree species. This reality resulted in subalpine fir plant associations representing moderate or mesic site conditions (subalpine fir/queencup beadlily, etc.) being assigned to the moist upland forest PVG, and subalpine fir plant associations representing harsh or cold bioclimatic conditions (subalpine fir/grouse huckleberry, etc.) being assigned to the cold upland forest PVG (Powell et al. 2007).

The classification concepts described here, which were used to separate subalpine fir plant associations into a moist bin and a cold bin, are closely aligned with ecosystem insights from other portions of the interior Pacific Northwest – Daubenmire (1956) found that the occurrence of subalpine forest types (as represented by the subalpine forest zone depicted in fig. 3) was more closely correlated with summer air temperatures (relating to the ecological distribution of cold forest) than with precipitation patterns (relating most closely to moist forest distribution).

Appendix 1 summarizes the potential vegetation composition associated with moist forests – it shows which of the potential vegetation types (PVTs, e.g., plant associations (habitat types), plant community types, plant communities) were assigned to the Moist Upland Forest potential vegetation group, and it shows how each of the PVTs were assigned to a plant association group. As described above, the Moist Upland Forest PVG includes five plant association groups.

Appendix 1 describes how some of the 'best examples' of moist forest in the northern Blue Mountains are associated with the Maritime-Influenced Zone, which is a Level IV ecoregion established for the Blue Mountains ecoregion (moist forest is also associated with the Mesic Forest Zone shown in app. 1) (Clarke and Bryce 1997).

3. HISTORICAL CONTEXT

Those who cannot remember the past are condemned to repeat it
George Santayana, American philosopher and poet

During the pioneer era of Euro-American settlement in the Pacific Northwest, timber production made significant contributions to economic development. In some portions of this region, timber was viewed initially as an obstacle to be cleared to make way for agriculture. The forests were considered limitless, and there was relatively little appreciation of forest values other than timber production.

By the 1960s and early 1970s, forest management practices were being closely scrutinized; controversies on the Monongahela (West Virginia) and Bitterroot (western Montana) national forests eventually spurred passage of the National Forest Management Act in 1976. Between the 1950s and the 1980s, an era when timber production still held primacy, the following land management strategies were commonly employed:

- Prompt suppression of wildfire.
- Sanitation and salvage of high-value trees at risk of insect or disease attack.
- Detection and chemical control of defoliating-insect outbreaks.
- Attempted eradication of introduced pests such as white pine blister rust.

The goals of this era were clear: protect the forest from natural and human-caused disturbances until the timber could be harvested (Puettmann et al. 2009). Disturbance was viewed as an economic disruption to be prevented (Berryman 1991), rather than an ecosystem process to be emulated (Perera et al. 2004). This philosophy was based on a command-and-control approach in "an effort to control nature in order to harvest its products, reduce its threats, and establish highly predictable outcomes for the short-term benefit of humanity" (Holling and Meffe 1996).

Over time, societal expectations for our forests evolved. Ecosystem services such as fish and wildlife habitat, water quality, recreation, and visual aesthetics are now appreciated and demanded by society (Daily et al. 1997). Good examples of evolving societal expectations are provided by the Brundtland Report (Brundtland and Khalid 1987) and the Millennium Ecosystem Assessment (Hassan et al. 2005). And during this era of broad-based societal change, managers were continually expanding their knowledge and awareness about the complexities of forest ecosystems.

Forested landscapes have definable characteristics with respect to their scenic attractiveness. People value highly-scenic landscapes, which research has shown to be those with a natural appearance based on their landform, vegetation patterns, and water characteristics (Lucas 1991, Magill 1992). In some portions of the Blue Mountains, landscape patterns have been altered by previous timber harvest practices, particularly regarding the effects of clearcutting. Often, one result of clearcutting was a visual pattern whose texture, form, line, and color were out of scale with natural landscapes. Square or rectangular clearcuts provide good examples of a human-induced pattern inconsistent with natural landscape patterns. Clearcutting patterns generated high amounts of societal concern about forest management.

It is generally thought that societal concern about the environment is a relatively recent development, perhaps dating from Rachel Carson's *Silent Spring* (Carson 1962) or the first Earth Day observance in 1970. Regardless of its source, an expanding environmental movement brought growing public and legislative pressure to reform natural resource management, as reflected by passage of the National Environmental Policy Act, Endangered Species Act, National Forest Management Act, and Clean Water Act during the 1970s.

An environmental perspective also began infiltrating the forestry profession: "the notion of a forest as a source of what humans need and want is being replaced by the metaphor of a forest as a living supra-organism, with measures of health and a sense of organismal integrity comparable to that of the human being. This metaphor views unmanaged forests that have developed over many tree generations without large-scale, intense disturbance as the epitome of nature" (Kimmins 1996).

Ecosystem Management

Beginning in the 1990s, the U.S. Forest Service adopted ecosystem management as a new paradigm emphasizing an ecological approach to natural resource management (Christensen et al. 1996, Grumbine 1994, Kessler et al. 1992). This paradigm change means "the agency no longer manages forests on federal lands for sustained timber yields. The recent approach focuses on sustaining human value systems such as conservation and preservation, on managing ecosystems, and on coming up with strategies to protect California spotted owl and other wildlife species" (Gruell 2001). Ecosystem management seeks an understanding of how human activities can be consistent with biodiversity and landscape conservation objectives (Berkes 2004).

Research in silviculture, insect and disease susceptibility, watershed management, wildlife habitat relationships, plant succession and forest dynamics, and fire ecology over the last 40 years has provided an excellent knowledge base for ecosystem management (Steen 1999). Even so, philosophical differences between advocates of active or passive management remain polarized to such an extent that land managers are continually seeking new ways to apply this knowledge while also obtaining some level of social license for resource management (Pfister 1993).

3.1 Integrated Forest Protection

Advances in the fields of animal and plant ecology, along with other basic and applied natural sciences, were incorporated in the 1970s into a new concept of dealing with pest-caused disturbances – integrated pest management (IPM) (Waters and Stark 1980). IPM was formally recognized in the National Forest Management Act of 1976; it is based on the concept of understanding an organism's life cycle, causal relationships, developmental dynamics, resource effects, and control or preventive measures. The paradigm of the previous era – pest control through the use of persistent, broad-spectrum chemicals – was deemphasized or phased out entirely under IPM (Stoszek 1988b).

IPM initially made its greatest strides with agriculturists and weed managers. Certain agricultural sectors have long relied on biocontrol measures, and demand for an IPM approach continues to increase as consumers demand more organic produce. Biocontrol emphasizes use of parasites, predators, and disease organisms to mitigate the damaging effects of insects and weeds. IPM techniques promote sustainability because as compared to chemical pesticides, they are either self-perpetuating or much less disruptive to nontarget organisms (Haack and Byler 1993).

Integrated Forest Protection And Active Management

We now recognize that disturbances, particularly those derived from biotic agents, develop largely in response to stress-related changes in the forest (Waring and Running 1998), and although it might be simplistic to state it in these terms – a disturbance event can function to alleviate or release the stress condition that prompted its occurrence in the first place (Castello and Teale 2011).

Research has shown that disturbance-prone situations are identifiable, and that many of them can be modified by using silvicultural practices (Seidl et al. 2011). For this reason, forest protection concerns can be, and often are, integrated into the silvicultural decision-making process. The research also demonstrates that the occurrence, frequency, and damage severity of disturbances tend to be associated with site- and stand-specific factors, both natural and anthropogenic. In this sense, many of the potential damage outcomes for key disturbance processes are predictable in both a spatial (where) and temporal (when) context (Stoszek 1988b).

Advances in forest science, particularly forest ecology, during the last four decades led to a deeper appreciation of the many roles played by biotic and abiotic disturbance processes, including their influence on stand dynamics as manifested in secondary forest succession (Johnson and Miyanishi 2007). In particular, the detrimental effects of fire exclusion, the impact of indiscriminate insecticide use, and the influence of nutrient stress have been identified as important factors affecting forest ecosystems across the western United States (Stoszek 1988b).

Successful bark beetle attacks are now viewed as symptoms of trees under physiological stress brought about by site- and stand-related factors, climatic patterns, effects of pathogenic organisms, and management-induced changes to the forest (fire exclusion, timber harvest, industrial pollutants, exotic organisms, etc.). Silvicultural interventions designed to address physiologically weakened stand conditions are now viewed as crucial to a forest health strategy (Castello and Teale 2011).

Scientific inquiry into host colonization patterns and mating behavior for major insect agents led to discovery of sex attractants (pheromones) and anti-aggregation compounds produced by insects (Wood 1982). Substantial contributions have been made in the area of root disease and stem decay ecology, insect and pathogen interactions with predisposing factors, dwarf mistletoe mitigation using silvicultural practices, and improved assessment of disturbance effects, including better remote sensing techniques providing enhanced spatial resolution (Sharik et al. 2010).

The widespread utilization of geographic information system (GIS) technology allows land managers to gain access to a wide variety of spatially-explicit information about ecological site conditions, mensurational stand metrics, land use allocations, and operability or implementation considerations. GIS permits grouping of forest stands into strata according to land allocations, site characteristics, ecological site potentials, and any number of other criteria (Horning et al. 2010).

Models conceptualizing the insights gained from studies examining host-organism relationships are available for a wide range of western insects and diseases. Managing a forest for 'desired future conditions' requires that managers have access to risk rating protocols and similar options for predicting the future, but to be successful, these tools must account for the functions of insects and diseases because they have such a strong influence on the future (Haack and Byler 1993).

It is now accepted that matching species and seed source to biophysical environment is essential to reducing impact from both biotic (insects/diseases) and abiotic agents (frost, heat, drought). It is no longer acceptable to culture forest stands by using traditional regeneration and density management practices (planting a single species at close seedling spacings such as 8 × 8 feet, followed quickly by a noncommercial thinning), and then assume that any future pest problems can be suppressed if and when they occur (Byler and Zimmer-Grove 1991, Stoszek 1988b).

Research has shown that improper planting techniques and/or poor seedling stock contributes to what can be termed 'self-destructive' forests because the ultimate effects of ill-formed root systems are expressed well beyond the stand initiation phase of forest development (Stoszek 1988b). This result suggests that insects and pathogens are capable of responding to seemingly insignificant changes in forest ecosystems, including poorly-formed root systems (Byler and Zimmer-Grove 1991).

Abiotic conditions (terrain and soil), in combination with the tree species present in an area, are all involved in the predisposition of an area to damage from snow, wind, frost, heat, and other abiotic disturbance agents. This means that consideration of protection measures relies on identification of processes predisposing trees and stands to disturbance-induced damage. It also suggests that disturbance planning must account for specific conditions at both the stand and landscape levels.

It has been found that high levels of animal damage often result from habitat conditions created by inappropriate timber harvest methods, site preparation practices, or shrub control techniques. In fact, animal damage levels are often predictable on the basis of forest ecosystem changes, particularly in how they are expressed in subordinate shrubs and herbs (Black 1992, 1994; Ferguson et al. 2005).

4. DISTURBANCE ECOLOGY CONCEPTS

SECTION CONTENTS

Section Summary

- Disturbance processes operate within a disturbance regime; the disturbance regime is used to characterize the prevailing spatial size, distribution, frequency, return interval, intensity, and severity of a disturbance process.
- Disturbance has an important influence on ecosystem diversity. When human activities (fire exclusion, etc.) modify a disturbance regime, resulting changes contribute to landscape simplification by modifying ecosystem components such as species composition, forest structure, and stand density. Disturbance processes may be especially imporant for maintaining limited vegetation types such as quaking aspen forest.
- The shifting mosaic concept of landscape development is discussed, especially in the context of the intermediate disturbance hypothesis (IDH). IDH is germane to moist forests because they tend to have moderate (intermediate) environmental conditions, resulting in moderate amounts of environmental harshness and moderate levels of disturbance intensity and severity. This moist-forest situation presents a sharp contrast to moisture-limited dry forests, and with temperature-limited cold forests.
- Forest disturbance initiates a developmental progression referred to as a sere; each stage in the progression is called a seral stage. Each seral stage has a particular suite of associated plants and animals. For any particular biophysical environment, an early-seral stage tends to have dramatically different tree and plant species than a late-seral stage. Effective management must recognize differences between seral stages, including how tree species seral status varies from one environment to another. Mixed stands containing both early- and late-seral species tend to develop differently than single-species stands, a process called physiognomic forest succession.
- Section 4 provides a detailed discussion about concepts and principles associated with disturbance emulation – can active management be used to emulate (mimic) the native disturbance regime, and how might this occur? Disturbance emulation, also known as the Natural Disturbance Model (Box 3), is a concept being used worldwide as a template for contemporary forest management.
- This section discusses old moist forest in a disturbance ecology context – how did disturbance regimes influence old forest on moist sites, and are some portions of a moist-forest landscape more likely to support old forest than others (e.g., refugia)?
- Seven principles govern disturbance regimes and their ecosystem effects (all are adapted from White 1987):

✓ Disturbances occur at a variety of temporal and spatial scales.
✓ Disturbances affect many levels of biological organization.
✓ Disturbance regimes vary, both regionally and within any particular landscape.
✓ Disturbances overlay environmental gradients, both influencing and being influenced by the gradients.
✓ Disturbances interact and can be synergistic.
✓ Disturbances may result from feedback between the state of a plant community and its vulnerability to disturbance.
✓ Disturbances produce variability in communities.

Disturbance, a primary initiator of plant succession, is important and integral for moist-forest ecosystems. A disturbance is defined as a relatively discrete event that disrupts the structure of an ecosystem, plant community, or population, and in doing so, it changes resource availability or the physical environment. Disturbances happen over relatively short time intervals: windstorms occur over hours to days, fires occur over hours to weeks, and volcanoes erupt over periods of days or weeks (Foster et al. 1998, Turner et al. 1997). At least 7 disturbance agents have a major influence on moist upland forests of the Blue Mountains (table 2).

4.1 Disturbance Regimes

Ecologists often distinguish between a discrete disturbance event – like an individual windstorm or wildfire – and the long-term disturbance regime that shapes an ecosystem or landscape. A disturbance regime refers to the spatial and temporal dynamics of disturbance events over long time periods (Turner 1998). "Taken together, the attributes of all the disturbances occurring in a system, the interactions between them, and their linkages with biotic and abiotic factors, define the disturbance regime" (White et al. 1999). Characterization of a disturbance regime typically includes the items described in table 3.

Disturbances come in all shapes and sizes, ranging from relatively minor to relatively major events. They can be caused by biotic agents (insects, diseases, animal damage) or by abiotic factors (wind, fire, and flood). Since disturbances vary in both frequency and magnitude, the spatial and temporal impact of any particular disturbance event depends upon the hierarchical scale being considered (fig. 5). An example of disturbance scaling is the burrowing activity of pocket gophers and other small mammals, which could be perceived as a disturbance process at a very fine-grained spatial scale (Meadows and Meadows 1991), but it is unlikely to be viewed in that way at a broad scale (Huntly and Inouye 1988, White 1979).

A disturbance, and the recovery period following it, can have important influences on ecosystem function. On the one hand, destabilizing forces (disturbance processes) are important for maintaining ecosystem diversity and resilience. On the other hand, stabilizing forces (growth and maturation) are important for maintaining inherent productivity and biogeochemical cycles (Holling 1996, Reice 1994).

External stresses are a normal part of any living system. Forest ecologist Hamish Kimmins expressed it this way: "Where these stresses are within the range that the system has experienced historically, the system has adapted to them and they are 'normal.' Where there is a long history of recurrent stress, such stress is usual" (Kimmins 1996). And ecological communities include feedback loops, which can promote resiliency in the presence of a wide range of ecosystem stressors.

Table 2: Common disturbance agents of moist upland forests.

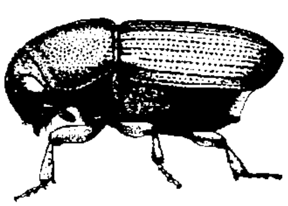
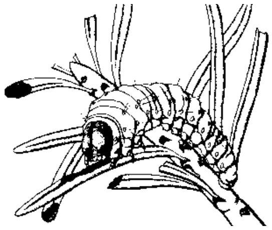
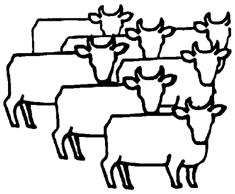
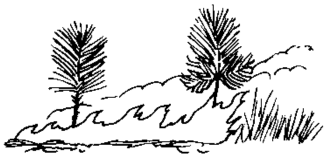
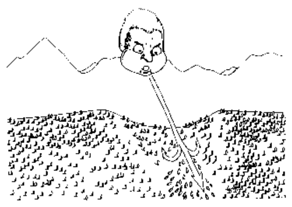
	**Bark Beetles.** Three primary bark beetles influence moist forests: Douglas-fir beetle, mountain pine beetle in lodgepole pine, and fir engraver. Fire, windstorm, disease, defoliating insects, and high tree density are predisposing factors that weaken trees and attract bark beetles. Localized outbreaks provide ecosystem services such as snag creation, but broad-scale outbreaks can contribute to heightened fire risk (Hayes and Daterman 2001).
	**Defoliating Insects.** There are two primary defoliators affecting moist forests: western spruce budworm and Douglas-fir tussock moth. Because defoliator impacts can be intense and widespread during an outbreak, much research effort has been focused on these insects. Large-scale suppression projects using both chemical and biological insecticides have been used to control defoliator outbreaks (Torgersen 2001).
	**Parasites and Pathogens.** Root diseases are localized but still cause tree mortality. Rust-red stringy rot caused by the Indian paint fungus stem decay is common in grand fir stands. White pine blister rust, an introduced disease, affects western white pine. Three species of dwarf mistletoe, a tree parasite, affect moist forests: Douglas-fir dwarf mistletoe, larch dwarf mistletoe, and lodgepole pine dwarf mistletoe (Parks and Flanagan 2001).
	**Timber Harvest.** Timber harvest is used to provide wood products and employment for human society. Tree harvest in the Blue Mountains began in the 1880s but at a much-reduced rate when compared with other areas in eastern Oregon (Weidman and Silcox 1936). Beginning in the early 1940s, tree harvest began increasing to meet a heightened demand during World War II, and to construct new housing after the war (Fedkiw 1999).
	**Ungulate Herbivory.** Historical cattle and sheep grazing affected vegetation conditions, particularly along ridgetops used as sheep driveways or as bedding grounds (Galbraith and Anderson 1970, Irwin et al. 1994, Tucker 1940). Native ungulates (deer, elk) increased dramatically over the last half century and continue at relatively high levels today (Case and Kauffman 1997, Humphrey 1943, Parks et al. 1998, Riggs et al. 2000).
	**Wildfire.** A period of uncharacteristically intense wildfire, which began in the mid 1980s and continues today, might be viewed as a symptom of impaired forest health for dry forests (Ottmar and Sandberg 2001), but probably not for moist forests. Ocean-atmosphere interactions, related primarily to trends in the ENSO and PDO cycles of climatic variability (see sec. 5.11), exert a strong influence on weather patterns and fire hazard.
	**Wind.** Wind was frequently mentioned as a disturbance agent in historical accounts (e.g., Smith and Weitknecht 1915). The infamous 1962 Columbus Day windstorm caused extensive damage in the coastal northwest but had limited impact in eastern Oregon (Lynott and Cramer 1966). A major windstorm occurred during the winter of 1989-1990, affecting moist forests across a relatively broad area in the northern Blue Mountains.

Table 3: Attributes of a disturbance regime.

ATTRIBUTE	DEFINITION
Area or size	Area disturbed, expressed either as area per event or area per time period
Type	Characterizes the function of a disturbance process in terms of initiating a new stand, or maintaining an existing stand (fig. 13)
Distribution	Spatial distribution of disturbance events
Frequency	Mean number of events per time period, or decimal fraction of events per year
Return interval	Inverse of frequency, or the average time interval between successive disturbance events
Intensity	Physical energy of the event per area per time; characteristic of disturbance rather than its ecological effect. Examples include energy release rates for a fire, and wind speed for a windstorm
Severity	Effect of a disturbance event on the organism, community or ecosystem; closely related to intensity because more intense disturbances are generally more severe. Tree mortality as related to fire effects would be one example of severity relationships.

Sources: Adapted from Turner et al. (1998).

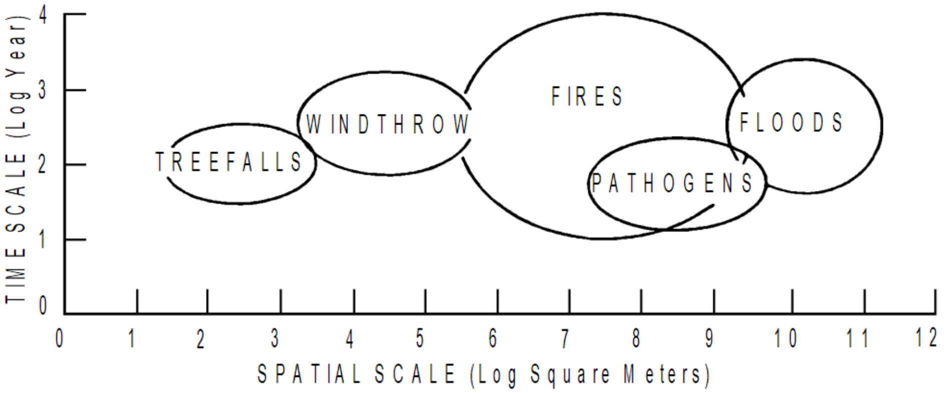


Figure 5 – Idealized temporal and spatial relationships among selected disturbance regimes (redrawn from Urban et al. 1987). This figure demonstrates that disturbance is implicitly scaled. For example, if disturbance is defined as events that kill trees prematurely (the 'treefalls' regime in this figure), then disturbance is confined to a relatively narrow timeframe (temporal scale) when considered in the context of the typical or maximum longevity of Blue Mountain tree species (see table 4). Large-scale disturbances such as fires, regional floods, and volcanoes are spatially heterogeneous; whether a large disturbance is qualitatively different from numerous small disturbances remains an unresolved issue in ecology (Turner et al. 1997).

Another representation of scales hierarchy is discussed in figure 74, presented in section 6 of this white paper. In this figure, the horizontal and vertical dimensions of the oval associated with a disturbance process are intended to convey the relative temporal and spatial extent of each process. Although this figure suggests that disturbance regimes interact (because the ovals overlap), figure 74 describes in more detail how disturbance processes are nested within a hierarchy ranging from gaps to landscapes.

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 the predominant species associated with each seral stage. Tree Longevity (age in years) pertains to the predominant tree species and is taken from Powell (2000).

Knowing the intensity and frequency of disturbance processes is important because plant and animal species are adapted to disturbance effects (e.g., the spatial pattern of vegetation composition and structure at a landscape scale). The species diversity of an area depends on the balance between disturbance frequency and intensity, and the level of competition existing between species (Parminter 1998). "The suppression of disturbances leads to the loss of biological diversity and may contribute to larger and more severe disturbance events later" (White et al. 1999).

The composition and structure of plant communities reflects complex interactions between species life history characteristics, disturbance intensity and frequency, and chance events – suggesting that both deterministic and stochastic factors influence ecosystem variation (Halpern 1989). "Natural disturbance maintains structural complexity, promoting plant and animal diversity" (Hansen et al. 1991).

When multiple disturbances act synergistically (fig. 6) and result in a preponderance of lodgepole pine across large watersheds, some degree of fragmentation might be desired as a way to circumvent landscape-scale insect outbreaks and their associated wildfires. "Beetle infestations create large amounts of fuel, increasing the probability of wildfire. Fire, in turn, initiates development of new, even-aged lodgepole pine stands, which repeats the cycle at some point in the future" (Perry 1988).

Disturbances frequently have a renewal effect by helping to release and recycle nutrients (Pickett 1980). Historically, fire was the predominant recycling agent for dry forests of the interior Pacific Northwest, but moist-forest decomposition relies equally on fire and microbes for this important ecosystem service (fig. 7).

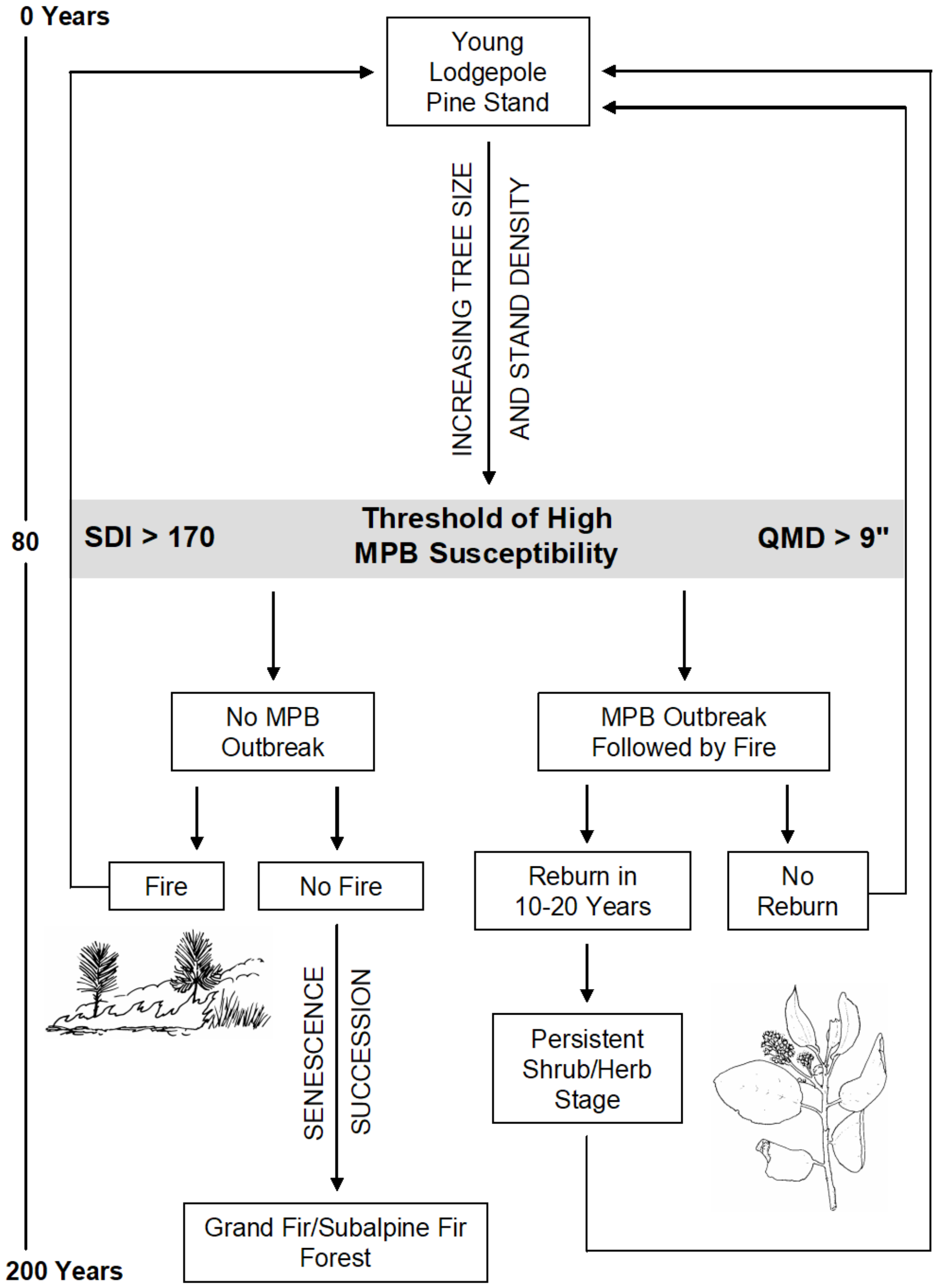


Figure 6 – Interactions between mountain pine beetle (MPB), fuel accumulation, and stand-initiating wildfire (adapted from Powell 2000). This diagram depicts lodgepole pine development and its relationship to wildfire and bark beetles, two disturbance agents influencing this forest type at a landscape scale (Arno et al. 1993, Barrett et al. 1991, Romme and Despain 1989, Stuart et al. 1989). This diagram shows generalized disturbance interactions – the combined effect of multiple agents may influence landscape patterns more than a single agent acting alone (Veblen et al. 1994, Wilson et al. 1998). The gray zone shows a stand development threshold where lodgepole pine susceptibilty to MPB is high: forests with a stand density index (SDI) exceeding 170, and a quadratic mean diameter (QMD) of 9 inches or more, are highly susceptible to MPB attack (Cochran et al. 1994, Peterson and Hibbs 1989).

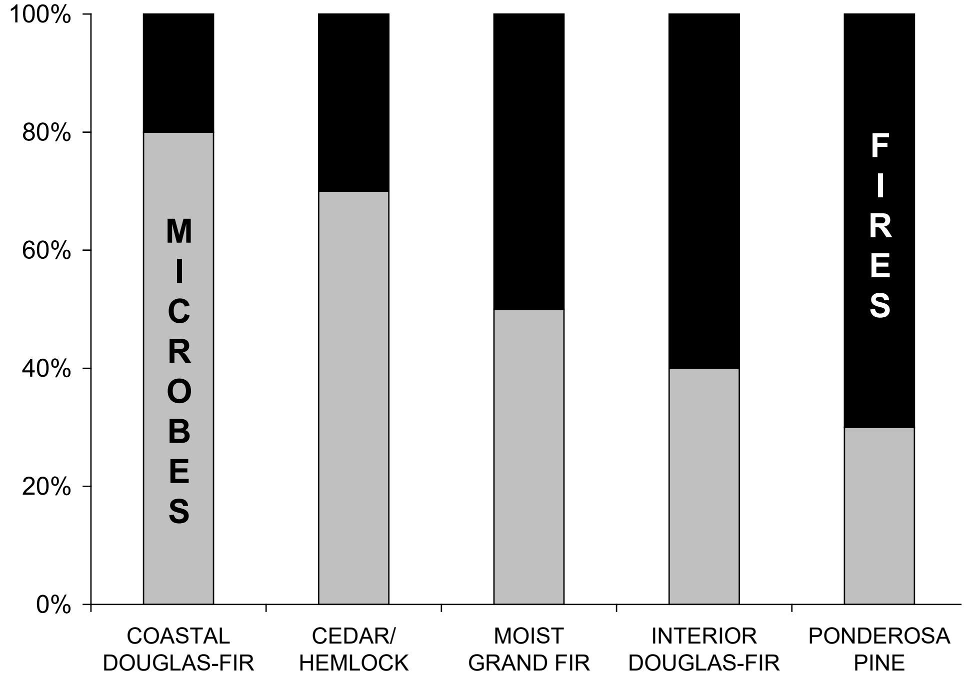


Figure 7 – Microbes and fire as agents of decomposition (adapted from Harvey et al. 1994). Fire (black portion of bars) and microbes (gray portion) are important decomposition agents. For the dry-forest climatic zone of the interior Pacific Northwest (the interior Douglas-fir and ponderosa pine forest types), surface fire was the primary cycling process because microbial decomposition is too slow to keep pace with biomass accumulation on these sites. For moist grand fir forest, decomposition relies equally on fire and microbes. For moist forests of western Oregon and Washington (coastal Douglas-fir, cedar/hemlock), a biological process (microbial decomposition) cycles dead wood; for dry forests of eastern Oregon and Washington, a physical process (fire) is the most important cycler of litter, wood, and associated nutrient sources.

4.2 Disturbance And Diversity

Because disturbances create and maintain the vegetation patterns we see on a landscape, intentional or unintentional shifts in a disturbance regime may cause dramatic changes, particularly for wildlife species whose welfare is influenced by the distribution and juxtaposition of vegetative habitat components (Turner 1998). In this respect, a lack of disturbance can be as damaging to biological diversity and ecological integrity as too much disturbance (Noss 1983).

Some effects of an altered disturbance regime are insidious, initiating successional changes occurring over decades or more (Sloan 1998). Vegetation changes can be so difficult for people to recognize as to be called the ‘invisible present’ (Magnuson 1990), evoking a perception of forest tranquility due to the seemingly timeless nature of large trees (Shugart and West 1981). Perhaps the reason for many contemporary writings focusing on disturbance as an unnatural phenomenon is that for any particular geographical location, severe, stand-replacing disturbance events generally occur on a much longer temporal cycle than human recollection (Parminter 1998).

After humans alter a disturbance regime, it can eventually lead to simplification (homogenization) of a landscape (del Moral 1972, Lehmkuhl et al. 1994, Turner 1998). When a landscape in the Blue Mountains undergoes simplification, the first elements to be affected are often limited vegetation components such as quaking aspen clones (fig. 8); riparian forests of black cottonwood, mountain alder, river birch or willows; western white pine stands; and certain shrubland types or individual shrub species (Box 1) (Fahnestock 1976, Gruell 1983, Habeck 1976, Habeck and Mutch 1973, Hall 1984, Hessl 2002, Voller and Harrison 1998).

A trend toward landscape simplification can ultimately result in impoverished vegetation diversity – extensive areas devoid of quaking aspen, black cottonwood, ninebark, parklike ponderosa pine, big huckleberry, western white pine, and other early- or mid-seral stages (Case and Kauffman 1997, Hunter 1999, Sloan 1998). Losing the early-seral composition is unfortunate because it contributes to biological diversity, scenic beauty, wildlife habitat, and recreational opportunities (Reice 1994).

Without fully functioning disturbance regimes to continually create new ecological niches for early-seral species, these biotic components cannot survive and prosper (Sloan 1998). And since disturbance regimes are responsible for creating and maintaining a landscape's structural and compositional complexity, their disruption can eventually degrade both plant and animal diversity (Hansen et al. 1991).

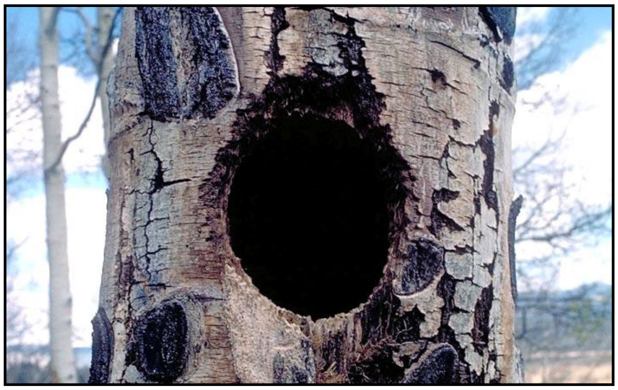
An example of change relating to landscape simplification involves quaking aspen ecosystems (fig. 8). Aspen is an ecosystem component valued for a myriad of benefits. Its leaves and buds are choice food for ruffed grouse, beaver, snowshoe hare, Rocky Mountain elk, and many other species. In winter, when foliage is no longer present, elk feed on its smooth white bark (DeByle 1985). And after dying, aspen may be used by as many species as when alive – dead aspen trees are prized by woodpeckers, flickers, and many other species utilizing cavities (DeByle 1985).

Where aspen communities occur in the western U.S., they are second only to riparian areas in terms of species diversity and abundance. Fire suppression on western landscapes, when coupled with excessive browsing of young aspen trees by livestock and wildlife, has led to rapid displacement of aspen communities by conifer forests (Bartos and Campbell 1998). Conifer forests transpire more water than aspen and support a sparse undergrowth with relatively few plant species, so aspen displacement can result in water yield reductions, along with declines in the number and kinds of plants and animals (Bartos and Campbell 1998, Kay 1994).

It has been predicted that biophysical environments with an intermediate amount of disturbance would have the highest levels of species and functional diversity (Huston 1979). This intermediate disturbance hypothesis (IDH) suggests that biodiversity and ecosystem function are higher in moderately disturbed habitats than either of the extremes (low and high disturbance levels), resulting in a bell-shaped curve when biodiversity and disturbance severity are compared (Biswas and Mallik 2010) (figure 9).

An aspen clone (from Powell 1994). Although historical information suggests that aspen was never abundant in the Blue Mountains, the lack of fire as a landscape process, along with ungulate browsing, caused aspen to seek refuge along streams, near moist meadows, and on other moisture-accumulating physiographic positions.

Bark wounding on quaking aspen caused by elk browsing during winter. Aspen provides important winter habitat for large native ungulates, which may cause bark wounding, eventually leading to establishment of stem cankers such as sooty-bark, black, cryptosphaeria, cytospora, and hypoxylon (Hinds 1985).



Bird cavity in aspen bole. Aspen often has heart rot (stem decay). Decayed aspen wood is soft and easily excavated, making aspen a favorite tree species for woodpeckers, flickers, and a wide variety of other birds within the cavity-utilizing guild.

Figure 8 – Selected features of aspen ecosystems in the Blue Mountains.

Box 1. Moist Forests and Ungulate Herbivory

Large native ungulates (primarily elk) can influence the vigor and longevity of Pacific yew and other shrubs of the Blue Mountains (Parks et al. 1998). Pacific yew is a shade-tolerant, woody perennial plant occurring occasionally as a small tree but most often as a tall shrub in the undergrowth of late-successional grand fir stands on moist grand fir plant associations. Biophysical environments supporting Pacific yew are cool and moist.

Pacific yew and several other plants are especially palatable and highly utilized by wild ruminants. Riggs et al. (2000) reported that wild ungulates reduced yew abundance in their study area, similar both floristically and ecologically to the study site used by Parks et al. (1998), to such an extent it affected the outcome of using classification keys to identify a plant association – the keyed association was grand fir/Pacific yew/twinflower inside an exclosure, whereas it was grand fir/queencup beadlily outside the exclosure. A similar situation exists for another so-called ‘ice cream’ shrub of the Blue Mountains – Oregon boxwood (*Pachistima myrsinites*). [Continued on next page.]

What was done. The study was conducted in September 1992 in the West Sinks area of the Umatilla National Forest, north of La Grande, Oregon. Field examinations in preparation for a timber sale had identified a high incidence of yew dieback and mortality in an unharvested grand fir stand with late-successional or old-growth characteristics. Many of the wounded or dead yew plants had obvious physical damage (bark stripping or rubbing causing stem wounds) in addition to foliage browsing indicators. Close examination revealed grooves on some yew stems that seemed to match the characteristics of incisors located on the lower jaw of elk, and no pathogen- or insect-caused evidence appeared to be associated with the dieback. Transect lines were established to sample 5% of the total area; data was collected on a range of yew characteristics and for wounded plants, detailed measurements were made of the wounds and their possible causes. Plant demography information was collected for a total of 712 Pacific yew stems.

What was found. Of the 712 stems measured in the study, about 66% were rated as less than 50% dead, about 17% were rated as dying (defined as 50-100% dead), about 1% were classified as recent dead, and less than 1% were older dead. Catherine Parks and her colleagues "deduced that a high percentage of stem damage on yew in the West Sinks area is caused primarily by deer and elk, who use their teeth to incise, then strip and consume the bark" (Parks et al. 1998, page 193). They went on to note that "nonfeeding damage by animals included antler rubbing, clawing by black bears, and hoof abrasions." The authors also noted that the literature contains abundant citations about ungulate browsing of yew foliage, but very few items dealing primarily with physical damage to yew plants caused by non-browsing activities.

Their demographic information indicated a narrow range of establishment dates for mature yew plants, leading to speculation that a low-intensity surface fire occurred in the area about 90-100 years ago and caused widespread mortality of mature yew plants, but was perhaps successful at stimulating yew regeneration arising primarily from seed (sprouting from the root collar or crown, however, might have been a more likely source for a new yew cohort). They also found that existing yew reproduction (fruit, vegetative, seedlings) was low, and they detected a correlation between the presence of fruit and sprout production (perhaps indicating that when plant vigor is high enough to produce fruit, then it is also sufficiently high to foster sprout production?).

The authors could not state statistically that a negative relation was present between male plants and sprouting, but felt their data supports such a conclusion. They noted that in other parts of the yew range, male yew plants tend to be single stemmed and female plants multi-stemmed. The authors believe that a lack of older regeneration reflects the fact that "yew seedlings are consumed by ungulates once they are large enough to be recognized" (Parks et al. 1998, page 195).

Summary. Parks et al. (1998) concluded that high concentrations of native ungulates appear to be an important factor influencing dieback and mortality of mature yew plants, and low amounts of young yew recruitment. Although Pacific yew, Oregon boxwood, and similar shrubs are relished by wild ungulates, and they provide important ruminant browse, this service can only be sustained when ungulate populations are in some sort of dynamic equilibrium with plant populations. If a great disparity exists between ungulates producing pressure on the plants, and opportunities for the plants to escape this pressure long enough to mature and reproduce, then we can expect negative ecosystem consequences in terms of low yew populations, along with reduced availability of ungulate browse if viable alternatives are not available from non-yew sources.

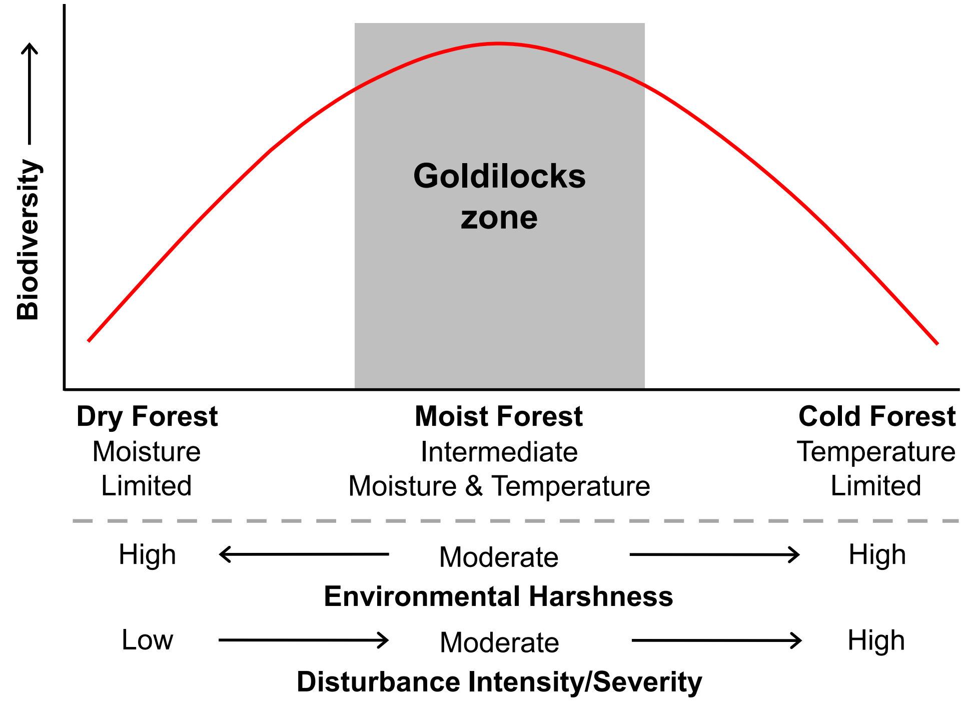


Figure 9 – Intermediate disturbance hypothesis (adapted from Biswas and Mallik 2010, Ewel 1999, and White and Jentsch 2001). The intermediate disturbance hypothesis suggests that biophysical environments with a moderate disturbance regime would have the highest levels of species and functional diversity (Huston 1979, Petraitis et al. 1989). The black, bell-shaped line depicts species diversity across a temperature-moisture gradient ranging from dry to cold forest. For the Blue Mountains, dry forests historically experienced frequent disturbance (surface fire on a cycle of 5-20 years), but the diversity of disturbance agents was relatively low because these moisture-limited sites have relatively low productivity. Cold forests have very infrequent disturbances (such as stand-replacing fire on a cycle of 80-110 years; see table 4), but the diversity of disturbance agents is also low because these temperature-limited sites have relatively low productivity. But moist forests exist in the Goldilocks zone because they are not too cold, not too dry – they are 'just right' in terms of their temperature and moisture relationships. Thus, moist forests experience moderate levels of environmental harshness and disturbance severity. The moisture, temperature, environmental harshness, and disturbance severity relationships depicted in this figure help explain why species composition and biodiversity levels tend to be highest for moist forests. Particularly diverse ecological settings are said to have high amounts of beta diversity (Perry et al. 2008); for upland forest environments of the Blue Mountains, moist forests have the highest levels of beta diversity.

IDH postulates that species richness is maximized by intermediate levels of disturbance – theoretically, they allow both early- and late-seral species to coexist (e.g., aspen with conifer). If disturbance is rare, then competitive exclusion results in domination by late-seral species; if disturbance is frequent, early-seral species predominate. But in either case, the result is similar – less species diversity than would be expected for intermediate disturbance levels (Kuuluvainen and Grenfell 2012).

Coarse Filter Approach For Maintaining Diversity

If historical disturbance regimes had been allowed to maintain an appropriate range of ecological conditions in Blue Mountain landscapes, then they could have played an important role in perpetuating both species and genetic diversity (Haufler 1994). This approach is referred to as a coarse filter for conservation of biological diversity; it is based on the premise that native species are adapted to indigenous disturbances and their resulting range of habitats (Hunter 1990, Hunter et al. 1988).

A coarse filter reflects the fact that we cannot even name or enumerate all of the species in a landscape, much less rationally plan for their habitat needs or anticipate their response to disturbance (Cissel et al. 1994). Applying prescribed fire on sites where the surface-fire disturbance regime has been suppressed since early in the 20th century, for example, resulted in a dramatic increase in butterfly abundance and species richness – and it is likely that this ecosystem response was a serendipitous and unexpected benefit of the treatment (Huntzinger 2003).

4.3 Shifting Mosaic Concept

Plant communities reaching a stable condition (the climax end-point of succession) are rare, although a mosaic of different communities at a broad scale can exhibit a stable frequency distribution of vegetative states (Lertzman and Fall 1998, Niering 1987). Many landscapes exist as a shifting mosaic of patch types (Clark 1991, Watt 1947). The shifting-mosaic concept demonstrates that disturbance is scale dependent; disturbance events occurring at a fine scale (the patch level) may be viewed as stabilizing forces at a landscape scale because scale differences keep the disturbance effects disjunct (Allen and Wyleto 1983).

Thus, steady-state conditions can exist at a landscape scale but not at the scale of individual patches – a single stand can be in any seral state (early-, mid-, or late-seral) at a particular time, but if the disturbance regime is relatively consistent, then large landscapes exhibit a quasi-equilibrium (for seral states) across long time periods. [And, this is reason that when disturbance process simulation modeling is used to estimate the long-term temporal patterns of disturbance regimes for a region, hundreds of simulations spanning thousands of years are completed to examine whether a dynamic equilibrium exists and, if so, what its characteristics might be.]

Although forest patches are always changing in response to the dynamic forces of disturbance and succession, the integrated effect at a larger scale is a shifting mosaic of developmental phases or structural stages. The shifting mosaic concept suggests that an overall balance of patch births and deaths can produce a dynamic equilibrium – there is local change at a fine scale, but the total number of young and old communities remains relatively constant at a broad scale in response to disturbance processes and plant succession (fig. 10) (Bormann and Likens 1979, Clark 1991, Drury and Nisbet 1973, Pickett and Ostfeld 1995, Urban et al. 1987, Vale 1988, Voller and Harrison 1998, White 1987).

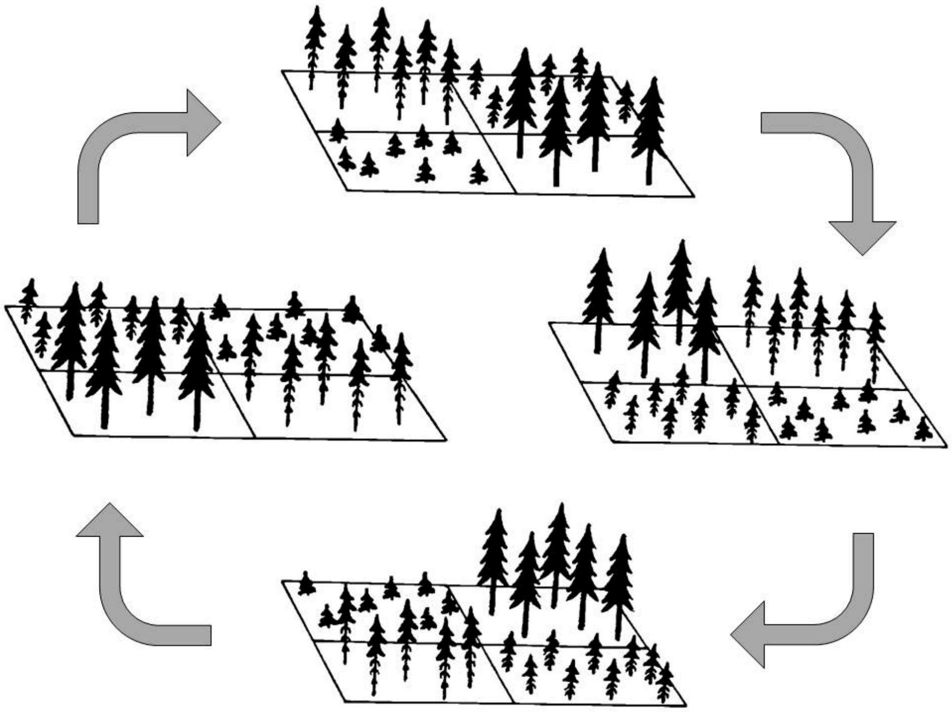


Figure 10 – Shifting mosaic concept of forest development. This concept suggests that an overall balance of patch births and deaths can produce a dynamic equilibrium at a landscape scale – there is fine-scale change, but the total number of young and old communities remains relatively constant at a broad scale in response to disturbance processes and plant succession.

This figure shows secondary succession for an idealized landscape consisting of four rectangular forest units. Start with the top diagram (unit) and follow the gray arrows in a clockwise direction – the mature stand (lower right grid) is eventually affected by a stand-replacing disturbance process, which transforms it to the stand initiation structural stage (these are the small trees in the lower right grid of the next diagram). The small trees grow and eventually reach a 'young middle-age' stage called stem exclusion (see diagram at the bottom). As growth continues, they reach a 'late middle-age' stage called understory reinitiation. When the trees develop to a point where they are once again mature (old forest), this 'circle of forest life' has been completed and we are once again back to the top diagram.

This figure illustrates that a balance exists between rates of disturbance and recovery, which maintains broad-scale diversity in a quasi-equilibrium pattern (Lertzman and Fall 1998), and that a forest landscape is a mosaic of successional communities whose composition and structure is determined by the types and severity of disturbance. It also demonstrates that in order to maintain a desired configuration of wildlife habitat through time, we must accept frequent and expected changes in existing vegetation conditions. We should also acknowledge and understand that wildlife habitat conditions will exist in different places and at different times over the long term, and that a properly functioning disturbance regime will successfully create and maintain a desired configuration of wildlife habitats, but it will do so as a shifting mosaic at a landscape scale.

4.4 Disturbance And Secondary Forest Succession

Scientists studying forests in the early 20th century believed that disturbance initiated a clearly identifiable and repeatable pattern of changes called succession, a process leading inevitably to an old-age stand. They believed that old stands persisted indefinitely and called them the climax stage (fig. 11). But ecologists had trouble finding examples of climax vegetation covering large areas. This caused them to wonder if the climax is not like "a phantom, always moving ahead into the future, and becoming visible for only relatively brief periods on small areas" (Graham 1941).

Once forest succession studies began in earnest, they showed that a single group of tree species is not predestined to inhabit an area, that recruitment of new trees into a forest often follows a disturbance event rather than occurring continuously, and that disturbance intensity often influences which tree species will dominate afterward (Bloomberg 1950; Cobb et al. 1993; Drury and Nisbet 1973; O'Hara 1995; O'Hara et al. 1996; Oliver 1980, 1981; Oliver and Larson 1996; Spurr 1952).

Scientific studies found that disturbance events, such as fires, windstorms, hurricanes, and landslides, operating over periods of 1 to 1,000 years, have an obvious influence on species richness (biodiversity) at local to landscape scales (Whittaker et al. 2001, Willis and Whittaker 2002). Studies examining secondary succession also found that plant diversity was consistently higher in forest stands managed using silvicultural treatments when compared with unmanaged reserves, and species richness was positively correlated with the amount of overstory canopy removed (high for clearcuts and shelterwoods; low for individual-tree selection) (Battles et al. 2001). As a result, species diversity is often linked to the frequency and intensity of disturbance (including activities designed to emulate disturbance processes) (Connell and Slatyer 1977, Huston 1979).

Daniel Botkin described the importance of forest succession this way: "The idea of succession is important to our understanding of nature and our management of natural resources; a problem in our management of forests has arisen because we have incorrectly projected the hypothetical endpoint of succession, believing that nature's melody leads to one final chord that sounds forever" (Botkin 1990a, p. 116).

After plant succession is initiated by disturbance, forests generally pass through at least four developmental phases – stand initiation, stem exclusion, understory reinitiation, and old forest (table 5, fig. 10; Oliver 1981, Oliver and Larson 1996). These developmental phases are related to important physiological processes.

Physiognomic Forest Succession

Mixed-species, single-cohort (even-aged) moist forests contain both early- and late-seral species (e.g., shade-intolerant and shade-tolerant species). In these forests, a mix of tree species generally initiate in the same year (or during the same discrete period) following a high-severity disturbance event such as wildfire or insect defoliation (Anderson and Romme 1991, Antos and Habeck 1981, Habeck and Mutch 1983, Oliver 1980). [An interesting note: research found that "mixtures of shade-intolerant and shade-tolerant species with different growth patterns may be more productive than single-species stands" (Chen et al. 2003).]

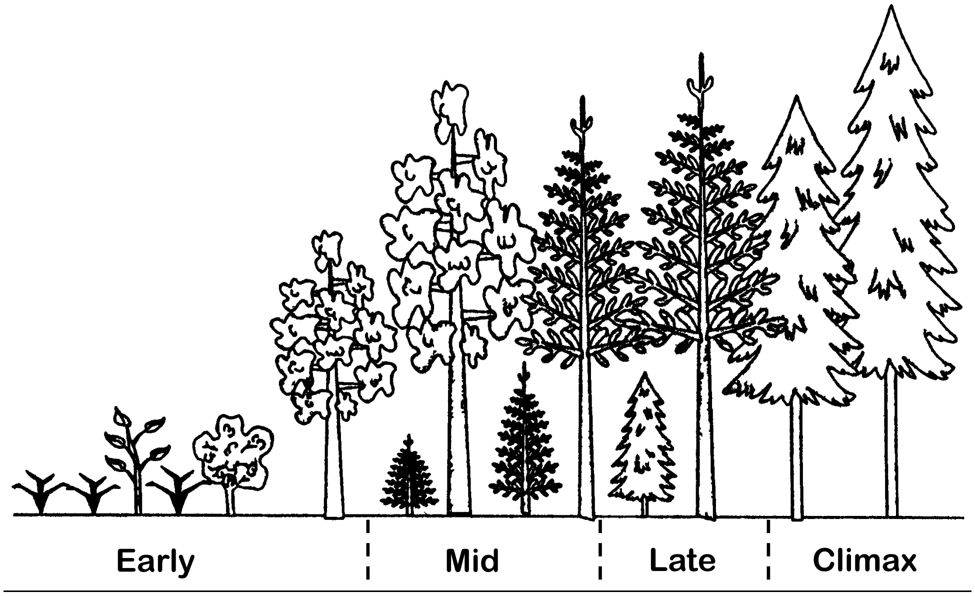


Figure 11 – Seral stages for a moist-forest plant association of the Blue Mountains. The series of stages shown in this diagram is called a sere. After a stand-initiating disturbance event such as crown fire or regeneration cutting, a new plant community gets established and it gradually transitions through a series of stages, progressing from a simpler, somewhat disorganized state (early-seral stage) to a relatively complex, highly organized state (the climax plant community). The early-seral stage is initially dominated by grasses, forbs, and shrubs (some ecologists refer to this non-tree phase as a very-early stage), but shade-intolerant, pioneer tree species also get established in early-seral communities. The mid-seral stage has a mix of species, with early-seral species (ponderosa pine above) and mid-seral species (Douglas-fir above) present in almost equal amounts. Late-seral stands have both mid-seral and late-seral tree species present (grand fir is the late-seral species above; table 19 provides the seral status for 7 primary conifers of moist forests). Although truly climax stands are relatively uncommon in our disturbance-influenced ecosystems, they feature a species composition where early- or mid-seral tree species are scarce or absent, and composition is dominated almost entirely by the late-seral species. Another reason for the rarity of truly climax forest stands in the Blue Mountains is that our early- and mid-seral tree species tend to live longer than the late-seral species (table 4).

Since tree species grow and develop at different rates, mixed stands gradually develop a stratified structure with fast-growing species such as western larch and lodgepole pine occupying the upper stratum, and slow-growing species such as Douglas-fir and grand fir relegated to a lower stratum (Cobb et al. 1993, McCune and Allen 1985, O'Hara 1995, Oliver 1980, Shea 1985, Veblen 1986) (Box 2, fig. 12).

Table 5: Description of forest structural stages.

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

Sources: Based on O'Hara et al. (1996), Oliver and Larson (1996), and Spies (1997). Other ecologists described similar process-based forest structural stages by using slightly different names (see Bormann and Likens 1979, Daubenmire and Daubenmire 1968, McCune and Allen 1985a, Peet and Christensen 1987, and Watt 1947).

Box 2. Development of Mixed-Species Forests on Moist Sites

An interesting study (Cobb et al. 1993) examined stand dynamics associated with mixed, single-cohort (even-aged) moist forests containing both early- and late-seral species. In these initial floristics communities, a mix of tree species regenerate in the same short, discrete period following a high-severity disturbance event such as stand-replacing wildfire or insect defoliation.

What was done. Six stands from the eastern Cascade Range in Washington (Wenatchee National Forest) were selected for intensive study and stand reconstruction. Selected stands had a mixed species composition, were single cohort, of natural origin (not plantations), exhibited no evidence of major disturbance since stand initiation (including thinning or partial cutting), were free of major insect or disease activity, and represented a range of stand ages, densities, and species composition. In a potential vegetation context, all stands were in the grand fir series. A circular 0.1-acre plot was established in a representative portion of each sample stand, and detailed mensurational data was then collected for each sample tree. A smaller subsample of plot trees was selected for stand reconstruction analyses. These trees were felled and then dissected to determine detailed height growth patterns (note that one of the six plots did not have dissection completed due to U.S. Forest Service restrictions precluding tree felling).

What was found. "The average ages of trees cored and dissected indicate that all species in all six stands were essentially even-aged, originating after stand replacement disturbances." But even on these relatively moist sites, the stand initiation phase following fire disturbance ranged "from less than 15 years on the Swauk plot to over 40 years for T-Creek" (Cobb et al. 1993, p. 546). All sample stands containing western larch and lodgepole pine had these species dominate since stand initiation, although minor fluctuations occurred from plot to plot in terms of which species was dominant at any particular time (fig. 12). When considering diameter-growth patterns, western larch sustained its diameter growth for a longer period than lodgepole pine because lodgepole pine tended to have a pronounced reduction in average diameter growth commencing about age 50.

When Douglas-fir and grand fir were both present, they experienced slower diameter and height growth than either western larch or lodgepole pine. However, when either of the mid- or late-seral species was present without the other, its diameter-growth rate would approach the rate for the slower of the two early-seral species, suggesting that interspecies competition was occurring within a stratum (i.e., grand fir and Douglas-fir were competing with each other in the lower stratum, and when the other species was not present, the diameter growth of a single late-seral species approached the rate for the slowest growing early-seral species).

Summary. "Successful mixed species management relies on a thorough understanding of mixed species stand development" (Cobb et al. 1993, p. 545). This study demonstrates that for moist forests of the eastern Cascade Range in Washington, early-seral species tend to grow faster in height than late-seral species when both groups are present in a single-cohort (initial floristics) forest stand. It also shows that height growth disparity is not necessarily related to tree age variation (fig. 12), demonstrating that tree age and size are not necessarily correlated – the larger (or taller) size of one species in a stand can create the false impression that it established prior to a smaller (or shorter) species. "The assumption that a vertically stratified stand has an uneven-aged structure would be incorrect for these stands, and could lead to some questionable silvicultural practices" (Cobb et al. 1993, p. 548).

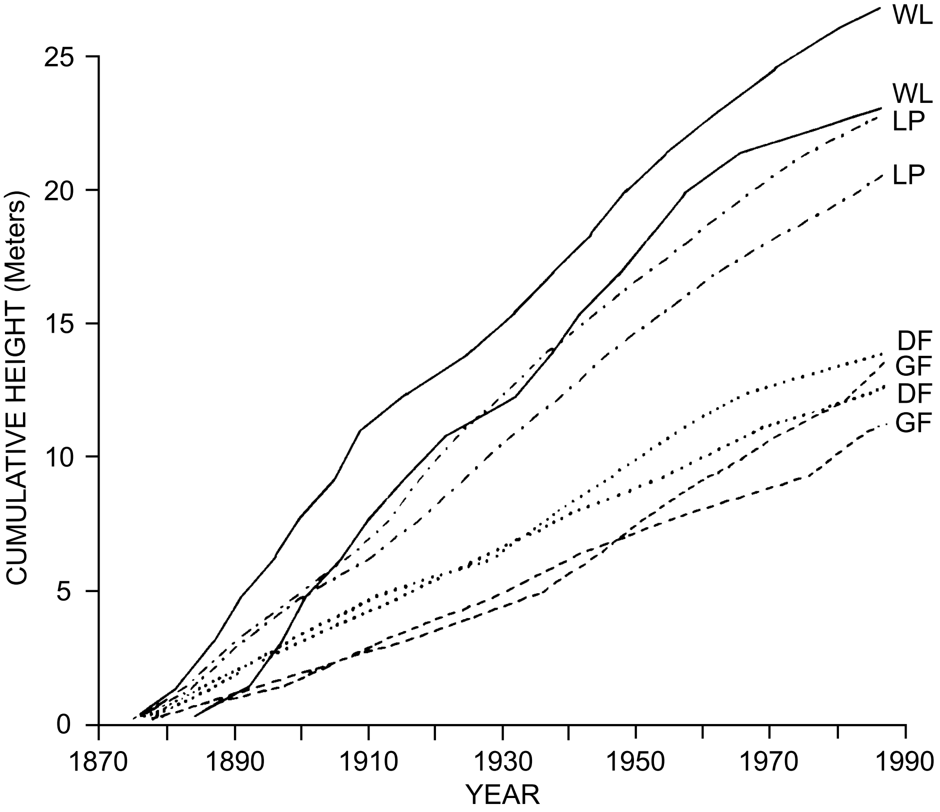


Figure 12 – Development of mixed-species, single-cohort stands on moist-forest sites (adapted from Cobb et al. 1993). Different tree species grow and develop at different rates. This figure shows that shade-intolerant species (western larch, WL, and lodgepole pine, LP) grow faster in height than shade-tolerant associates (grand fir, GF, and Douglas-fir, DF) when both groups are present in a single-cohort (even-aged) stand. The end result of this developmental progression is a multi-storied structure sometimes mistaken for an uneven-aged condition, even by silviculturists who neglect to use an increment borer to check whether a stratified structure is even- or uneven-aged (Bloomberg 1950, Larson 1986, O'Hara 1995, Oliver 1980). [Note: other differences between intolerant and tolerant tree species also contribute to this developmental pattern – intolerant species allocate more of their growth resources to stem and branch production, and have crowns with lower leaf area density, than for tolerant species (Canham et al. 1994), so an upper stratum of intolerant trees then allows more light transmission to a lower stratum of tolerant species.]

This multi-layered structure results from disparate height growth rates between species (Johnson and Fryer 1989, Veblen 1986). Early-seral trees grow faster in height than late-seral species when both groups are present in a single-cohort mixed stand (Cobb et al. 1993, Oliver 1980, Turner 1985, Veblen and Lorenz 1986). This developmental pattern shows that even if no species interactions occur during succession (such as density-dependent, intertree mortality related to self-thinning), life history traits alone (height growth rates in this case) can exert enough influence to directly affect the ultimate forest structure (Chapin et al. 2002).

Multi-layer forests are often assumed to be multi-cohort (e.g., uneven-aged), primarily because of attitudes associated with the relay floristics paradigm – the presumption is that overstory trees are older, and that they probably helped the younger understory cohort get established by casting shade or otherwise moderating environmental conditions (Oliver and Larson 1996).

This concept was historically referred to as the 'nurse crop' in forestry, and as facilitation in plant ecology (Holmgren et al. 1997). An example is that foresters often considered quaking aspen to be a nurse crop for conifers. Early work examining forest succession after fires in the central Rocky Mountains, however, seldom found examples where burns were occupied first by an aspen nurse crop and then subsequently by conifers. When age counts were made in mixed stands, aspen was found to have started simultaneously with, or later than, the conifers (Gardner 1905).

4.5 Using Active Management To Emulate Disturbance

"During the 1990s, a consensus emerged among ecologists that the most promising approach for conserving biological diversity and ecosystem function in managed forests was to emulate the disturbance processes that drive forest succession and dynamics in natural, unmanaged forests (Hunter 1993, Attiwill 1994, Christensen et al. 1996, Bergeron and Harvey 1997, Kohm and Franklin 1997). There is broad agreement that the challenge for forest managers is to develop techniques of resource extraction that do not cause local extinctions of species, reduce ecosystem productivity, or damage the processes that underlie the resiliency of forest ecosystems to disturbance" (Haeussler and Kneeshaw 2003) (Box 3).

When land is managed to produce water, wood, and other commodities desired by human society, an underlying assumption is that the greater the similarity between the effects of management activities and the effects of a native disturbance regime, the higher the probability that inherent ecological processes will continue without significant impairment (Aplet and Keeton 1999; DeLong and Tanner 1996; Hunter 1990, 1996, 1999; Mitchell et al. 2002; Roberts and Gilliam 1995; Rowe 1992).

Ecologist J. Stan Rowe made the following observation about the relationship between active management and disturbance patterns: "The only way to satisfy the popular demand for preserving biodiversity is to practice silviculture and harvesting within large regions in ways that maintain landscape ecosystems in mosaic patterns which approximate or mimic natural mosaic patterns" (quoted in Parminter 1998). [This idea is discussed further in the Disturbance Emulation portion of section 6.6.]

Once land managers acknowledge that disturbance is necessary for many kinds of ecosystems, and the frequency, kind, and degree of change is most important, not just change itself, they can better account for disturbance and its benefits (Botkin 1990a). Stand-level disturbance that might be viewed by some managers as negative at a small scale might be beneficial, if not integral, to ecosystem structure and function at a landscape scale (Kimmins 1996, 2004). Heterogeneous landscapes are inherently resilient due to species and functional diversity (Gunderson et al. 2010).

Box 3. The Natural Disturbance Model

Designing forest management practices that more closely emulate natural disturbances has been a theme of sustainable forest management. The natural disturbance model (NDM) is advocated by many forest ecologists, and it is implicitly promoted in a number of corporate and governmental forest management policies. Advocates of 'close-to-nature forestry' have promoted the concept in one form or another for more than a century, although debate has usually been couched as a preference between even-aged or uneven-aged silviculture. Social demand for near-natural forest management and continuous-cover forestry continues to grow, particularly in Europe (Malcolm et al. 2001).

If 'nature for nature's sake' was the only motivation for using NDM to guide forestry, we might be accused of committing a 'naturalistic fallacy,' a term coined by the philosopher G.E. Moore (1903) to refute the conclusion that there is inherent goodness in the state of nature. This can be considered an extension of David Hume's admonition that 'what is' never provides ethical guidance on 'what ought to be.' Perhaps the environmentally-aware children of the 1960s have blithely extended Barry Commoner's 'Third Law of Ecology' (i.e., 'Nature knows best'; Commoner 1971) too broadly, when its primary intent was to warn of the dangers of introducing unnatural chemicals into the environment (but nature-knows-best is basically a moral stance, not true 'biomimicry' like NDM).

Widespread adoption of NDM is not primarily an ethical or even aesthetic decision, although many of its proponents may be so motivated. Rather, its utility is an application of the 'coarse filter' approach to biodiversity conservation (Hunter 1991, Burton et al. 1992). The principles of forest sustainability, along with international commitments to biodiversity, mean that it is important to provide for viable populations of all species. But we can't even identify all forest-dependent species (when considering non-vascular plants, fungi, arthropods, unicellular organisms, etc.); we can't possibly develop hundreds of species-specific habitat management plans (this is the 'fine filter' approach for species at risk of becoming extirpated or endangered), and policies promoting any one species typically have negative effects on others (Cumming et al. 1994). So the prudent strategy is to maintain the composition, age-class distribution, landscape pattern, and stand-level structures under which indigenous species have persisted through history.

Habitats created by forest harvesting rather than disturbance processes such as wildfire can never completely duplicate their natural analogues, and we have no assurance they are optimal for any given species. Nevertheless, we know that natural habitats prevailing until the modern era were adequate for all the species left for us to manage today, so maintaining the closest possible similarity between wild and managed landscapes is an inherently conservative approach. Generating forest activity units with sinuous boundaries and islands of green trees can also serve social and aesthetic benefits that should not be discounted, and it may allow us to think of a landscape as largely wild and untamed. But for every person who likes to maintain an illusion of wilderness, there is another person who prefers a landscape of orderly domestication. Not only is it invalid to assert that natural patterns are right and artificial patterns are wrong (or vice versa), we can't automatically describe naturalistic patterns as 'pleasing to the eye' either. Whether landscapes sculpted according to NDM are 'good' or 'better' (for meeting human needs and sensibilities, protecting biodiversity, or achieving sustainable forest management) remains unsubstantiated until research and monitoring inform us of the tradeoffs and net effects associated with this approach (also see Chase 2001 and Namkoong 2005).

Note: this text was adapted almost verbatim from Adamowicz and Burton (2003, p. 58).

These scale-dependent relationships affirm that patterns and processes at one scale can be completely different, even contradictory, at broader or finer scales (this concept is referred to as emergent properties) because they are driven by different mechanisms (Levin 1992). Thus, landscape ecology research needs to be conducted by using multi-scale studies because a particular phenomenon cannot be explained solely by observations collected at a single scale (Sagarin and Pauchard 2010).

Ecological principles are the foundation of an approach based on the assumption that if the effects of silvicultural treatments closely emulate those of the historical disturbance regime, then the risk of losing native species and other ecosystem components is greatly reduced (Buse and Perera 2002, DeLong and Tanner 1996, Mitchell et al. 2002, Perera et al. 2004, Rowe 1992). "The act of emulation can range from one extreme of absolute mirroring or mimicry (i.e., repeating natural disturbances exactly), to another of more learning from nature (i.e., using nature only as a guide and selecting which aspects/effects to emulate)" (Buse and Perera 2002).

Research found that certain silvicultural treatments (clearcutting) may have less impact on nutrient cycling and other ecosystem functions than the disturbance processes they are designed to emulate (stand-initiating wildfire) (Clayton and Kennedy 1985, DeBell and Ralston 1970). Table 6 provides ideas for how silvicultural practices could be used to emulate disturbance processes (Dale et al. 1998, Smith 1977).

"Age distributions of forested landscapes (and the amount of old-age forests) are controlled by large landscape-scale processes and not by small-scale processes involved in forest management. Although there has been some shift towards more of a landscape perspective on management, there is still some problem with understanding and appreciating that these landscape level processes may be beyond any kind of management control" (Johnson et al. 1995).

The challenge is to integrate a suite of management treatments that effectively and appropriately emulate the native disturbance regime for moist-forest landscapes. Successfully meeting this challenge will produce a semblance of historic structure and composition – a desirable outcome not because the resulting condition is historic, but because it is sustainable (e.g., vigorous, self-perpetuating, and with characteristic levels of fire, insect, and disease activity) (Fiedler 2000b).

"The object of disturbance emulation is not to make a clearcut look like a fire, but rather to recognize and understand the differences between management and natural disturbance, and to use that knowledge to improve harvest methods so as to impair ecosystems and their processes as little as possible" (Thompson 2002). Legacies are especially important – "the importance of biological legacies is one reason that we should be careful in accepting management treatments as analogs for natural disturbances" (White et al. 1999).

Table 6: Silvicultural practices, and the disturbance processes they can emulate.

PRACTICE	PROCESSES BEING EMULATED
Clearcutting (with reserve trees)	• Stand replacing fires on sites where they are characteristic (e.g., on cold-forest and upper-elevation moist-forest sites). • Mountain pine beetle outbreaks in pure, even-aged forest of lodgepole pine. • Windthrow occurring in pure, even-aged forest.
Overstory Removal (with reserves)	• Western pine beetle in uneven-aged ponderosa pine forest. • Douglas-fir beetle in multi-layered Douglas-fir forest. • Spruce beetle in forests with spruce in the overstory and spruce and true firs in the understory (Veblen et al. 1991). • Mountain pine beetle in multi-layered lodgepole pine forest (especially with shade-tolerant species in the understory). • Subalpine fir mortality caused by balsam woolly adelgid. • Windthrow in subalpine forest with an overstory of lodgepole pine and an understory of spruce and fir (Veblen et al. 1989).
Salvage Cutting	• A reburn event in which a second fire consumes trees killed by the first burn, typically after they topple over. • A wildfire after insect-caused mortality – insects kill the trees, they topple over, and their stems are consumed by fire.
Selection Cutting	• Root disease in mature, mixed-species forest (emulated by group selection). • Dwarf-mistletoe parasitism in mature, mixed-species forest (emulated by individual-tree selection). • Lightning-caused tree mortality on upper-slope physiographic positions (emulated by individual-tree selection). • Senescence (tree death) caused by old age in mature forest.
Shelterwood or Seed-Tree Cutting	• Bark beetles in mature, even-aged, mixed-species forest. • Mixed-severity wildfire in mature, mixed-species forest. • Avalanche chutes on steep slopes in the subalpine zone (emulated by the strip variant of the shelterwood cutting method). • Douglas-fir tussock moth defoliation in mixed-conifer forest.
Thinning	• Surface fire in multi-layered forest of ponderosa pine, or in mixed forest of ponderosa pine and Douglas-fir/true firs. • Budworm damage in dense, even-aged, mixed-conifer forest. • Low-intensity fire in areas with a mixed-severity fire regime.
Understory Removal	• Budworm damage caused by the 'feeding ladder' effect in multi-layered forest of budworm-host species, where larvae rain down from taller trees onto shorter trees (see fig. 22). • Surface fires in mixed forest with fire-resistant overstory species (ponderosa pine, larch, Douglas-fir) and fire-susceptible understory species (lodgepole pine, true firs, spruce).

Sources/Notes: Silvicultural practices are defined in Helms (1998).

Natural processes and their silvicultural analogues can be grouped into two distinctly different categories: releasing disturbances that kill from the top down, and maintenance disturbances that kill from the bottom up (figs. 13 and 14). Thinning emulates natural processes that kill trees from the bottom up. Thus, thinning in moist forest to emulate the budworm feeding ladder effect (see fig. 22) supports this central axiom of ecological forestry: any manipulation of a forest ecosystem should mimic the native disturbance processes of a region, as they existed prior to extensive human alteration (Hunter 1999).

Should Managers Attempt To Emulate Disturbance?

Natural is a term used to describe something existing in nature, something that is either not artificial or entirely devoid of human influence (Egan and Howell 2001, Hunter 1996, Kay and Simmons 2002). This attitude of humans being unnatural is a fundamental precept of many federal laws and policies. The Wilderness Act of 1964, Public Law 88-577, is a good example because it defines wilderness as "an area where the earth and its community of life are untrammeled by man, where man himself is a visitor who does not remain" (Drury 1998).

The concept of divorcing humans from nature is inconsistent with archaeological evidence showing human habitation of interior Columbia River basin ecosystems for at least 15,000 years (Knudson 1980). Separating humans from nature offers no admission that the environment changes constantly, regardless of what humans do or don't do. And it fails to recognize that native people influenced ecosystems at various scales in many, although certainly not all, of the landscapes where contemporary management activities are occurring (Barrett 1980, Bonnicksen 2000a, Boyd 1999, Denevan 1992, Kay and Simmons 2002, Robbins 1997, Shinn 1980, Vale 2002).

The question of whether humans should intervene in natural processes is moot, since ecosystems around the world bear their indelible imprint (Barrett 1980, Botkin 1995, Knudson 1980, Stewart 1963). And divorcing humans from nature is inconsistent with the principles of ecosystem management (Christensen et al. 1996):

- An acknowledgment of humans as ecosystem components is explicit.
- Long-term sustainability is a fundamental value.
- Clear, operational goals are vitally important.
- Sound ecological models and understandings are evident.
- An appreciation of ecosystem complexity and interrelations is incorporated.
- Recognition of the dynamic character of ecosystems is apparent.
- An attention to context and scale is obvious.
- A commitment to adaptability and accountability is included.

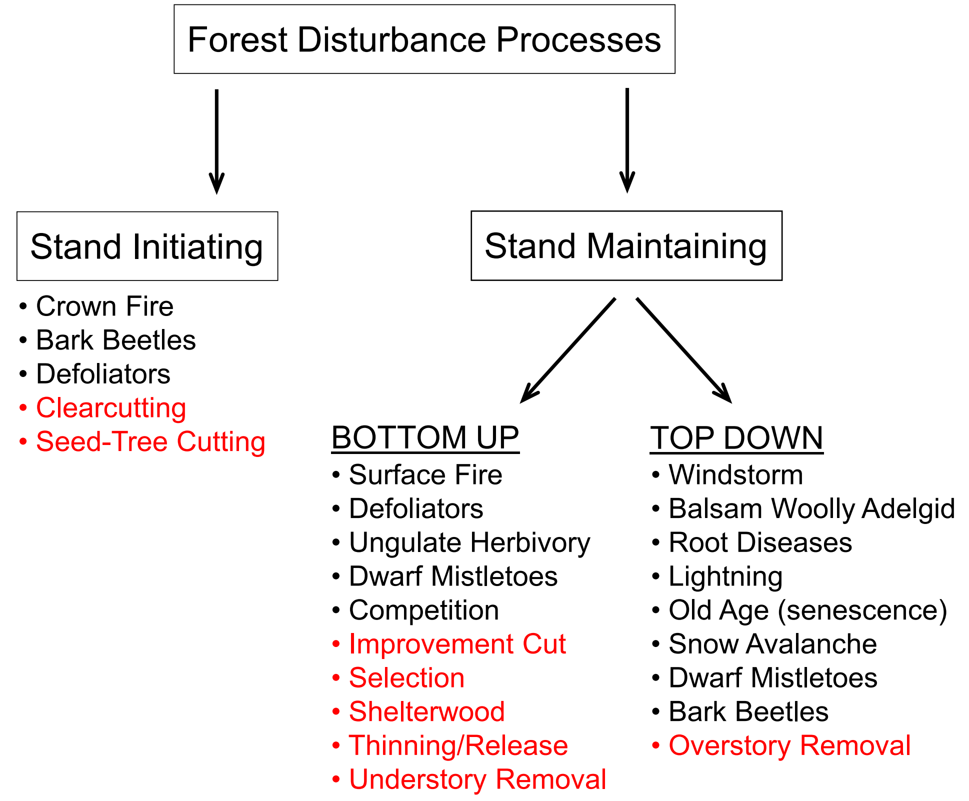


Figure 13 – Diagram classifying forest disturbance processes as either 'stand initiating' or 'stand maintaining' (or both for some processes), and then further assigning the stand-maintaining processes to either a 'bottom up' or a 'top down' functional group. Anthropogenic processes, shown in red text, were assigned to the functional group for which they are most compatible in a disturbance-emulation context.

Timber Harvest And Landscape Pattern

In some managed landscapes, inherent vegetation patterns have been altered by previous tree harvest practices, particularly regarding the effects of clearcutting (Franklin and Forman 1987). Often, the result of clearcutting was a visual pattern whose texture, form, line and color were out of scale with natural landscapes. [Natural landscapes are defined as those whose elements – texture, form, line, color, etc. – were produced by a disturbance regime free of influence from tree harvest, fire exclusion, and other anthropogenic factors.]

On national forest system lands in eastern Oregon and eastern Washington, the size of patches created by anthropogenic disturbances such as tree harvest is limited to no more than 40 acres by the National Forest Management Act of 1976 (Public Law 94-588). When considered from a landscape perspective, patches of 40 acres or less are often inconsistent with vegetation patterns resulting from the inherent disturbance regime (DeLong and Tanner 1996). In the DeLong and Tanner (1996) study, harvested patches had simpler boundaries and less remaining mature forest than burnt areas, and fire patches were more complex as they got bigger.

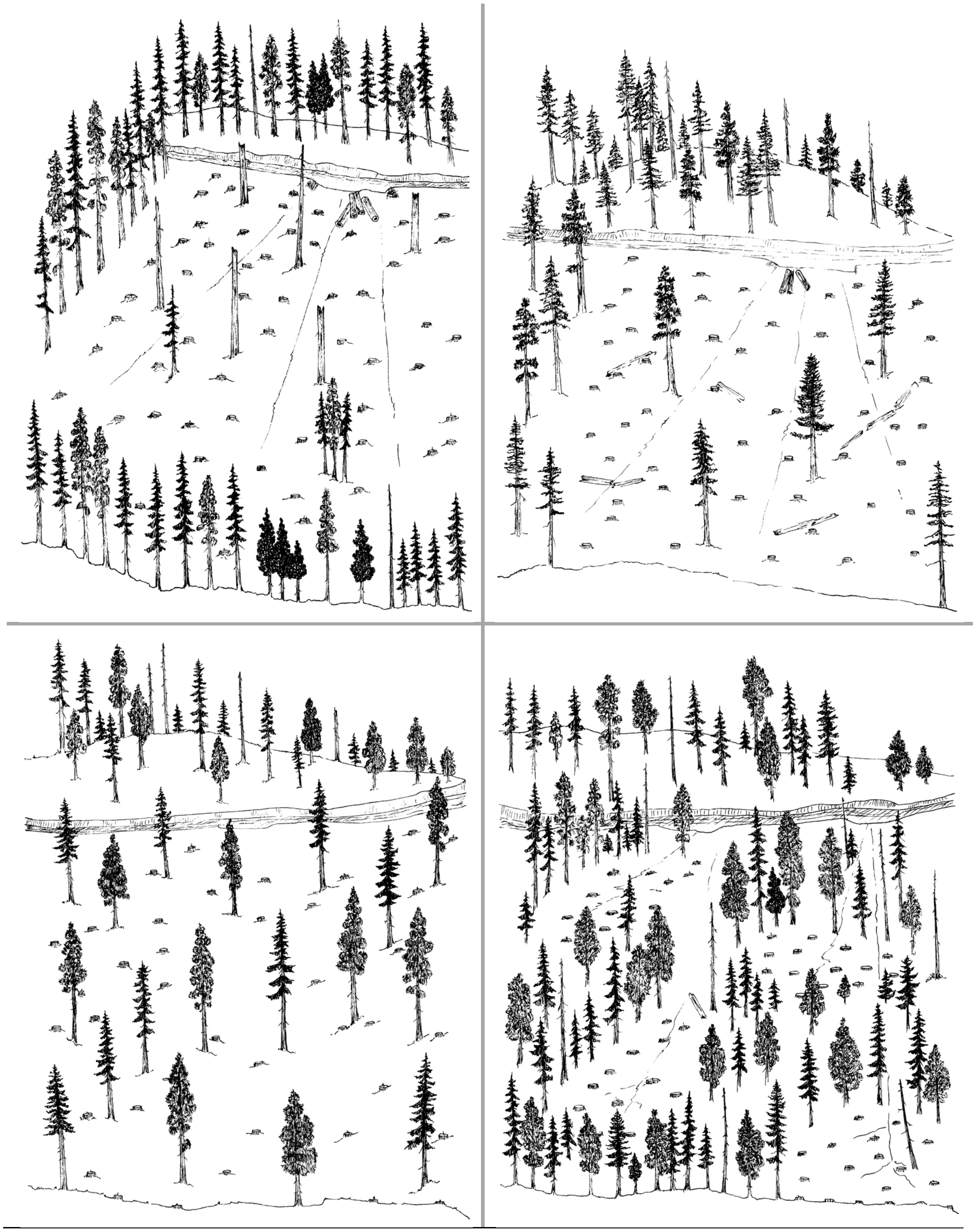


Figure 14 – Examples of stand initiating and stand maintaining silvicultural treatments (from Powell 1994; see fig. 13). Clearcutting with reserves (upper left) and seed-tree seed cuts (upper right) are examples of stand-initiating silvicultural treatments. Shelterwood cutting (lower left) and group selection (lower right) are examples of stand-maintaining silvicultural treatments. Traditional large-patch clearcutting removed all of the live trees and most of the dead wood from an area; clearcutting with reserves retains some live trees and more of the dead wood, both of which function as intergenerational ecosystem memory (e.g., biological legacies).

Historical patch sizes were analyzed for the Desolation Creek watershed, located within the North Fork John Day River subbasin. Table 7 summarizes patch characteristics for historical cover types and structural stages for the Desolation Creek watershed. It shows that Douglas-fir had the most cover type patches, followed by grand fir and then lodgepole pine. The largest patches were comprised of lodgepole pine, however, since they averaged 434 acres in size and ranged up to a maximum of 18,126 acres. In a landscape classification framework, lodgepole pine would be considered the matrix element for species composition (USDA Forest Service 1999).

Table 7 shows that the stand initiation structural stage had the most patches in the Desolation watershed in 1939 (excluding the nonforest patches (NF) that were ignored for forest structural stage purposes), followed by old forest single stratum and then young forest multi strata. The largest patches were those comprised of old forest (old forest multi strata and old forest single stratum), since they averaged 214-223 acres in size and ranged up to 6,271 acres. In a landscape classification framework, old forest would be considered the matrix element for forest structure.

The Desolation Creek watershed analysis, for example, suggests that limiting patch size to 40 acres is inappropriate for lodgepole pine and other ecosystem types adapted to episodic disturbance processes affecting wide areas (Stuart et al. 1989). Limiting patch sizes to 40 acres or less is particularly inappropriate if anthropogenic patches are intended to emulate the vegetation mosaic created by native disturbance processes (Buse and Perera 2002; also see table 6 and fig. 13) because table 7 shows that 40-acre patches are smaller than the mean value for many cover types and structural stages and, with one exception, 40 acres is smaller than the maximum patch size for every cover type and structural stage. This finding demonstrates that a 40-acre limitation is somewhat arbitrary and, while it may be well-meaning in terms of addressing public aversion to clearcutting (Bliss 2000) and even-aged management, it is not well aligned with disturbance emulation principles or concepts.

Future activities in managed landscapes could attempt to rehabilitate undesirable visual conditions by restoring a natural vegetation pattern, e.g., a pattern in synchrony with, or analogous to, one produced by the native disturbance regime (Perera et al. 2004). As an example, existing clearcut units could eventually be expanded and shaped in such a way as to approximate the pattern, juxtaposition, and size of patches created by historical occurrences of stand-initiating wildfire (Botkin 1990a, Keenan and Kimmins 1993, Urban et al. 1987).

This approach to land management has been referred to as 'prescriptive scaling.' Urban et al. (1987) described it this way in their article about landscape ecology: "Foresters use prescriptive scaling when they mimic natural disturbances with clearcuts, a practice that represents the collective wisdom of generations of foresters, who have found a successful clearcutting strategy through trial and error." [Note: this quote often elicits strong reaction from those who disagree with the 'wisdom' of clearcutting. But rather than endorsing clearcutting, the quote actually extols the value of passing 'lessons learned' from one generation of foresters to the next.]

Table 7: Patch analysis for the Desolation Creek watershed (1939 conditions).

	Patch Type	**Number of Patches**	**Minimum Patch Size**	**Average Patch Size**	**Maximum Patch Size**
			<————	Acres	————>
Cover Types	Subalpine fir	23	1	348	6,400
	Douglas-fir	85	3	152	3,087
	Engelmann spruce	10	3	48	273
	Lodgepole pine	59	1	434	18,126
	Ponderosa pine	35	4	103	1,421
	Western larch	22	5	68	502
	Grand fir	80	6	150	2,027
	Mixed composition*	20	12	119	471
	Nonforest	85	1	36	589
	Total	419			
Structural Stages	Nonforest	85	1	36	589
	Old forest multi strata	72	9	214	6,271
	Old forest single stratum	85	3	223	4,840
	Stem exclusion closed	69	1	183	2,815
	Stem exclusion open	3	19	24	31
	Stand initiation	117	1	63	838
	Understory reinitiation	12	11	98	423
	Young multi strata	73	1	149	2,009
	Total	516			

Sources/Notes: This analysis was based on information derived from interpretation of aerial photography acquired in 1939-40 (the first aerial photography flown for the Blue Mountains was completed in 1939-40). For the Desolation watershed, the Forest Service procured the photography from the National Archives, and it was provided to a well-experienced photo interpreter to characterize historical conditions. Private lands in the watershed were included in the analysis. Patch metrics were calculated by using spatial analysis software (Ager 1997).
* The mixed composition cover type includes forests where no single species reached 50% of the composition; other types have a predominance of the species for which they are named.

Forest Fragmentation

The loss of landscape connectivity, often described as habitat or forest fragmentation, is considered by some landscape ecologists to be among the greatest threats to biological diversity (Noss and Cooperrider 1994). It is generally recognized that fragmenting an area of contiguous forest into smaller patches diminishes both the quantity and quality of the remaining forest interior habitat. Fragmentation is also believed to increase the risk of some types of 'catastrophic' disturbance such as windthrow, fire, insects, pathogens, and landslides (Franklin and Forman 1987).

A major scientific conference was convened in Portland, Oregon to address forest fragmentation, including consideration of its consequences on wildlife populations in the western United States and Canada. A book containing 15 peer-reviewed chapters was published to summarize findings from the conference (Rochelle et al. 1999).

The conference discussed fragmentation in light of the theory of island biogeography, which considers forest patches as islands separated from each other by a sea of hostile habitat. Much of the ‘island biogeography’ paradigm was derived from research conducted in the midwestern and eastern United States in situations where forest habitat had been converted to agricultural or urban uses (Harris 1984, MacArthur and Wilson 2001).

Conference findings drew a clear distinction between forest fragmentation, where older and younger patches are juxtaposed on the landscape, and habitat loss, which occurs when forest is converted to a completely different land use (such as forest conversion for a WalMart store and its paved parking area). Key findings from a forest fragmentation conference included these items (Rochelle et al. 1999):

- Response of vertebrate populations to fragmentation differs, but for most species the effects of habitat loss are more significant than changes in habitat pattern.
- Some research suggests that vertebrate survival will be affected if the area of suitable habitat falls below a threshold of 20-30 percent.
- Negative effects on vertebrates occurring after forestland was converted to agricultural or suburban development are not apparent in western forests fragmented by timber production.
- Pacific Northwest forests were naturally fragmented by disturbances such as fire and defoliating insects; in drier forests of eastern Oregon and Washington, fire suppression has been de-fragmenting the indigenous patterns of fuel distribution and accumulation, thereby increasing the potential for large wildfires.
- Discussions of forest fragmentation are most meaningful if made in the context of historical landscape conditions.
- Connectivity of the landscape is not uniformly important to forest vertebrates because they vary in their gap-crossing abilities – there was little evidence that lack of connectivity is a threat in forests of the Pacific Northwest.
- There is little evidence from the Pacific Northwest to indicate the degree to which corridors affect animal movements – either limited by their absence, or enhanced by their presence.
- Bird abundance doesn’t change significantly in a forest stand from 40 years old to maturity; although some vertebrate species are restricted to old growth, there are no apparent old-growth dependent communities (groups of species) among amphibians, reptiles, birds or mammals (with the possible exception of bats).
- Leaving relatively small amounts of habitat structure (shrubs, snags, decaying wood, live conifers, and broadleaf trees) after tree harvest apparently makes the matrix area between patches more hospitable, suggesting that most late-successional species can be maintained in managed stands by retaining these structural elements (see figure 71 in section 6.5 later in this white paper).
- Riparian areas are often richer in species and more densely inhabited than are upland areas, but few forest-dwelling species are restricted to riparian areas.
- More vertebrate species are positively than negatively associated with edge habitat, although research has documented a few species that are consistently associated with forest interior habitat.

- The increased predation and nest parasitism observed when forest edge is surrounded by agricultural or suburban development are not evident when forest lands remain in forest use.
- There is no ideal patch size – management approaches that result in a variety of habitat conditions and patch sizes on the landscape will meet the needs of the greatest number of species.
- Forest managers in the Pacific Northwest should not blindly apply the results of research studies from other regions such as the eastern and central United States.
- Forest fragmentation can best be understood from the perspective of individual species – "a landscape is not inherently connected or fragmented; the same landscape may be both from the perspective of two different species."

Disturbance As A Management Template

Cissel et al. (1994) described a six-step process to analyze disturbance regimes at a landscape scale, and then use the results to generate management actions:

1. Assess historical and current disturbance regimes for terrestrial and aquatic ecosystems.
2. Integrate this information by using an appropriate mapping and/or narrative technique, and define a desired landscape condition and associated management approach for sub-areas, or strata, with similar disturbance regimes, potential vegetation, and human use patterns.
3. Project the management approach into the future by using a geographical information system; assume no natural disturbances, but allow for plant succession; model management activities that emulate the historical disturbance regime.
4. Analyze the resulting landscape pattern to see if adjustments are needed to meet established management objectives (current conditions, for example, may be outside the range of desired conditions).
5. Adjust the frequency, intensity, or location of future harvest units; change the amount or shape of reserves; allocate ecosystem restoration practices.
6. Identify management actions that will encourage development of the desired landscape condition.

4.6 Old Forest On Moist Upland Sites

Many people enjoy experiencing an old forest because of what they see and how it makes them feel (Hoffman et al. 1999). People tend to think of old forests as tranquil, unchanging places (fig. 15). "Because we are unable directly to sense slow changes and because we are even more limited in our abilities to interpret their cause-and-effect relations, processes acting over decades are hidden and reside in what is called the invisible present" (Magnuson 1990).

Figure 15 – Moist, old-forest stand in the upper Mottet Creek area of the Umatilla National Forest. This example shows a relatively dense overstory of grand fir, and an undergrowth dominated by low forbs (primarily ginger, twinflower, and darkwoods violet for this area). Note the sward of bracken immediately behind the large trees, and Sitka alder behind the bracken. In the lexicon of contemporary forest structural stages (table 5), the immediate foreground condition would classify as 'old forest single stratum.' The more typical old forest stage encountered on moist sites is 'old forest multi strata.'

Some moist forest areas, by virtue of their topographic position, soil type, or a combination of environmental and vegetation conditions, are less frequently affected by stand-replacing disturbances than the surrounding landscape. These areas may be thought of as semi-stable elements in a dynamic landscape because their environmental settings allow them to function as disturbance refugia. [But also note that disturbance and structure are synergistic, and have interacting feedback loops: a moist landscape dominated by old forest structure can have a different disturbance regime than a similar moist-forest landscape featuring young stands.]

Disturbance refugia are often associated with specific physiographic settings such as upper headwalls, the confluence of two stream channels, areas with perched water tables, and sheltered ravines or valley bottoms immediately adjacent to perennial streams (Camp et al. 1997, Romme and Knight 1981, Taylor and Skinner 1998). They typically differ from the surrounding landscape matrix in species composition, or in structural attributes such as tree height, stand density, or diameter distribution. Refugia may harbor plant and animal species that would otherwise be absent if an entire landscape was affected by the same disturbance regime.

Depending on the ecological context, fire was often the most influential disturbance process for matrix areas of the landscape. But disturbance refugia were most commonly affected by insects and diseases, thereby creating soft snags and other biotic components largely missing from the fire-influenced forest (Camp et al. 1997). The forest structure associated with moist-forest disturbance refugia typically consists of late-successional species occurring in multi-cohort, high-density stands (grand fir, Engelmann spruce, or subalpine fir forests with multiple canopy layers and large tree diameters; fig. 15). The structural condition associated with disturbance refugia was referred to historically as 'old growth' (fig. 16).

A common misconception is that old growth is synonymous with climax conditions (Box 4, and see figs. 11 and 16 for additional context). For forests of the interior Pacific Northwest, this generalization is seldom true because many old-forest stands feature an overstory with large-diameter trees of early- or mid-seral species (ponderosa pine, Douglas-fir, western larch, or western white pine; see table 4), although the understory may have a predominance of small-diameter, late-seral or climax tree species (Hall 1984, Hall et al. 1995).

As a result of fire suppression, ungulate herbivory, and selective timber harvest, there has been a pronounced shift in old-forest physiognomy – reflected by a shift from open ponderosa pine forest to closed pine and Douglas-fir stands for dry forests, and for moist forests, the transition has been from tall white pine-western larch stands to relatively short grand fir-spruce-subalpine fir stands (Graham et al. 1999, Hessburg et al. 1999b, Lehmkuhl et al. 1994, Powell 1994).

Is Old Forest Reduced From Characteristic Levels?

Many definitions of old forest include the onset of death in the initial tree cohort, accompanied by down wood accumulation, creation of canopy gaps, and growth of understory trees into the overstory. Studies of old growth forest often focused on stand-level attributes (number and type of trees per acre, down logs, number of canopy layers, etc.) rather than examining the geographical representation of old growth at a landscape scale (Franklin et al. 1981). This lack of a landscape perspective may have contributed to a common belief that much of North America's forest was old growth prior to Euro-American settlement (Botkin 1990a, Harris 1984, Johnson et al. 1995, Kimmins 1996, Sprugel 1991).

Recent bioregional assessments suggest that old forest structure is reduced in most of eastern Oregon and Washington from Euro-American settlement era levels (Hessburg et al. 1999b, Lehmkuhl et al. 1994). Quantified information about historical amounts and distribution of old forests is scarce, however, so it is unclear whether a reduction has actually occurred and, if so, what its magnitude might be.

Camp et al. (1997) found that old forests historically occupied a relatively small proportion of the landscape in the Wenatchee Mountains of eastern Washington (9 to 16 percent). Peter Lesica (1996) calculated presettlement old-growth percentages for numerous forest ecosystem types in the northern Rocky Mountains; his estimates were based on a 'negative exponential model' of fire frequency (fig. 17; Johnson and Gutsell 1994, Van Wagner 1978).

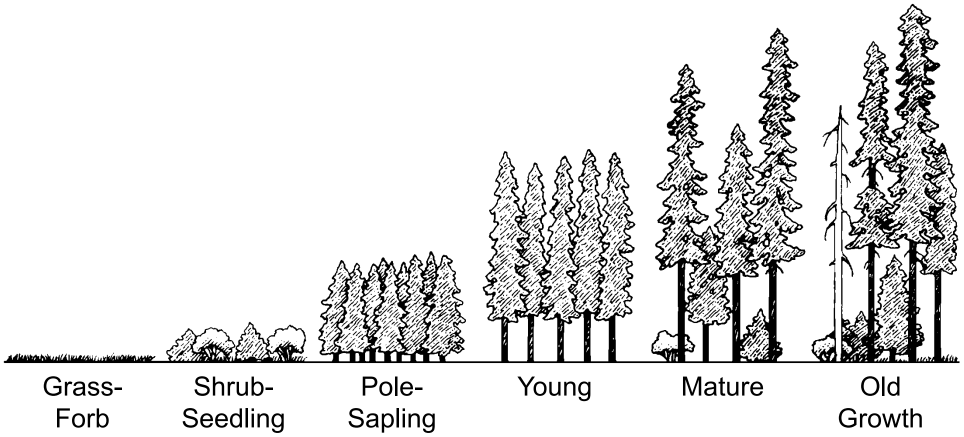


Figure 16 – Physiognomic succession in a coniferous forest (adapted from Thomas et al. 1979). After a stand-initiating disturbance event, a new forest develops by passing through successive and predictable stages. These changes result in a progression of vertical structure (physiognomy) through time. This figure shows a six-stage chronosequence, beginning with a grass-forb stage and culminating in a late-seral, old-growth stage. Compare this system for classifying forest structure with contemporary approaches involving criteria based on physiological characteristics and stand dynamics (see table 5).

Since the negative exponential model is an age-class or cohort-analysis tool, it is most compatible with biophysical environments where stand-replacing disturbance processes tend to replace old patches and initiate new ones (Johnson et al. 1995). Areas influenced primarily by surface fire or another stand-maintaining disturbance regime are incompatible with the negative-exponential model.

But, this concept could also be integrated into an overall planning strategy for moist forest – negative exponential modeling could be effective for the stand-replacing component of a disturbance regime, and another analytical procedure could reflect secondary (and non-replacing) influences such as budworm defoliation.

Even though negative exponential modeling is generally perceived as being incompatible with stand-maintaining disturbance regimes, including those most influential in shaping the characteristics of dry-forest ecosystems, it is interesting that Hessburg et al. (2007) found the patch size distributions for low-, mixed-, and high-severity fires in dry forests to fit a negative exponential distribution.

The negative exponential model also describes tree diameter distributions well (Rubin et al. 2006), and when used in a fire planning context, it predicts that about two-thirds of a region will have stands younger than the fire cycle, and about one-third will have stands that are older than the fire cycle (Rowe 1983).

Peter Lesica's negative-exponential analysis for the northern Rockies showed that old forests might have occupied between 4 and 46 percent of the landscape, varying with geographical location and potential vegetation setting (Lesica 1996).

Box 4. Will Similar Old Growth Develop on Similar Sites?

How much of the floristic composition of an area is due to abiotic site factors versus historical factors? A fascinating study examined this question by sampling moist, old growth forests in western Montana. The objective of the study was "to measure the relative influence of site factors and historical factors on species composition of old-growth forests in a narrow range of environments" (McCune and Allen 1985).

What was done. Mesic, low-elevation forests were sampled in 11 east-west trending canyons along the east slope of the Bitterroot Range in western Montana. With one exception (a canyon with previous timber harvest was excluded from consideration), all canyons were adjacent, located on similar parent material (granitic geology of the Idaho batholith formation), and similar in size, shape, and stream discharge.

Species composition data was collected from transects in each sample canyon and then stratified by tree size class, providing information about vertical forest structure and "species composition in relation to stand age and topographic position." Detailed environmental data (moisture, nutrients, and light) were collected in 13 old-growth stands distributed among 4 of the 11 sample canyons. Old-growth stands had a predominance of grand fir, western redcedar, or Pacific yew. Ordination and regression analyses were used to evaluate the central question: "how well do the site variables account for compositional differences?"

What was found. Since the sampled areas had no evidence of recent disturbance and were sites with low environmental variability – each old-growth stand occurred on granitic parent material and experienced a similar macro-climatic regime – it was assumed that site factors and vegetation were largely independent of each other. Surprisingly, McCune and Allen (1985) found that only 10 percent of the floristic composition associated with old-growth forests in the Bitterroot canyons was due to abiotic site factors, while the other 90 percent was presumably related to historical (stochastic) factors, or to unmeasured variables or imprecise measurements.

These results have interesting implications because a central tenet of plant ecology is convergence – once plant succession progresses to the point of a climax community, there should be a high degree of floristic similarity from one area to another because climax vegetation supposedly represents a consistent expression of abiotic site factors (Whittaker 1953). The cone-shaped model of plant succession is based on convergence because its broad base consists of early-seral communities, the smaller middle portion is comprised of mid-seral communities, and the narrow pointed top represents the ultimate stage of secondary plant succession – a single climax plant community (Steele 1984).

Perhaps the most interesting implication is that "climax communities are used as standards for comparison: standards for comparing sites on both local and regional scales, standards of relative stability for community dynamics, and standards for evaluating the effects of disturbance and pollution by humans." But "if similar sites need not develop similar communities, then we should be cautious in using climax vegetation as a standard for comparisons that rest upon the assumption that composition at climax is solely determined by site characteristics" (McCune and Allen 1985, p. 368, 375).

Summary. McCune and Allen's (1985) work provides compelling evidence of the dominant role that chance (stochasticity) plays in vegetation development, and it demonstrates that plant succession can have multiple endpoints. It also suggests that initial, post-disturbance composition affects subsequent plant succession, and that the influence of initial composition can still be apparent even centuries after disturbance occurred.

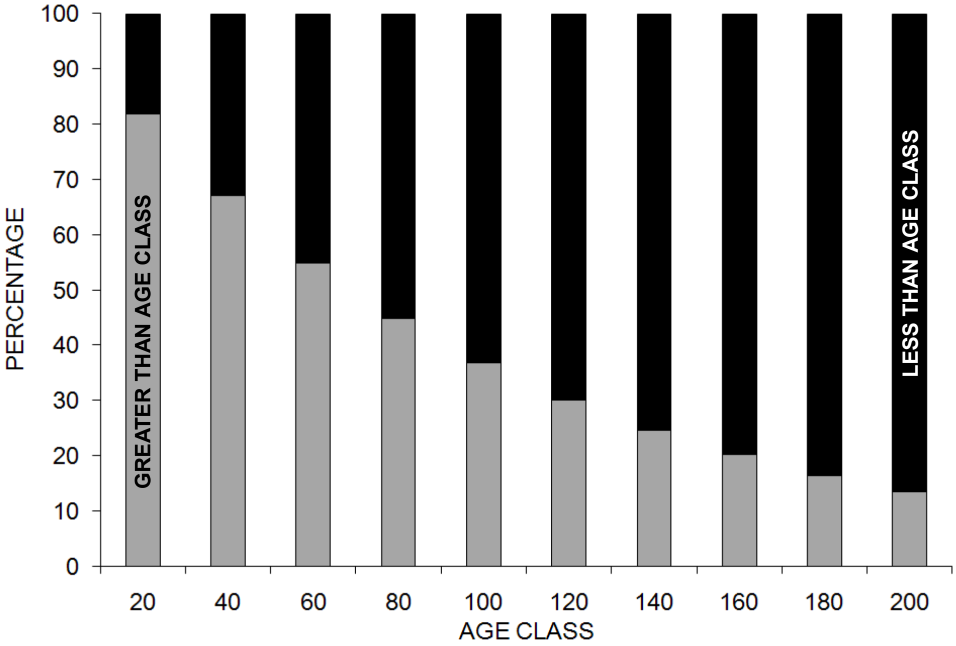


Figure 17 – Cumulative age distribution for a stand-initiating disturbance interval of 100 years (adapted from British Columbia Ministry of Forests 1995b).

Once the return interval for stand-initiating disturbance is known, a projected age-class distribution can be calculated. When using the negative exponential model shown here, it is assumed that the probability of disturbance is mostly independent of forest age, so it is most appropriate for forests where fires are stand replacing and fire return intervals are influenced more by broad-scale climatic trends than by fuel accumulation (such as fire regime III and IV sites in the Blue Mountains) (Johnson and Gutsell 1994).

The gray portion of a bar shows the landscape percentage that is greater than the age class; the black portion shows the percentage less than the age class. For the 20-year age class, for example, about 82% of a landscape influenced primarily by stand-initiating disturbance occurring with a 100-year return interval would support forests greater than 20 years of age; 18% of the landscape would have forests less than 20 years old.

But as a general 'rule of thumb' for the negative exponential model, one-third of the stands will be older than the stand-replacing interval, and two-thirds are younger. If attempting to mimic this age distribution, we need to provide old forest, but not 'so old' (e.g., > 200 yr) as to be unnatural and highly susceptible to uncharacteristic insect or disease levels (DeLong 2011).

4.7 Summary: Disturbance Principles

Disturbances are fundamentally important in controlling landscape pattern and ecological function for moist-forest ecosystems. Peter White (1987) described seven general principles regarding disturbance regimes and their effects:

1. **Disturbances occur at a variety of temporal and spatial scales** – disturbances act on both small and large spatial scales, and can influence ecosystem components for short or long time periods.
2. **Disturbances affect many levels of biological organization** – disturbances can disrupt ecosystem development, return areas to earlier seral stages, and change terrestrial or aquatic habitat mosaics.
3. **Disturbance regimes vary, both regionally and within any particular landscape** – disturbance regimes commonly vary with potential vegetation setting: 80% of individual tree mortality in cold-forest types is typically related to wind, whereas 20% or less may be wind-related in dry-forest types.
4. **Disturbances overlay environmental gradients, both influencing and being influenced by those gradients** – disturbances behave differently depending on which environmental setting they interact with (dry versus moist versus cold). Non-specific disturbances, including many abiotic agents (wind), may occur in every biophysical environment, but act differently in each one.
5. **Disturbances interact and can be synergistic** (see fig. 6) – mountain pine beetles attack lodgepole pines weakened by a slow-moving fungus *(Phaeolus schweinitzii)*, the resulting dead trees eventually contribute to a forest fire, fire causes basal wounds on surviving trees, the wounds provide new infection points for the fungus, and the cycle begins again (Geiszler et al. 1980).
6. **Disturbances may result from feedback between the state of a plant community and its vulnerability to disturbance** – the composition and structure of dry-site forests, for example, was a direct result of frequent surface fires, which ensured a sustainable ecosystem by reducing its vulnerability to crown fire (by limiting development of a multi-layered structure and its associated ladder fuel), and by preventing succession to the late-seral tree species (on dry-forest sites, the late-seral species are not fire tolerant, and they would not be sustainable in a frequent-fire environment).
7. **Disturbances produce variability in communities** – disturbances promote plant and animal diversity by influencing species composition, age, edge characteristics, and distribution of structural stages across the landscape.

> “There is no doubt that some birds, especially woodpeckers and many owls, do indeed require old stands to nest in. But old-growth forest is not ecological nirvana. It is just one of many different habitats that different species of birds have adapted to. A world of nothing but old-growth forest is just as much a biological desert as a world lacking in old-growth” (DeGraaf 1991).

5. DISTURBANCE PROCESSES

SECTION CONTENTS

Section Summary

Many disturbance agents influence moist forests of the Blue Mountains; ten important agents or processes are discussed in this section.

1. Defoliating insects (western spruce budworm, Douglas-fir tussock moth, and larch casebearer) are especially important. Western spruce budworm (section 5.2) and tussock moth (section 5.3) are described individually because they are so influential. Figure 18 shows that the areal extent of defoliating insects has been greater than any other disturbance process affecting the Umatilla National Forest.
2. Dwarf mistletoes affect three common tree species of the moist-forest zone: Douglas-fir, western larch, and lodgepole pine.
3. Stem decay caused by the Indian paint fungus is common in older moist forests with a predominance of grand fir; it also provides important ecosystem services for cavity-nesting bird species and other species utilizing tree-stem holes and hollows.
4. Root diseases (Annosus, Armillaria, and laminated) are widespread, site-level organisms of moist-forest ecosystems; their impact can be minimized by how moist forests are actively managed (by incorporating a species preference during thinning, etc.).
5. Bark beetles exert strong control on the species composition of moist forests because they tend to be host-specific: mountain pine beetle affects lodgepole pine primarily, Douglas-fir beetle affects Douglas-fir, and fir engraver affects grand fir primarily.
6. As would be expected, wildfire has an important influence on the composition, structure, and development of moist forests. Moist forests generally have a mixed-severity fire regime (FR); three fire subregimes are also recognized: relatively high frequency (FR IIIa), moderate frequency (FR IIIb), and low frequency (FR IIIc).
7. Wind is an important disturbance process, particularly for moist-forest sites with tall, shallow-rooted species (Engelmann spruce) and fine-textured soils such as ash caps. Widespread, episodic wind events are common in the northern Blue Mountains.
8. Timber harvest occurs in the moist-forest zone, as it has since the mid-1950s. Clear-felling was the predominant cutting method until the early 1990s, and it contributed to establishment of dense, uniform, young stands now requiring active intervention to restore an ecologically appropriate composition, structure, and density.
9. Non-native invasive species have also played a role in moist forests, with white pine blister rust (affecting western white and other five-needled pines) and balsam woolly adelgid (affecting subalpine fir) being notable examples.
10. Climate change is included as a disturbance agent because it is already influencing other disturbance processes, including wildfire occurrence and severity, defoliating insect impact, and bark-beetle distribution and severity. Due to drought and extreme weather events, climate change also causes direct changes in vegetation conditions.

Section 5 discusses how active management practices can be used to influence stand and forest susceptibility to defoliating insects, how active management strategies could incorporate the ecological role of disturbance processes (including enhancement of birds and other natural enemies of defoliating insects), and how density management practices can be implemented to not only reduce stand susceptibility to bark beetles, but also to increase a tree's capacity to produce plant defensive chemicals such as phenols and terpenes.

The last fifty years saw a period of rapid change for millions of forested acres in the Blue Mountains (Gast et al. 1991, Wickman 1992). Some of this change was related to normal forest growth and maturation, but much of it resulted from disturbance agents such as mountain pine beetle, western spruce budworm, Douglas-fir beetle, and fir engraver (fig. 18). Disturbance agents can be assigned to two broad categories depending on whether they are derived from living or non-living components of the environment: biotic and abiotic. And weather and climatic influences (including drought), an important abiotic factor, often interacts synergistically with disturbance agents from both categories.

Biotic disturbance agents of notable importance for moist-forest sites include defoliating insects such as spruce budworm and tussock moth; bark beetles such as Douglas-fir beetle, fir engraver, and mountain pine beetle; dwarf mistletoes affecting Douglas-fir, western larch, and lodgepole pine; three root diseases; one stem decay fungus; and non-native, invasive forest pests. Two abiotic agents exert an important influence on moist forest sites: wildfire and wind. And finally, two anthropogenic activities include timber harvest (major influence historically), and climate change. Each disturbance agent is discussed individually in this section of the white paper.

5.1 Defoliating Insects

When considering moist upland forests in the Blue Mountains, three defoliating insects have been most influential: western spruce budworm, Douglas-fir tussock moth, and larch casebearer (Torgersen 2001). Of the three, budworm and tussock moth have been most active and widely distributed, particularly at a landscape scale; for this reason, they will be discussed in individual sections.

At the stand scale, population increases of western spruce budworm and Douglas-fir tussock moth seem to be related to certain site conditions associated with development of nutrient deficiencies or imbalances in their host trees (Stoszek 1988b). Moisture and nutrient stresses cause changes in the synthesis of chemical defense compounds in host foliage, perhaps functioning as a predisposing or preconditioning factor for insect success, and they are considered to be important factors contributing to population increases for both insects (Carlson et al. 1985, Cates et al. 1983).

The magnitude of defoliator outbreaks can often be traced to man-caused changes in forest conditions that tipped the biological balance to favor these insects. Direct suppression using chemical or biological insecticides can reduce damage for short periods, for example, but it has not been found to change the overall course of an outbreak (Carlson et al. 1983, Torgersen et al. 1995).

Disturbance in forests affected by defoliating insects happens when periodic insect population oscillations increase to a point where they have a measurable impact on the composition, structure, or function of the ecosystem. Due to the periodicity of defoliator outbreaks, repeated infestations have been considered to have relatively similar ecological effects as recurring, non-lethal fire (Swetnam and Lynch 1993, Yazvenko and Rapport 1997).

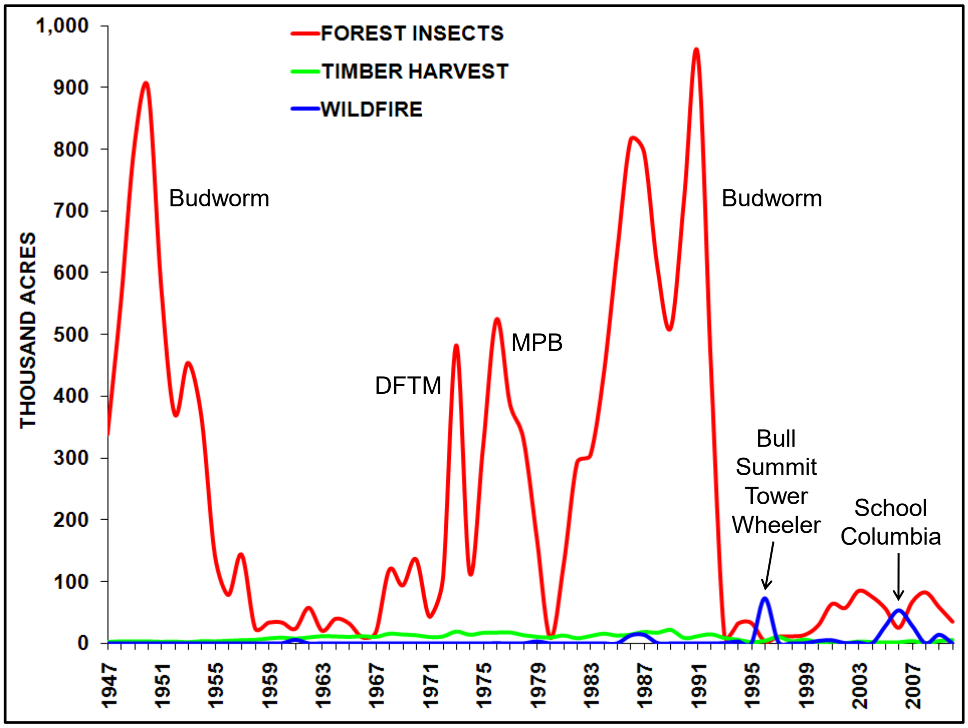


Figure 18 – Disturbance process trends from 1947 to 2010 for the Umatilla National Forest. Although fire hazard has recently received the most attention and funding, this disturbance history for the Umatilla National Forest suggests that forest conditions are influenced more by forest insects than by wildfire or timber harvest. This chart shows two major outbreaks of western spruce budworm (1944-1958 and 1980-1992), and outbreaks of mountain pine beetle (MPB) and Douglas-fir tussock moth (DFTM) in the early to mid 1970s. Six of the Forest's largest recent fires are also shown (Bull, Summit, Tower, Wheeler Point, School, and Columbia Complex). It has been noted that native forest insects are the greatest forces of change in North American forests – the forest area affected by insects and pathogens in the United States is approximately 45 times greater than that affected by fire, with an economic impact almost five times as great (Dale et al. 2001). Trends shown here for the Umatilla National Forest parallel broader trends for the United States (i.e., much more area is affected by insects than by wildfire).

Interactions between insects and other disturbance processes are common – "bark beetles transmit pathogens between trees, and pathogens reduce trees' ability to resist beetles" (White et al. 1999). The issue of whether an interaction exists between insects and fire has been controversial, but research suggests that an interaction can occur (Li et al. 2003). Several studies from central Canada, for example, indicate that spruce budworm activity tends to precede fire disturbance by about four years (Bergeron and Leduc 1998, Fleming et al. 2002).

Outbreaks of western spruce budworm or Douglas-fir tussock moth tend to be:

1. Spatially synchronized (across all of western North America for budworm), and temporally periodic (periods of high impact followed by periods of low impact, across wide areas). Regional climate trends (e.g., drought) can function as a synchronizing factor for budworm outbreaks (Campbell et al. 2005).

2. Predictably selective, which results in different post-disturbance legacies than are created by abiotic agents such as fire or wind because defoliators tend to have very specific biological relationships with their host plants (Holling 1992).

5.2 Western Spruce Budworm

Western Spruce Budworm *(Choristoneura occidentalis)*

TREE SPECIES AFFECTED: Douglas-fir, grand fir, subalpine fir, and Engelmann spruce.

DAMAGE CAUSED: Reduced tree growth following defoliation; reduced seed production from budworm feeding on cones, and from decreased tree vigor; top killing; tree mortality.

IMPACTS AND EFFECTS: Cyclic outbreaks are common – the Blue Mountains were most recently affected from 1944 to 1959, from 1980 to 1992, and from 2001 to 2012.

CONTROL MEASURES: Spray with insecticides; favor early-seral, non-host trees during stand management; create single-story stands when dealing with pure host type; stocking-level control; create host/non-host mosaics over large areas. Much of the 1944-59 outbreak was sprayed with DDT; portions of the 1980-92 outbreak were sprayed with carbaryl or Bt.

FOREST INSECT AND DISEASE LEAFLET: No. 53 (Fellin and Dewey 1986).

Western spruce budworm, a native insect, is usually an unobtrusive inhabitant of mixed-conifer forests containing grand fir, Douglas-fir, Engelmann spruce, or subalpine fir. But occasionally, after weather and other environmental conditions become ideal for its growth and survival, budworm populations increase very rapidly during what is called an outbreak.

The Blue Mountains have experienced two budworm outbreaks during the last 70 years – the most recent one extending from 1980 to 1992, and an earlier outbreak discovered in its early stages on the Heppner Ranger District in 1944 (Whiteside 1956). The earlier outbreak lasted until 1958. The 1980-92 outbreak had a broader areal extent in the Blue Mountains than the 1944-58 outbreak (fig. 19).

Wickman et al. (1994) identified possible spruce budworm outbreaks by analyzing the ring patterns of host and nonhost trees growing on the same site. They inferred that a severe outbreak was present in the northern Blue Mountains between 1898 and 1909; other outbreaks apparently occurred during 1870-78, 1838-42, 1822-30 and 1775-85. Outbreaks were also active during these periods in northern New Mexico (Swetnam and Lynch 1993), suggesting that historical budworm irruptions may have been synchronous over wide areas of western North America (as they also seem to be recently).

During a budworm outbreak, many of which last for a decade or more, trees and stands containing susceptible host species are affected in a variety of ways:

1. **Defoliation.** An adult western spruce budworm is a small, grayish moth. It causes tree damage in larval (caterpillar) stages of its life cycle. Damage occurs when the larvae feed on a tree's foliage and buds in a process called defoliation.
2. **Topkilling.** Topkilling occurs when budworm larvae consume the foliage and buds near the top of a tree, eventually killing that part of its stem (fig. 20). Even

if defoliation is not severe enough to kill the top, height growth is reduced or eliminated during each year that feeding occurs.

3. **Direct Tree Mortality.** Direct mortality occurs when budworm defoliation occurs long enough to kill trees outright. During the 1980-1992 Blue Mountains outbreak, budworm had a pronounced effect on tree mortality in mixed-conifer forests of the Blue Mountains (fig 21).

   Note that budworm-impact studies have consistently shown that tree mortality is selective (Carlson and Wulf 1989, Hummel and Agee 2003, Powell 1994) – short trees are killed more often by budworm than tall ones, primarily due to a phenomenon called the ‘feeding ladder effect’ (fig. 22).
4. **Indirect Tree Mortality.** Indirect mortality related to budworm feeding occurs in two situations: a) when defoliation stress predisposes trees to be killed by bark beetles, drought, and other causes; and b) when trees that were stressed before defoliation began – such as Douglas-firs infected with dwarf mistletoe or any host tree weakened by suppression or drought – died as a result of additional stress caused by budworm defoliation.
5. **Reduced Tree Growth.** Growth reductions occur after a tree loses enough foliage to inhibit photosynthesis, the biochemical process trees use to fix carbon and produce wood fiber. Growth increment usually starts declining during the year that budworm feeding begins, and is progressively reduced with each additional year of defoliation. After an outbreak collapses, defoliated trees require several years to replace their missing foliage. During this recovery period, growth increment slowly recovers to its pre-outbreak level.
6. **Reduced Wood Volume Production, and Persistent Tree Damage.** Wood production declines as defoliated trees experience reduced diameter and height growth. Forks, crook, sweep, and other stem deformities (indicators of previous stem wounds) occur after topkilling causes lateral branches to turn upward and attempt to gain dominance as a tree’s new top (fig. 23).

When Powell (1994) analyzed a recent budworm outbreak for the Blue Mountains, he found that the area of budworm-caused defoliation was about three times greater at the height of the 1980-92 outbreak than at a comparable point in the previous outbreak (1944-58). Increases in budworm habitat (e.g., mixed-conifer forest) over time were believed to be an important reason for the increase in area defoliated by western spruce budworm (figs. 18 and 19).

Between 1936 and 1980, the area of mixed-conifer forest (budworm host habitat) more than doubled for the central and southern Blue Mountains, with a corresponding decline in the acreage of ponderosa pine forest (non-host habitat) (fig. 24). Also, budworm host habitat became more susceptible over time because on average, each acre of mixed-conifer forest supported a higher density of host trees in 1989 than it did in 1968, particularly for grand fir (Powell 1994).

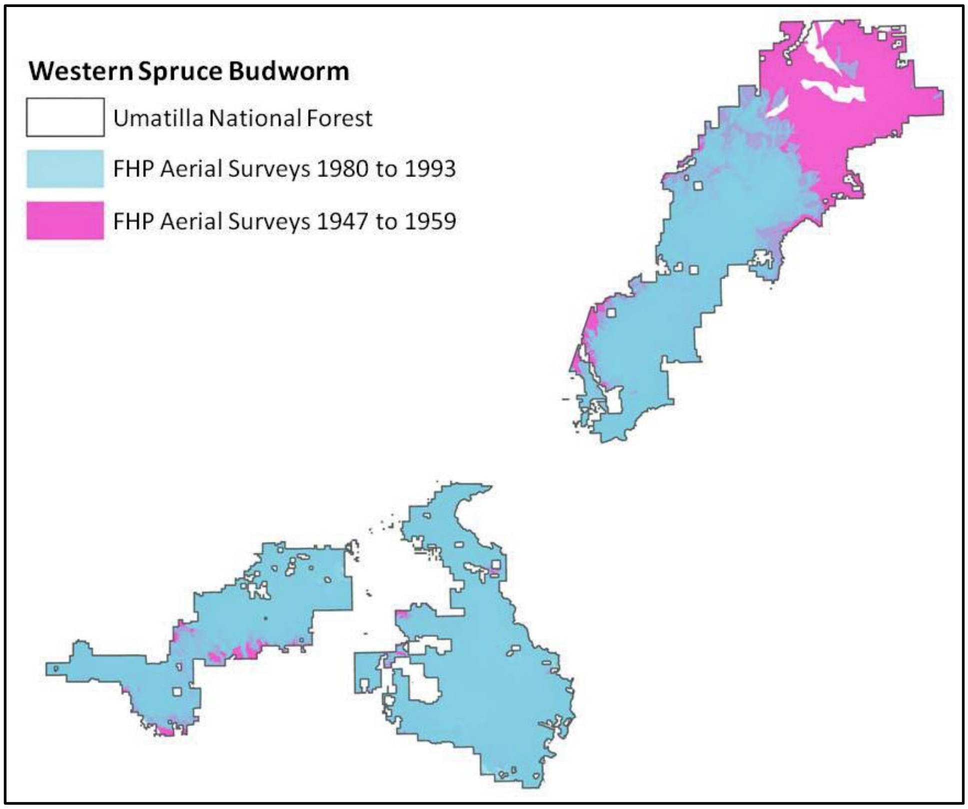


Figure 19 – Distribution of western spruce budworm impact for two recent outbreaks on the Umatilla National Forest in the northern Blue Mountains (this information was derived from aerial survey 'sketch maps' completed annually in the Pacific Northwest Region of the US Forest Service since 1947). Note that the most recent outbreak (1980-92) covered more area than the earlier outbreak, and that budworm is so ubiquitous that very little of the 1.4 million acre Umatilla National Forest was unaffected during either of the outbreaks (and also note that the 1944-58 outbreak was more extensive than is suggested by this map because the transparency settings (in GIS) for the 1980-92 outbreak do not allow as much of the 1944-58 outbreak to 'show through' the 1980-92 outbreak as was anticipated).

Reacting To Spruce Budworm Defoliation

Managing budworm populations typically involves four approaches, used independently or in concert. The four approaches are described below:

1. Taking no direct action; letting the outbreak run its natural course (this is the no-action approach).
2. Suppressing budworm populations by applying an insecticide (this is the direct suppression approach).
3. Reducing budworm impacts by improving tree and stand vigor (this is the stand vigor approach).
4. Avoiding budworm impacts by modifying the insect's habitat to remove or reduce the amount of host type (this is the avoidance approach).

Figure 20 – When budworm defoliation persists near the top of a host tree's crown, topkill is often the result. Many of the small Douglas-firs shown here have dead tops caused by budworm defoliation. As budworm larvae feed in the upper part of a tree, they also destroy most of the developing cones, reducing seed production and natural regeneration capacity. Dead tops with a basal diameter of three inches or more are likely to result in stem decay (Ferrell and Scharpf 1982). Topkill is also responsible for stem deformity (fork, crook, and sweep) affecting a tree's merchantable (usable) volume for lumber-type wood products (see fig. 23). In addition to topkill, defoliation caused by herbivorous insects can cause significant and sustained tree-growth reductions and ultimately tree mortality (fig. 21); mortality tends to be a function of both the magnitude and persistence of defoliation.

Figure 21 – Numerous studies demonstrate that western spruce budworm is a symptom rather than the problem (from Powell 1994). When this herbivorous insect causes tree mortality, as shown here from Starr Ridge in the central Blue Mountains (Malheur National Forest), the underlying cause can almost always be traced to an uncharacteristic species composition and structure – a common situation influencing defoliating insects throughout the western United States (Anderson et al. 1987, Hadley and Veblen 1993, Monnig and Byler 1992, Sampson and Adams 1994, Shepherd 1994, Sloan 1998, Wickman 1992, Williams et al. 1980). "In the late 1970s, much of the old-growth ponderosa pine in this area was removed by using helicopters. The residual stands were comprised mostly of Douglas-firs, many of which were infected with dwarf mistletoe. Note that budworm did not provide a uniform, evenly-distributed thinning – trees were often killed in groups" (Powell 1994). Gray trees reflect older mortality caused by either budworm or Douglas-fir beetle; most of the red-topped trees (recent mortality) were killed by Douglas-fir beetle responding to tree stress caused by budworm defoliation or dwarf mistletoe parasitism. Although the 1980-92 budworm outbreak seemed especially severe as it was progressing, tree mortality did not ultimately occur across large areas, and much budworm host-type in the Blue Mountains now awaits the next outbreak.

A no-action approach can be appropriate when budworm damage is not expected to affect accomplishment of management objectives for an area, such as attaining its desired future condition. Land managers have recently been considering a no-action approach more often than previously, especially as more information becomes available about the important roles that budworm and other insects play in the ecosystem (see table 8 later in this section).

But during the 1980-1992 outbreak in the Blue Mountains, many managers selected the direct suppression option (see fig. 25 later in this section) because budworm impact was more severe and extensive than they could accept.

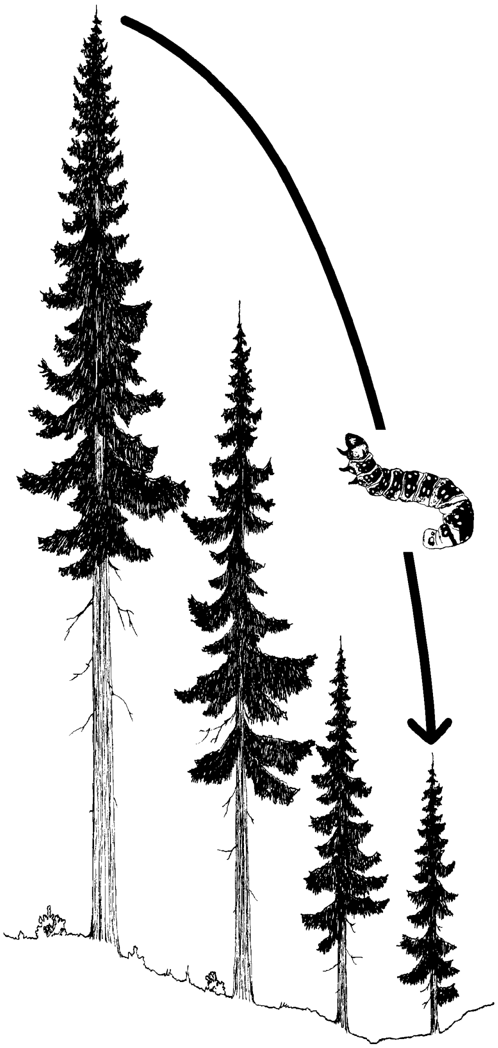


Figure 22 – Budworm feeding ladder effect in multi-layer forests (from Powell 1994). In stands comprised of two or more canopy layers, budworm impacts are usually concentrated on the smaller trees (the lower canopy layers). Some budworm larvae disperse from taller trees (the upper layers) and are intercepted by trees in the lower layers before reaching the forest floor, where they would have died from exposure or been preyed upon by carpenter and thatching ants, jumping spiders, carabid beetles, many bird species, small mammals, and other natural enemies (table 8 describes common budworm predators). The feeding-ladder effect tends to concentrate budworm impact (defoliation, topkilling, tree mortality) on the lower stand layers. This means that episodic defoliator outbreaks (both for budworm and tussock moth) may have served a similar and complementary role to that of surface fire by keeping live-fuel accumulations low and delaying conflagration wildfires or stand-replacing defoliator outbreaks (Holling 1981, McCullough et al. 1998). Note that the budworm larva is much enlarged (thankfully☺).

During the 1944-58 budworm outbreak in the Blue Mountains, almost 4.7 million acres were sprayed (some acres more than once) with an insecticide called DDT (Dolph 1980) (fig. 25). DDT became popular after two early successes – it was used to control Douglas-fir tussock moth in northern Idaho (Carlson et al. 1983) and the northern Blue Mountains (Wickman et al. 1973) in 1947, and it was used for experimental suppression of spruce budworm populations on a portion of the Heppner Ranger District and adjacent industrial lands (Kinzua) in 1948 (Eaton et al. 1949).

DDT, a powerful chemical applied in a fuel oil diluent, affected many organisms besides budworm (Hunter 1990). Although DDT was commonly used during the 1944-58 outbreak (fig. 25), land managers eventually realized that direct suppression provided little long-term benefit because it failed to address the fundamental issue – host-type habitat for western spruce budworm and Douglas-fir tussock moth proliferated throughout the western United States (Carolin and Coulter 1971, Fellin 1983, Williams et al. 1980). Because of its environmental persistence and the broad spectrum of organisms affected by it, DDT was banned for all uses in 1972.

Figure 23 – After budworm populations collapse and defoliation ceases, there are still long-term implications from an outbreak (from Powell 1994). One frequent consequence of budworm-induced topkill is stem deformity, such as the large, deep fork in this Douglas-fir. This tree was tallied during the 1980-92 outbreak, and the deep fork was probably a result of the previous outbreak in 1944-58. Research found that stem deformities (forks, crooks, sweeps, etc.) seldom result in volume losses exceeding 5% (Ferrell and Scharpf 1982), although it would certainly be higher for a fork like this one. Not only are host trees with large topkills more susceptible to stem deformity, but they also have higher risk of stem decay (Filip and Schmitt 1990). These deformities demonstrate that host trees and budworm can coexist for long time periods; overstory Douglas-firs have survived seven outbreaks spanning three centuries (Campbell et al. 2005).

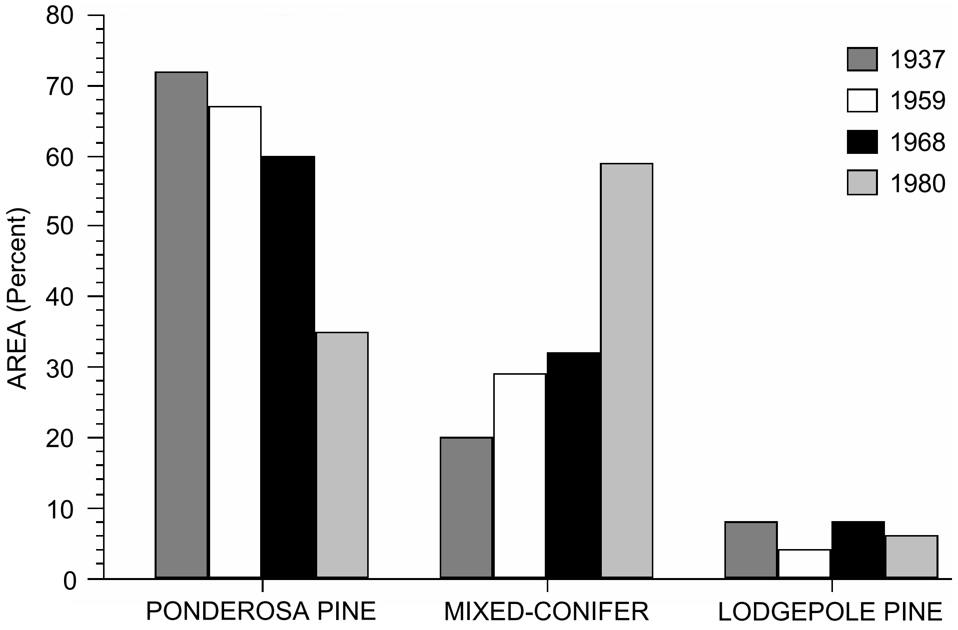


Figure 24 – Change in forest cover types for the Malheur National Forest, 1937-80 (from Powell 1994). Ponderosa pine forest declined by more than half between 1937 and 1980, the mixed-conifer type increased by an equivalent amount during this period, and the lodgepole pine type remained relatively constant. This figure shows that mixed-conifer forest – prime habitat for defoliating insects – increased by 195% between 1937 and 1980. The increase in mixed-conifer forest was an important reason for the unprecedented magnitude of a Blue Mountains budworm outbreak between 1980 and 1992.

During the 1980-92 budworm outbreak in the Blue Mountains, many direct suppression projects were also completed. These projects involved applying a chemical or biological insecticide, on areas of varying size, with an objective of suppressing budworm populations to non-damaging levels for the remainder of the outbreak (Powell 1994, Torgersen et al. 1995).

A chemical called carbaryl was applied for most general-forest situations, whereas another chemical called mexacarbate was used near streams and for sensitive environmental settings. Beginning with an experimental application in 1983, the insecticide used for all subsequent projects was a natural bacterium called *Bacillus thuringiensis* (B.t.). B.t. was always mixed with water, rather than the diesel oil diluent used with carbaryl (Sevin©).

By the mid 1980s, B.t. was the insecticide of choice because of its low risk to the environment and human health. Unlike DDT and similar broad-spectrum insecticides, B.t. directly affects a narrow range of organisms – only butterflies and moths in the Lepidoptera insect order are killed. B.t. is similar to the toxins contained in some spider and snake venoms in that it is cytolytic – after ingestion, it causes cells in the guts of susceptible insects to rupture and disintegrate (Ware 1989).

Figure 25 – Trailing plumes of insecticide, an airplane applies DDT and diesel oil diluent to moist-forest stands being defoliated by western spruce budworm in the northern Blue Mountains in June of 1951 (from Powell 1994). The direct suppression approach (applying DDT from fixed-wing aircraft) was used four times on the Umatilla National Forest during the 1944-58 spruce budworm outbreak (and it was also used in other years, and on the other two national forests, during the 1944-58 outbreak).

During the 1980-92 budworm outbreak, more benign chemical insecticides (when compared with DDT) came into widespread use, and eventually a narrow-spectrum biological insecticide (B.t.) was adopted for all direct-suppression situations. Widespread adoption of the direct suppression tactic was eventually abandoned after land managers recognized it did not address the underlying cause of budworm outbreaks (Torgersen et al. 1995) – host-type habitat for defoliating insects proliferated throughout much of the western United States (Carolin and Coulter 1971, Fellin 1983, Torgersen et al. 1995, Williams et al. 1980).

A switch to B.t. did not occur because previous insecticides (carbaryl, acephate, and mexacarbate) were banned, as was true when DDT was abandoned in the early 1970s – it reflected an increasing awareness that chemical insecticides may have had adverse environmental consequences. By using B.t., land managers could maintain more of the pretreatment arthropod diversity (jumping spiders, birds, thatch ants, etc.; see table 8) than was possible with carbaryl, acephate, or mexacarbate.

For a variety of reasons, the "suppressing budworm populations to non-damaging levels for the remainder of the outbreak" objective was not achieved by using the direct-suppression approach (e.g., spraying insecticide) (Torgersen et al. 1995). One possible reason is that insecticides were not applied to all of the area with visible defoliation, so that budworms in untreated areas could reinvade the treated units. Another reason is that an insecticide application does not eradicate budworm or its impacts – it typically suppresses budworm populations and associated tree impact for a period of two or three years at best.

Direct control methods – spraying a chemical or bacterial insecticide onto host-tree foliage – can be effective but short-term solutions to insect herbivory (Torgersen et al. 1995). But the direct-control methods don't deal with the most important underlying issue: much of the dry and moist mixed-conifer forest types in the Blue Mountains have high levels of tree and stand susceptibility to defoliating insects (Powell 1994, Wickman 1992). If stands and forests are highly susceptible, only a relatively small proportion of a defoliator's population must survive from one generation to the next to allow expansion from endemic to outbreak levels, particularly in conjunction with favorable weather conditions.

Budworm Susceptibility Considerations

Certain weather conditions have been observed to favor spruce budworm increases, particularly on a year-to-year basis within a longer outbreak cycle. Research in central British Columbia suggests that the combination of an early spring following a winter with low precipitation amounts, and then normal spring air temperatures, fosters successful budworm emergence, while potentially compromising the host trees' cabability to defend against budworm feeding (Campbell et al. 2005).

It is interesting that drought also affects moist-forest susceptibility to defoliating insects – foliage quality in drought-stressed trees is more favorable to budworm development, and it also tends to promote insect survival (Parks 1993, Mattson and Haack 1987). Note that drought in this context is related to both causes described above: a lack of precipitation (climatic drought), or reduced soil moisture caused by high levels of intertree competition (physiological drought). Stressed trees in dense stands, where they experience density-dependent, intertree mortality caused by self-thinning, or in stands on poor sites (warm and dry), produce foliage that is chemically better suited to budworm development (Carlson et al. 1985, Cates et al. 1983).

Other studies have noted a correlation between abiotic (physiographic) factors and budworm susceptibility – "other analysis indicated that WB [western budworm] defoliations are heavier in host stands in the seral stage of succession, in stands on southwesterly aspects, on ridgetops and upper slopes, on grand fir and subalpine fir habitat types, and in WB host-dominated stands. True fir species sustained heavier defoliation than Douglas-fir or spruce" (Stoszek and Mika 1984, p. 147). In addition, mature forest stands have been observed to sustain substantially higher budworm-caused tree mortality than immature stands (MacLean 1980).

Differences in the nutritional quality of Douglas-fir foliage explains why some individual trees are more resistant or susceptible than others to budworm. Susceptible trees had lower levels of nitrogen and sugars than resistant trees. Susceptible trees had a greater proportion of total terpenes that were monoterpenes, while resistant trees had a greater percentage of oxygenated monoterpenes. Bud burst and shoot expansion were delayed for resistant trees versus susceptible trees; resistant trees also accumulated more radial growth over the past 25 years, implying higher vigor levels than susceptible trees (Chen et al. 2001, 2002; Clancy et al. 1993).

Additional Information

Note that dozens of literature references about western spruce budworm and its disturbance ecology are provided in the References section of this white paper. Synthesis reports about western spruce budworm and possible management responses to its impact (defoliation, topkilling, mortality) are provided by Brookes et al. (1985, 1987), and an insect and disease leaflet (Fellin and Dewey 1982). Gast et al. (1991) and Powell (1994) describe tree and stand effects for the 1980-1992 spruce budworm outbreak in the Blue Mountains; Powell (1994) provides detailed information about budworm-caused impact on the Malheur National Forest in the central and southern Blue Mountains. British Columbia's Defoliator Management Guidebook (British Columbia Ministry of Forests 1995c) is also a useful spruce budworm reference.

5.3 Douglas-fir Tussock Moth

Douglas-fir Tussock Moth *(Orgyia pseudotsugata)*

TREE SPECIES AFFECTED: Douglas-fir and grand fir.

DAMAGE CAUSED: Severe top-killing; reduced radial growth; tree mortality.

IMPACTS AND EFFECTS: Cyclic – populations increase about every 9-11 years, building quickly over 2-4 years to outbreak levels, and then disappearing. Recorded outbreaks occurred in 1928-29 (near Seneca, OR), 1937-39 (80,000 acres near Rudio Mountain north of Dayville, OR), 1947-48 (1,500 acres near Snow Mountain and Gold Hill in the southern Blue Mountains), 1963-65 (almost 65,000 acres near Antelope Mountain and King Mountain in the southern Blue Mountains), 1972-74 (almost a million acres in the northern Blue Mountains), and 1999-2004 (more than 75,000 acres on the Umatilla NF, including the Indian Creek and Bologna Basin areas at Heppner RD; see fig. 27).

CONTROL MEASURES: Spray with insecticides; favor non-host species and mosaics of host and non-host stands; stocking-level control, particularly for dryer sites. Portions of the 1963-65 and 1972-74 outbreaks were suppressed using DDT, a chemical insecticide.

FOREST INSECT AND DISEASE LEAFLET: No. 86 (Wickman et al. 1981).

Douglas-fir tussock moth is a native insect of the Blue Mountains; it was not introduced from elsewhere like larch casebearer, white pine blister rust, or balsam woolly adelgid. Tussock moth defoliates host trees from the top down, killing trees outright or setting them up for future attack by bark beetles such as Douglas-fir beetle or fir engraver. The northern Blue Mountains have experienced at least three tussock moth outbreaks during the last 70 years (fig. 26).

Douglas-fir tussock moth is an important defoliator of Douglas-fir, grand fir, and white fir in western North America. Tussock moth infestations are usually short-lived – populations build quickly and collapse over a 3-4 period (Wickman 1973).

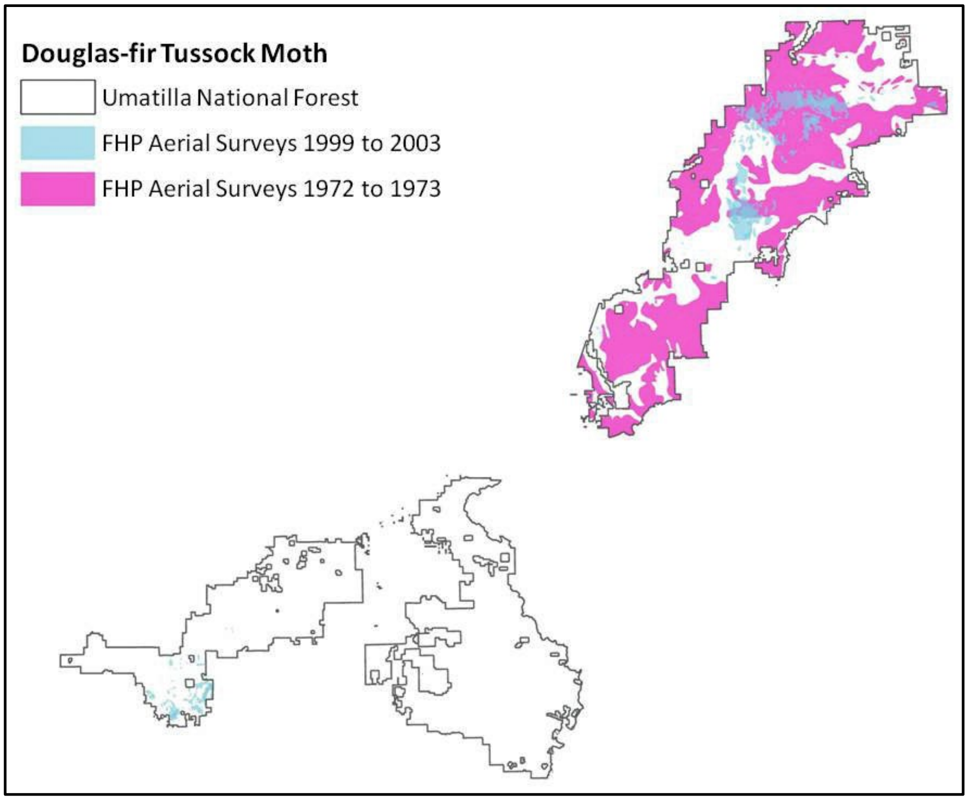


Figure 26 – Tussock moth distribution for two recent outbreaks on the Umatilla National Forest in the northern Blue Mountains (derived from aerial survey sketch maps completed annually in the Pacific Northwest Region of the US Forest Service since 1947). The 1970s tussock moth outbreak was the largest one ever recorded for North America (Brookes and Campbell 1978); it caused extensive impact on northern half of the Umatilla National Forest. A smaller outbreak is depicted from late 1990s and early 2000s. This map does not include outbreaks from mid 1940s (northern Blue Mountains near Troy) or mid 1960s (southern Blue Mountains near Burns). [Note: when comparing tussock moth and budworm outbreaks (figs. 19 and 26), northern half of Umatilla NF sustained less budworm damage than southern half (see fig. 19), but the opposite situation was true for tussock moth.]

Tussock-moth larvae, which hatch soon after bud burst, can severely defoliate host trees within 10 to 12 weeks (Wickman et al. 1981). Such defoliation may cause tree mortality, but most often results in temporary growth losses, top-killing, or increased susceptibility to bark beetles and other secondary insects or diseases (Kulman 1971).

Tussock moth populations are cyclic, rising on average about every 9 years in the western U.S. Not every population peak results in an outbreak – in northeastern Oregon, it appears that an outbreak coincides with every second or third population peak. Outbreaks occurred somewhere in the Blue Mountains in 1928-29, 1937-39, 1946-48, 1963-65, 1972-74, 1992-93, and 2000-01.

Dendrochronology studies found that tussock moth has been active in the Blues for as long as their mixed-conifer forest habitat has been available (note that dendrochronology is the analysis of tree cores to infer climatic cycles, fire occurrence, insect outbreaks, and other natural events with a strong temporal signal). Dendrochronology analysis, for example, indicates that Douglas-fir tussock moth might have defoliated mixed-conifer stands in the Drumhill Ridge area (Walla Walla Ranger District) during 1843-45, 1852-54, and in 1875 (Wickman et al. 1994).

Mid-1940s Tussock Moth Outbreak

The Umatilla National Forest has maps and written records for two of the early Douglas-fir tussock moth outbreaks. The first one was discovered on August 20, 1946 when a pine beetle survey crew was cruising a check plot near Troy, Oregon. After the survey crew reported the outbreak, an entomologist was dispatched from the Forest Insect Laboratory in Portland to examine the situation.

When entomologist Walter J. Buckhorn visited the Troy area on September 18-19, 1946, he found 10,000 to 12,000 acres of mixed-conifer forest had been defoliated by tussock moth, with complete tree mortality occurring on some 500 to 600 acres in patches ranging up to 50 acres. Local residents told him that 1946 was the second year of the outbreak. Heavy egg deposits indicated that tussock moth populations were still increasing, and that many trees would probably be killed in 1947.

Buckhorn was particularly interested in the Troy infestation because it coincided with a much larger tussock moth outbreak in central Idaho and northeastern Washington. Extensive control operations were already being planned for the Washington and Idaho areas. He decided that the Troy area was close enough to the central Idaho outbreak to coordinate a control program with one already planned there.

Buckhorn and the Walla Walla District Ranger (Willis W. Ward) completed an aerial survey of the Troy outbreak on March 24 and 25, 1947 by using a Stinson Voyager airplane on loan from the Forest Service's Division of Fire Control. On the first day, they flew about 1,000 to 1,800 feet above the tree tops and at a speed of 110 miles per hour, which was deemed to be a little too fast for best results.

As soon as the survey began, they realized it was going to be difficult to distinguish the undamaged tree tops of western larch, which looked gray because they had not yet leafed out, from the gray tops of grand firs and Douglas-firs defoliated by tussock moth. On the second day, Buckhorn decided to fly closer to the tree tops (only 200 feet above the canopy) in an attempt to separate western larch from the other species. But this strategy was also ineffective because the trees passed by too quickly, and were too close at hand, to permit accurate species determinations.

Buckhorn's final sketch map showed that 56,065 acres were infested with tussock moth at some level of severity, and that heavy defoliation occurred on 1,265 acres, moderate defoliation on 23,890 acres, and light defoliation on the remainder (30,910 acres). Between June 24 and July 1 of 1947, 14,000 acres of the Troy tussock moth outbreak were sprayed with insecticide. A C-47 airplane carrying 1,000 gallons of spray solution per trip was used for the project; it operated from the Moscow, Idaho airport located about 65 miles from the Troy infestation (Buckhorn 1947).

The spray was a solution of DDT in fuel oil applied at the rate of one pound of insecticide (DDT) in one gallon of diluent (fuel oil) per acre. The results were 'phenomenal' with practically 100 percent reduction in tussock moth numbers. But on unsprayed areas, a polyhedral virus disease appeared near the end of the feeding period and caused widespread mortality of tussock-moth larvae and pupae. On four small areas north and east of Troy, the virus in combination with natural predators virtually wiped out the tussock moth population, and no spraying was necessary.

Later in the season, however, tussock moth developed to a greater extent than anticipated in certain untreated areas and caused extensive damage there; the same thing happened in another location about eight miles south of the project area near Promise, Oregon (Buckhorn 1947).

Early-1970s Tussock Moth Outbreak

The next broad-scale tussock moth outbreak affecting the northern Blue Mountains occurred in the early 1970s. The first damage was noticed as 2,400 acres of defoliation in the Okanogan Valley of north-central Washington in 1971. In 1972, over 197,000 acres were defoliated in Oregon and Washington.

Perhaps some of the worst tussock moth damage occurred on the northern half of the Umatilla National Forest. By 1974, 44% of defoliated acreage in the outbreak area (including state, private, and other federal ownerships) was on the Forest – 353,850 acres out of a total outbreak area of 800,000 acres! How did the Umatilla National Forest respond to the 1970s outbreak? The Forest Service acted quickly and decisively in response to tussock-moth damage, and the political aspects of this story are interesting.

DDT, the powerful chemical insecticide used in the 1947 spray project near Troy and for experimental control of western spruce budworm populations on the Heppner Ranger District and adjacent Kinzua Pine Mills lands (Eaton et al. 1949), was found to affect many organisms beyond insects that defoliate trees. Due to its environmental persistence and the broad spectrum of organisms affected by it, William Ruckelshaus, director of the Environmental Protection Agency, banned DDT on June 14, 1972.

From the perspective of the 1970s tussock moth outbreak, the EPA's ban couldn't have come at a worse time. Banning DDT removed the most effective weapon against tussock moth during the first year of what would turn out to be the largest and most severe outbreak ever recorded. Following the DDT ban, the Forest Service immediately began testing other potential insecticides, including Zectran, carbaryl (Sevin), Pyrethroid, and Dylox, all of which were chemical compounds, and two biological control agents – *Bacillus thuringiensis* (a bacterium), and a natural nucleopolyhedrosis virus of tussock moth. After a Forest Service petition requesting emergency use of DDT was denied by EPA in June 1973, 32,000 acres of the Walla Walla watershed was sprayed immediately with Zectran as a test project.

On Thursday, August 16, 1973, when United States Senator Bob Packwood was reviewing tussock moth damage near La Grande, Oregon, a forest fire broke out near Perry and burned nearly 6,000 acres in a short period of time, including an area damaged by tussock moth. This Rooster Peak fire directly threatened La Grande, burning several homes at its edge and coming within yards of others. Over 1,500 people fought the Rooster Peak fire, and many of them were local residents of La Grande. The National Guard was mobilized to evacuate homeowners from foothill areas. Shortly after this fire event, an area-wide fire closure was implemented because of high fire danger from tussock moth damage and an on-going drought.

Initially, Senator Bob Packwood had no official position regarding EPA's ban on the use of DDT. Following the Rooster Peak fire, and after examining thousands of acres of tussock-moth damage from both the air and the ground, Senator Packwood eventually expressed this opinion regarding the DDT ban: "But, now I'm convinced their decision was wrong" (referring to EPA's decision not to authorize use of DDT for tussock moth control).

Following Packwood's visit and the Rooster Peak fire, petitions began circulating in northeastern Oregon requesting that the EPA's ban be lifted so DDT could be used against tussock moth. On August 31st of 1973, Secretary of Agriculture Earl Butz visited the Blue Mountains to view tussock moth damage firsthand.

In January 1974, the EPA held hearings in Portland to consider possible DDT use against tussock moth. On January 30, 1974, the Tussock Moth Control Association of La Grande, Oregon presented petitions containing 57,000 signatures to Vice President Gerald Ford; the petitions requested that DDT be allowed for emergency use against tussock moth. On February 26, 1974, EPA director Russell Train authorized emergency use of DDT against tussock moth only.

After a Johnny Appleseed clean-up weekend in early June 1974, when 2,000 volunteers performed clean-up work in tussock moth damaged areas, a tri-State DDT spray project began on June 9, 1974 on the Colville Indian Reservation. By June 22, DDT spraying was underway in the Blue Mountains, eventually concluding on July 25, 1974. A total of 426,559 acres were sprayed in 1974 to reduce tussock moth populations, including 32,706 acres on the Umatilla and 72,717 acres on the Wallowa-Whitman national forests (Graham et al. 1975).

Although applying an insecticide was the primary Forest Service response to tussock moth defoliation, salvage sales to harvest damaged and dead timber were also completed. The first Umatilla National Forest salvage sale was sold on November 28, 1972. The last of 40 tussock-moth salvage sales was awarded on September 3, 1974. Many old harvest units in places like Ruckel Ridge, Phillips Creek, and upper Tiger Canyon date from the tussock-moth salvage program of the early 1970s.

The following notes, which were prepared by Paul Bouchard (retired forester), describe how the Pendleton Ranger District responded to the tussock moth outbreak (the Pendleton Ranger District, later consolidated into the Walla Walla Ranger District, had more impact from tussock moth than other areas on the Umatilla National Forest).

> "The 1973 aerial sketch map showing tussock moth defoliation became the planning map for the salvage timber harvest program. The heavy infestation and damage areas were used to rough out potential timber sale area boundaries. By estimating the potential treatment area and timber volume by damage classes, a rough estimate of total sale acreage and salvage volume was then available for program management purposes (personnel, supplies, funding needs, etc.).
>
> It was estimated that the tussock moth salvage program could involve as much as 210 million board feet of timber volume from a gross analysis-area acreage of 66,000 acres, of which 38,000 was forested. All of the potential treatment areas were reconnoitered from the air and sale area boundaries then established on 4-inch-to-the-mile aerial photographs enlarged from a 1970 high-altitude reconnaissance flight. Areas more than 800 feet below the ridgelines and areas with small-diameter, dead white fir (usually in the headwaters of intermittent drainages) were eliminated from timber sale consideration.
>
> Three potential silvicultural treatments were mapped from the aerial reconnaissance: clearcutting (completely dead areas); shelterwood cutting (mostly dead areas); and partial cutting (areas with intermixed mortality). Nearly pure inclusions of non-host tree species and light damage of host species were also mapped. Due to time and personnel limitations, a very extensive and limited ground check and plot cruise was made (plot locations were noted on aerial photographs). The cruise amounted to about a 2/10 of 1% sample.
>
> By correlating plot data with experienced estimates, a salvage timber harvest program was developed for 128 million board feet covering 55,000 gross acres and 23,000 net acres. Approximately 137 million board feet was cut and removed under 13 timber sale contracts (105% of the appraised amount)."

Another Forest Service reaction to the early-1970s tussock-moth outbreak was to develop an early-warning system utilizing pheromone traps to monitor tussock moth population levels (pheromones are biochemicals whose odor attracts particular insects). This early-warning system was implemented West-wide in 1980 as one way to help predict tussock moth outbreaks, and it is still in use today.

Late-1990s Tussock Moth Outbreak

Since tussock moth develops rapidly, the early-warning system was designed to predict population increases with enough lead-time to implement a treatment program before serious tree damage could occur. Beginning in 1998, this early-warning system indicated that the northern Blue Mountains were facing another tussock-moth outbreak.

An outbreak actually began in the spring of 2000, and 39,392 acres on the Umatilla and Wallowa-Whitman national forests were sprayed with TM-BioControl, a natural virus affecting tussock moth only, during June and July of 2000. The objective of this spray project was to minimize tussock-moth damage in specific areas of high concern, such as old-growth stands and bull-trout habitat. A few areas that weren't sprayed sustained high levels of defoliation and tree mortality (fig. 27).

Now that federal lands are no longer managed for high levels of commodity outputs, there is less economic justification for large, widespread spray programs like those of the 1950s and the 1980s (see fig. 25).

Figure 27 – Effects of Douglas-fir tussock moth defoliation as it existed in the Indian Creek and Bologna Basin areas of the Heppner Ranger District around 2000. The late 1990s and early 2000s tussock moth outbreak was not as widespread as the early 1970s outbreak, but it did affect localized areas of moist forest on both the north and south ends of the Umatilla National Forest, such as this area in the southwestern portion of the Forest.

Additional Information

Note that dozens of literature references about Douglas-fir tussock moth and its disturbance ecology are provided in the References section of this white paper. Synthesis reports about tussock moth, and possible management responses to its impact (defoliation, topkilling, tree mortality), are provided by Brookes et al. (1978) and an insect and disease leaflet (Wickman et al. 1981). Gast et al. (1991) provide a comprehensive discussion about tussock moth outbreaks, and resulting impacts, for the Blue Mountains. British Columbia's Defoliator Management Guidebook (British Columbia Ministry of Forests 1995c) is also a useful tussock moth reference.

5.4 Ecological Role Of Defoliating Insects

Defoliators and other landscape-scale insects often respond to increased landscape homogeneity. For western Montana, Anderson et al. (1987) found that 90 years of reduced fire frequency allowed extensive areas with high budworm susceptibility to develop. Timber harvest preferentially removed non-host tree species (e.g., ponderosa pine and larch), serving to aggravate the problem (see fig. 21 and 24).

Anderson et al. (1987) concluded that after selective tree harvests and reduced fire frequency caused changes in forest structure and composition at a landscape scale, spruce budworm outbreaks increased in both duration and intensity, but not in frequency. Bruce McCune (1983) came to a similar conclusion after examining western spruce budworm trends for the Bitterroot Range in western Montana. He suggested that recent increases in outbreak intensity were linked to fire suppression and its impact on conifer species composition.

Baskerville (1975) characterized spruce budworm as a ‘super silviculturist’ and did not view periodic budworm outbreaks as a sign of ecological instability. When budworm defoliation events were evaluated on a time scale appropriate to their function – at least 50 to 100 years – then outbreaks could be viewed as one factor contributing to ecological stability at a landscape scale.

Baskerville (1975) believed that to some extent, the direct suppression strategy as a management policy had been counterproductive: by spraying large areas of mature host type that was not being immediately harvested, managers were perpetuating an ideal food supply for the next budworm outbreak. He noted that the forest cannot be managed by harvesting small ‘bites’ because this practice maintained broad-scale conditions conducive to an outbreak. The longer that susceptible forest was protected in excess of what could be harvested, the longer that high-risk area was being exposed to a future outbreak.

Although Baskerville’s (1975) stability perspective may be useful for areas unaffected by human activity, recent trends toward increasing budworm impact and habitat (host type) at a broad scale (Williams et al. 1980), as depicted in figures 19 and 24, suggest we shouldn’t pretend we’re dealing with a natural situation now. [Range of variation analyses for species composition, forest structure, and stand density frequently demonstrate that existing conditions provide more high-quality budworm habitat than was associated with reference (historical) conditions.]

Defoliating insects can influence nutrient cycling and site productivity. A study conducted near Mammoth Lakes, California found that radial growth of trees defoliated by Douglas-fir tussock moth (DFTM) was significantly greater than that of non-defoliated trees, even 40 years after the outbreak had subsided (Wickman 1980). It is hypothesized that the enhanced tree growth was related to increased nutrient cycling from insect frass (excrement) and litter fall, in combination with a thinning effect (e.g., selective tree mortality) caused by DFTM defoliation (Wickman 1990).

In the long run, defoliated forests may produce as much or more tree growth than if insect defoliation had not occurred, in part due to compensatory growth rate increases such as those described by Wickman (1980) following a DFTM outbreak in California (Mattson and Addy 1975, Schowalter and Withgott 2001, Stuart et al. 1989). These studies suggest that insect outbreaks may not be detrimental in the long term, and they could in fact be useful for maintaining ecosystem integrity by functioning as 'normative outbreaks' (Mattson 1996).

These results agree with many other studies examining forest response to insects. Native insects work to thin trees, control crowding, reduce stress, and lessen competition for water and nutrients. Moderate levels of insect herbivory (foliage feeding removing as much as 40 to 50 percent of the canopy) may even be healthy for trees and forests, resulting in little or no difference in plant growth or survival (Attiwill 1994). And depending on the ratio of host to non-host species, it has been found that maintaining high levels of tree species diversity will reduce herbivory impact by forest insects (Jactel and Brockerhoff 2007).

The research findings described above suggest that over the long run, thinning and similar silvicultural practices may be the most effective way to deal with defoliating insects such as western spruce budworm. Research from Montana found that thinning improved budworm resistance by increasing stand vigor, increasing budworm larval mortality during their dispersal period, and by reducing budworm-host species in mixed-conifer forests (Fauss and Pierce 1969). Thinning provided short-term protection for treated stands, and would presumably contribute to long-term resistance once landscape-sized areas were treated (Carlson and Wulf 1989).

"Moderate levels of insect and pathogen activity may reduce the risk of catastrophic fire by thinning stands and preventing excessive fuel buildups; insects and pathogens may also promote forest health by gradually culling weakened trees" (White et al. 1999). This observation is supported by recent research indicating that "fire intensity and crowning are reduced with increasing defoliation" by western spruce budworm (Cohn et al. 2014).

Insects, pathogens, and wildfires are important disturbance processes, and they also play a vital role in ecosystem function (fig. 28; Attiwill 1994, Castello et al. 1995, Franklin et al. 1987, Harvey 1994, Hunter 1990, Mattson and Addy 1975, Schowalter and Withgott 2001, Thomas 1979, Wickman 1992). Disturbance processes create dead or dying trees, which in turn affect plant succession and the diversity of animal communities in an area (Oliver et al. 1997).

Localized outbreaks of defoliating insects can provide important ecosystem services such as creating snags (Quigley et al. 2001) or dead-top live trees used as maternity roost sites by big brown bats (Rancourt et al. 2007). It is important to recognize that living systems develop and evolve. Understanding these systems requires a shift in focus from structure to process, including evolution, renewal, and change.

Figure 28 – Insects and pathogens are important components of biodiversity (line drawing from Doliner and Borden 1984). Forests support an amazing diversity of life, ranging from soil microbes to large, long-lived, woody plants. Insects and pathogens are an important component of this diversity. Not only do they contribute to nutrient cycling, productivity, and other ecosystem services, but they also create wildlife habitat (dead trees, and cavities in living trees), and insects and fungi are a food source for many wildlife species (Hunter 1990, Schowalter and Withgott 2001).

Dead wood provides habitat for a variety of wildlife species (Bull et al. 1997) and is a key factor influencing the species diversity of an area (Franklin and Forman 1987). In the Blue Mountains, standing dead trees (snags) are used for nesting or shelter by 39 bird and 23 mammal species; coarse woody debris (CWD; downed wood) is used by at least 179 wildlife species for cover, foraging, or reproduction (Thomas 1979). Some familiar animal users of CWD include woodpeckers, owls, salamanders, voles, and mice. Factors affecting vertebrate use of CWD include its horizontal or vertical orientation, size, decay class, the tree species it was derived from, and its overall abundance (Bull et al. 1997, Thomas 1979).

Natural Enemies Of Defoliating Insects

Go to the ant, thou sluggard; consider her ways, and be wise.
Biblical King Solomon
Study the insects and learn of their ways.
F.P. Keen, Influence of insects on ponderosa pine silviculture (1950)

Defoliators, and populations of other insects as well, are typically constrained to relatively low ‘endemic’ levels by the activity of birds, ants, viruses, and similar biological agents (Campbell et al. 1983, Fellin 1980, Langelier and Garton 1986, Mason 1992, Takekawa and Garton 1984, Torgersen et al. 1990). These agents are often referred to as ‘natural enemies’ in a pest management context (Berryman 1982).

Land managers may be able to influence the intensity, duration, or extent of future defoliator outbreaks by how they affect the natural enemies of defoliating insects (fig. 29, table 8). Ants, birds, yellowjackets, and other defoliator predators can be affected by prescribed fire, silvicultural treatments, insecticide applications, timber harvest activities, and other land management practices (Powell 1994).

At least 26 birds have been documented as budworm predators, but it's likely more species than that play such a role. Mountain chickadees and red-breasted nuthatches are very important predators in the Blue Mountains. Budworm-feeding birds can be promoted by using the following practices (Langelier and Garton 1986):

- Provide stands with horizontal and vertical diversity.
- Avoid large clearcuts.
- Provide edges.
- Avoid high-grading timber harvest.
- Avoid homogeneous, plantation-like stands.
- Leave some slash.
- Reduce herbicide use.
- Control livestock grazing.
- Provide habitat for cavity-nesting birds.
- Provide salt.
- Provide water.

Managers can also affect bird populations by how they handle insects or pathogens other than budworm. For example, a study found bird diversity and abundance to be greater in forests infected with dwarf mistletoes. In fact, the number of bird species increased as the dwarf mistletoe infection level increased. The study also found that dwarf mistletoe was not being used as food – its berries are small and hard – but the witches' brooms provided nesting and roosting sites, and served as habitat for butterflies, moths, and other insects that birds feed on (Mlot 1991).

Ants and ground-dwelling spiders can be affected by site preparation activities, fuel or residue treatments, insecticide applications, and timber harvest practices (fig. 30). Retaining down woody material, particularly large logs and standing dead trees (snags), is important for sustaining carpenter ants and other forest-floor invertebrates. Recent recommendations for retention of logs and snags (Bull et al. 1997) for pileated woodpeckers and other vertebrates, long-term site productivity, fungi and bryophytes, nutrient cycling, and for other purposes, would probably be adequate to meet many of the needs of ants and forest-floor spiders.

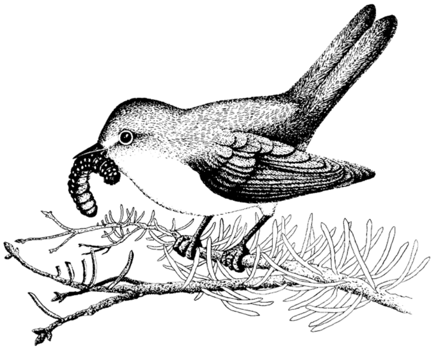
During timber harvest and other activities involving heavy equipment, thatch-ant nests should be avoided if possible. Even if thatch ants are not protected in recognition of their intrinsic value to the ecosystem, the nests should be avoided because thatch ants are important predators of spruce budworm larvae (fig. 30).

Figure 29 – Interactions between tree harvest, pileated woodpeckers, carpenter ants, Vaux's swifts, and western spruce budworm (drawing by Susan Lindstedt, and taken from Bull et al. 1995). There are intricate and subtle interactions between dead wood, hollow trees, carpenter ants, pileated woodpeckers, Vaux's swifts, and western spruce budworm. Moist forests that provide standing and down dead wood, in appropriate amounts when balancing fuels, wildlife, and forest health considerations, have been found to experience less budworm damage because they provide habitat for ants and many of budworm's natural enemies (Bull et al. 1995, Torgersen and Bull 1995).

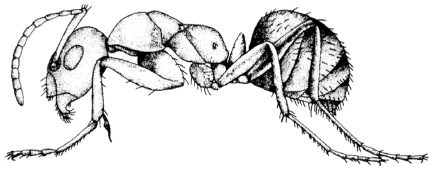
Murphy and Croft (1990) examined the effect of applying a chemical insecticide (carbaryl) for spruce budworm population suppression on the foraging behavior and species diversity of ants. After spraying, ant species diversity declined in treated plots. Post-spray ant foraging decreased in all plots, including the untreated controls, but the decrease was more rapid and pronounced in treated plots.

Ants that are arboreal foragers, nearly all of which are also budworm predators, showed a significantly lower foraging rate in the budworm-treated areas. It was concluded that reduced ant predation on the sparse budworm populations present after spraying may have contributed to a budworm resurgence in the treated area (Murphy and Croft 1990).

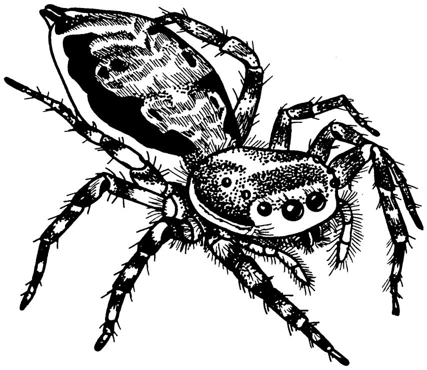
Table 8: Common predators of defoliating insects (adapted from Powell 1994).



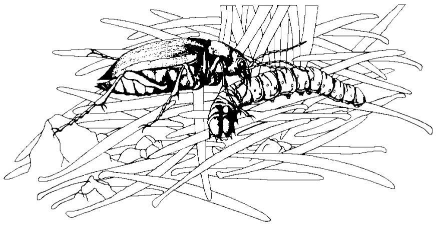
Birds are important budworm predators. At least 26 birds have been documented as budworm predators (Garton 1987), but more species than that may play such a role. The ruby-crowned kinglet, shown here feeding on a budworm larva, and mountain chickadees and red-breasted nuthatches are particularly important budworm predators. Many of these avian budworm predators require dead wood (standing snags or tall stumps) in which to nest.



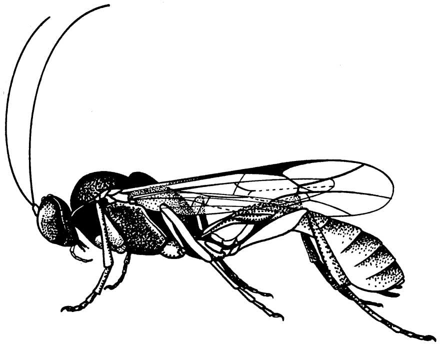
Ants are important predators of budworm. Thatch ants (shown here) and carpenter ants are effective at preying on budworm larvae that fall to the forest floor (Finnegan 1971). Some ants are arboreal foragers; they can be very important by feeding on budworm larvae that fall onto seedling- and sapling-size host trees (Murphy and Croft 1990).



Spiders outnumber all other arthropods on the foliage of interior Douglas-fir and true firs. In the Blue Mountains, hunting spiders are much more plentiful than the web spinners (Mason 1992), although both types prey on budworm. An especially effective budworm predator is the jumping spider shown here. Note that “spiders are the primary invertebrate predator of the eastern spruce budworm, a close relative of the western spruce budworm” (Perry 1988).



Beetles and the true bugs (stink bugs, etc.) are just two examples of insect groups that feed on budworm larvae that fall to the forest floor. Little information is available about the importance of these insects as budworm predators. A carabid beetle feeding on a budworm larva is shown here.



Many species of wasps and flies are known to parasitize insect larvae. Parasitic insects find budworm larvae, pupae, or eggs, where they then lay their own eggs. After their eggs hatch, the developing parasite feeds on the budworm host, eventually killing it. The wasp shown here is a common Blue Mountain species *(Hyposoter masoni* Torgersen). Note that successful control of larch casebearer has largely been achieved by introduction of parasitic wasps (Torgersen 2001).

Figure 30 – Large thatch ant nest on a moist-forest site (from Powell 1994). Ants, spiders, and other forest-dwelling arthropods (table 8) can be affected by timber harvest, site preparation activities, fuel or residue treatments, insecticide applications, and other management practices (Campbell et al. 1983, Fellin 1980, Mason 1992, Murphy and Croft 1990, Torgersen et al. 1990). During timber harvest operations and other activities involving mechanized equipment, thatch-ant nests should be avoided because ants are important predators of defoliating-insect larvae (photo courtesy of Torolf Torgersen). And perhaps not unsurprisingly, these nests also function as hotspots for nutrient cycling because the ants' shredding action of organic nest materials aids decomposition processes and nutrient release. Note that Bull et al. (1995) provide an interesting case study about active management of an old-growth, moist-forest stand in the Blue Mountains, and its effect on birds, ants, and large woody debris. Bull et al. (1997) also provide guidelines about sustaining trees and logs (both standing and down) for wildlife, and wildlife in their context also includes invertebrates such as ants. Also, recent research noted increased bat activity in areas affected by silvicultural treatments, and this may translate into increased foraging on Lepidoptera (like budworm), an important component of their prey base (Dodd et al. 2012).

Management interest in maintaining or promoting habitat for birds, ants, spiders, beetles, wasps, and other natural enemies of defoliating insects (table 8) is more prevalent now than historically because recent defoliator outbreaks operated differently than they did in the past (Gast et al. 1991, Wickman 1992). Previous outbreaks tended to be small, widely scattered, and subsided quickly from natural causes (Torgersen 2001), but a substantial increase in defoliator habitat across the Blue Mountains (Powell 1994, Williams et al. 1980) suggests that this change will overwhelm the capacity of natural enemies to exert an influence on the severity, duration, or spatial extent of future outbreaks. If the underlying cause of defoliator intensification – increased habitat or host type – is not addressed in some fashion, then it would be unreasonable to expect that providing enhanced conditions for natural enemies could change the course of future outbreaks.

5.5 Dwarf Mistletoes

Douglas-fir Dwarf Mistletoe *(Arceuthobium douglasii)*

TREE SPECIES AFFECTED: Douglas-fir.

DAMAGE CAUSED: Top-killing; reduced growth; deformed stems; brooms; tree mortality.

IMPACTS AND EFFECTS: A severe parasite – one survey found that 42% of the Douglas-fir type on the east side of Region 6 is affected. Causes extensive mortality on poor sites.

CONTROL MEASURES: Regenerate mature stands; leave buffers between infected trees and uninfected seedlings; quickly remove infected trees from partial cuts; discourage thinning of infected stands because brooming is suppressed and infection is latent.

FOREST INSECT AND DISEASE LEAFLET: No. 54 (Hadfield et al. 2000).

Larch Dwarf Mistletoe *(Arceuthobium laricis)*

TREE SPECIES AFFECTED: Western larch; lodgepole pine and subalpine fir (occasional).

DAMAGE CAUSED: Reduced growth and seed output; lumber defect; brooms; tree mortality.

IMPACTS AND EFFECTS: A serious larch enemy – one survey found 47% of host type in the Pacific Northwest is affected. Causes tree mortality quicker than other dwarf mistletoes.

CONTROL MEASURES: Same as below for lodgepole pine dwarf mistletoe (and as would be done for western dwarf mistletoe in ponderosa pine).

FOREST INSECT AND DISEASE LEAFLET: No. 169 (Beatty et al. 1997).

Lodgepole Pine Dwarf Mistletoe *(Arceuthobium americanum)*

TREE SPECIES AFFECTED: Lodgepole pine; ponderosa pine (occasional).

DAMAGE CAUSED: Reduced vigor and growth; stem cankers; tree mortality.

IMPACTS AND EFFECTS: Severe – one survey found that 42% of host type in Pacific Northwest is infected.

CONTROL MEASURES: Clearcut mature stands; sanitize young stands by removing infected trees during thinning; promptly remove infected overstory trees from partial cuts.

FOREST INSECT AND DISEASE LEAFLET: No. 18 (Hawksworth and Dooling 1984).

Dwarf mistletoes are parasitic seed plants widely distributed in coniferous forests of the northern hemisphere, particularly in older, multi-storied stands (Hawksworth and Wiens 1996). Spreading most effectively from tall trees onto shorter trees, dwarf mistletoes withdraw water and nutrients from their hosts and can eventually kill them, especially when infections occur throughout the crown (fig. 31). In addition to directly parasitizing their hosts for resources, dwarf mistletoes induce abnormal growth patterns and typically cause dramatic changes in a tree's allocation of water, photosynthate, and other growth-related resources.

A western Montana study, for example, found that heavily infected western larches had reduced water potentials, but greater whole-tree water use, when compared with uninfected trees. This result may have contributed to further depletion of limited soil water resources for mature infected stands late in the growing season (Sala et al. 2001). These findings might provide a clue for why larch dwarf mistletoe seems to cause higher levels of tree mortality in western larch than is observed for other combinations of tree species and dwarf mistletoe in the Blue Mountains.

Dwarf mistletoes spread slowly in a stand, averaging only one to two feet per year, and their progress can be slowed or controlled altogether by removing infected trees, and then planting non-susceptible species (Quigley et al. 2001). Witches' brooms, which are characteristic infection symptoms for a number of dwarf mistletoes (fig. 31), can function as crucial wildlife structure for several avian and mammal species (Bull and Heater 2000, Hedwall and Mathiasen 2006).

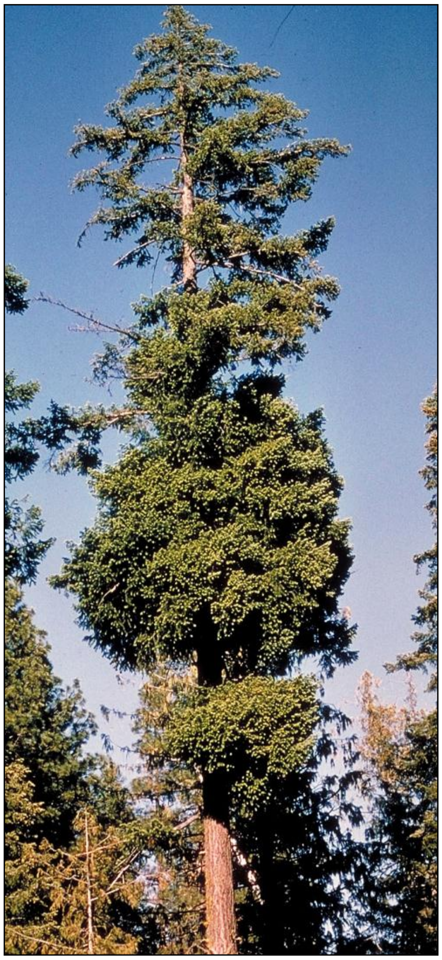


Figure 31 – Interior Douglas-fir infected with dwarf mistletoe (from Powell 1994). This interior Douglas-fir has several areas with dense, bunched branches called 'witches' brooms' caused by Douglas-fir dwarf mistletoe. A recent study found bird diversity and abundance to be greater in forests infected with dwarf mistletoes (Mlot 1991). The study showed that dwarf mistletoe was not being used as food – its berries are small and hard – but the witches brooms it caused provided nesting and roosting sites, and served as habitat for butterflies, moths, and some of the other insects that birds feed on. Since birds are important predators of defoliating insects such as western spruce budworm and Douglas-fir tussock moth (table 8), it may not be prudent to remove all of the mistletoe-infected trees for mixed-conifer forests with high defoliator susceptibility (Parks et al. 1999). When considering an appropriate level of dwarf mistletoe retention during project planning, land managers could consult range of variation information contained in Schmitt and Powell (2012).

Early descriptions of Blue Mountain forests often described effects of infection by dwarf mistletoes. Harold P. Gilkey noted in an annual silvical report for the Whitman National Forest that dwarf mistletoes were abundant throughout the western larch type, with saplings frequently infected by the time they were an inch in diameter. He also noted that dwarf mistletoe infection levels varied by physiognomic type: 79% of western larch was found to be infected in mixed stands on the dry slope type (western yellow pine), while only 27% of larch was infected on the moister north slope and transition sites (Gilkey 1912).

When George Bright completed a survey of the Wenaha National Forest (north half of the Umatilla National Forest) in 1914, he made this observation about Douglas-fir dwarf mistletoe: “Mistletoe, it is thought, is on the increase. It is killing many Douglas fir. Nearly every large or medium-sized Douglas fir will often be found to be infested with this disease, on certain north slopes” (Bright and Powell 2008).

In addition to western larch and Douglas-fir, lodgepole pine and ponderosa pine are also affected by host-specific dwarf mistletoes. Of the common tree species associated with moist, mixed-conifer sites in the Blue Mountains, Engelmann spruce, subalpine fir, and grand fir are not affected by dwarf mistletoes (although grand fir is infected in central Oregon and other portions of its range).

Dwarf mistletoes often predispose trees to attack from insects or pathogens. A budworm-impact study for the central and southern Blue Mountains found that western spruce budworm killed Douglas-firs infected with dwarf mistletoe more often than would have been expected, as based on the infection frequency for the pre-outbreak forest. Presumably, infected trees were killed more often than uninfected trees because Douglas-fir dwarf mistletoe caused physiological stress, which predisposed them to budworm-induced mortality (Powell 1994). Many studies identified interactions between dwarf mistletoes and other disturbance agents (Filip et al. 1993, Filip and Parks 1987, Hoffman et al. 2007, Knutson and Tinnin 1986, Knutson and Toevs 1972, Schmitt and Hadfield 2009, Wagner and Mathiasen 1985).

For moist-forest sites where dwarf-mistletoe infection levels are high (such as those with more than 50 percent of the host trees infected), treatment prescriptions to develop or maintain an even-aged forest structure may be warranted. When uneven-aged management (group or individual-tree selection) is being considered for moist forests, at least 75 percent of the host trees should be free of dwarf mistletoes to avoid problematic increases in infection levels (Evans et al. 2011).

When Hessburg et al. (2008) examined the efficacy of prescribed fire and thinning as active management treatments for addressing dwarf-mistletoe infection in mixed-conifer forest, they found that thinning produced the greatest reductions in tree stocking and mistletoe severity, but they also found a trend of reduced mistletoe spread and intensification over time for both treatments. This study also found that treatment effects diminished after 20 years, suggesting that a 20-year treatment cycle might be effective for actively managing dwarf mistletoe levels.

Additional Information

In addition to citations in this section, much other literature about dwarf mistletoes, their disturbance ecology, and possible management responses to their impact is included in the References section. Schmitt (1997) provides a Management Guide for Douglas-fir dwarf mistletoe in the Blue Mountains. Schmitt also provides a report discussing the pros and cons of managing Douglas-fir dwarf mistletoe or larch dwarf mistletoe for wildlife use (Schmitt 1996). Agriculture Handbook 709 (Hawksworth and Wiens 1996) provides a comprehensive synthesis of the biology, pathology, and systematics of dwarf mistletoes. Gast et al. (1991) provides discussions for dwarf mistletoes of the Blue Mountains. British Columbia's Dwarf Mistletoe Management Guidebook (British Columbia Ministry of Forests 1995d) is a useful reference. Citations for three Forest Insect and Disease Leaflets describing dwarf mistletoes are provided at the beginning of this section.

5.6 Holes And Hollows: Stem Decay From Indian Paint Fungus

Rust Red Stringy Rot Caused by Indian Paint Fungus *(Echinodontium tinctorium)*

TREE SPECIES AFFECTED: Grand fir primarily; subalpine fir (occasionally) and Engelmann spruce (rarely).

DAMAGE CAUSED: Stem decay of the heartwood tissue.

IMPACTS AND EFFECTS: Widespread in old-growth, mixed-conifer stands; most important stem decay disease in the Blue Mountains.

CONTROL MEASURES: Maintain vigorous stands; use short rotations; avoid bole wounding; avoid managing advanced regeneration more than 50 years old.

FOREST INSECT AND DISEASE LEAFLET: No. 93 (Filip et al. 2009).

Stem decay fungi are common in moist forests, with incidence levels tending to rise as forest age increases (Hansen and Goheen 2000). The most common stem decay fungus is rust-red stringy rot caused by the Indian paint fungus (fig. 32). Although airborne spores of stem-decay "fungi are abundant in forest stands of all ages, old-growth stands have a much higher incidence of heart rot than do young stands. Non-resinous species such as the true firs are more prone to stem decay than are species such as Douglas-fir, pines, or larch. Grand (white) fir is the most suceptible species" (Parks and Flanagan 2001).

Indian paint fungus is more common on moist grand fir sites (such as the grand fir/twinflower and grand fir/queencup beadlily plant associations) than on the dryer end of the grand fir series (the grand fir/elk sedge, grand fir/pinegrass, and grand fir/birchleaf spirea associations). Nearly 80 percent of the wood decay associated with old grand fir stands is attributed to Indian paint fungus (Filip and Schmitt 1990). And in the old stands, nearly every fir stem may eventually be decayed, as described in this early account: "White fir in this region is very poor and should be considered a weed. Trees of this species over 16 inches D.B.H. are seldom sound because of the heavy attacks of Indian paint fungus" (Starker 1916).

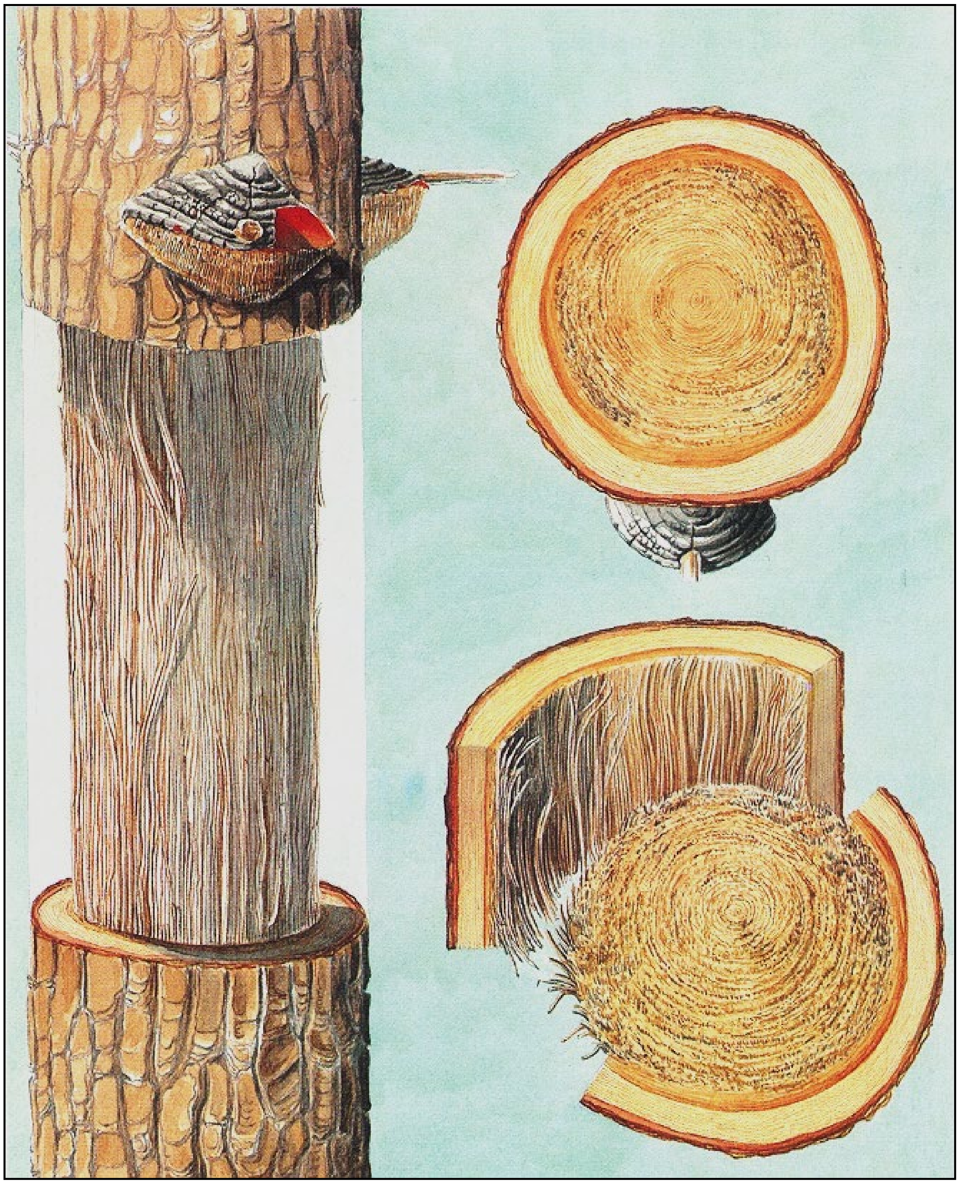
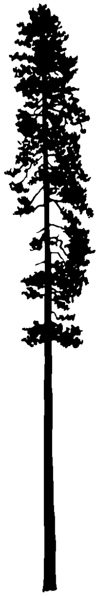


Figure 32 – Rust-red stringy rot in a grand fir tree caused by a stem decay organism called the Indian paint fungus (from Shigo 1979). Indian paint fungus is a common pathogen and wood-decay organism affecting older grand fir trees in the Blue Mountains. The fungus usually enters the stem after a long growth period in a dying branch. When many branches die at about the same time, the fungus may convert the entire central column of the stem into a stringy, fibrous mass of decayed material. Indian paint fungus is an example of a decay organism that creates a valuable type of wildlife tree: hard outer wood surrounding decay-softened inner wood. This type of wildlife tree is used extensively by a variety of avian species such as yellow-bellied sapsucker, red-naped sapsucker, red-breasted sapsucker, hairy woodpecker, three-toed woodpecker, black-backed woodpecker, and pileated woodpecker (Conner et al. 1976, Keisker 2000). The inside of the fruiting body of this fungus has a brick-red color, and Native American peoples of the Columbia River basin (including the Canadian portion of the basin) used the fruiting body to produce a red pigment, which was used as a dye or stain (Turner et al. 1990).

Table 9: Selected life history traits for eight common conifers of moist upland forest.

	Ponderosa Pine	Western Larch	Lodgepole Pine	Interior Douglas-fir	Western White Pine	Grand Fir	Engelmann Spruce	Subalpine Fir
								
Shade Tolerance	Intolerant	Very intolerant	Intolerant	Intermediate	Intermediate	Tolerant	Tolerant	Very tolerant
Fire Resistance	High	Very high	Low	Moderate	Moderate	Moderate	Low	Very low
Survival Strategy	Resister	Resister	Evader	Resister	Resister	Avoider	Avoider	Avoider
Wind Resistance	Very high	Very high	Moderate	High	Moderate	Moderate	Very low	Low

Sources/Notes: Shade tolerance is from Burns and Honkala (1990a), Daniel et al. (1979), or Keane et al. (1996a). Fire resistance is from Flint (1925) and Starker (1934); fire survival strategy is from Rowe (1983). Wind resistance integrates rooting and stem breakage characteristics. Column heading color is seral status: orange is early-seral; green is mid-seral; and blue is late-seral. More life-history information is provided in tables 10 and 16.

Although most prominent in older trees and stands, Indian paint fungus actually infects a full range of tree sizes: “the Indian paint fungus is responsible for 80% of the decay in old-growth grand and white fir stands in eastern Oregon and Washington and 30% of the decay in advance fir regeneration” (Filip et al. 2009).

The most conspicuous indicator of Indian paint fungus infection is a large, hoof-shaped protuberance or conk (fruiting body or sporophore) produced on the tree stem and just below an old branch (fig. 33; note that two of the three trees near the center of the photograph have obvious conks located immediately below dead branches). The conks are woody with a black, cracked upper surface, and gray, spiny or toothed lower surfaces – the interiors are a bright red or orange color, leading to their use as a dye or pigment by Native Americans (fig. 32).

High shade tolerance (table 9, previous page) contributes to the capability of grand fir to persist almost indefinitely as an understory species, surviving on sunflecks and other ephemeral light sources (Canham et al. 1990, Chazdon and Pearcy 1991). But the understory persistence of grand fir also serves to prolong its exposure to infection by Indian paint fungus (fig. 32). Although Indian paint fungus is not a particularly virulent pathogen, it does occasionally kill trees and function as a gap-phase disturbance agent (Hennon 1995).

Indian paint fungus is adept at infecting suppressed grand fir trees because they grow slowly and have low vigor, causing branchlet stubs to heal slowly. When unhealed stubs remain exposed for a long period, Indian paint fungus can infect a high proportion of them. Vigorous, rapidly growing trees have unhealed stubs for relatively short periods, greatly limiting their infection exposure (Filip and Schmitt 1990). Infections can lie dormant for many decades, only to be initiated later by wounding.

Research suggests that substantial amounts of stem decay caused by Indian paint fungus is likely in advanced grand fir regeneration under these circumstances:

- It is growing under a grand fir overstory infected with Indian paint fungus.
- It has been suppressed for more than 50 years, which means that dormant (latent) infections are probably present. Indian paint fungus infects a host at twig stubs – the small wounds created when twigs are shed from larger branches.
- It has wounds caused by tree harvest; bombardment from above by branches, dead tops, dwarf-mistletoe brooms, and other stand debris; or from other causes such as surface fire (any of this wounding can activate a dormant infection).
- It has poor vigor because of low site quality, intertree competition caused by high stand density, incipient root disease, or for other reasons.

Many sapsucker and woodpecker species of the Blue Mountains require mature trees of large diameter, with central heartwood softened by fungal decay, in which to excavate their nests. Grand firs infected with rust-red stringy rot caused by the Indian paint fungus often provide this specific and important type of habitat for cavity-nesting birds (fig. 33).

Figure 33 – An old-growth grand fir stand in the northern Blue Mountains (Walla Walla Ranger District). Note that this stand contains many of the features expected for an old-growth ecosystem on moist sites – a predominance of late-seral tree species (grand fir in this instance), generally a multi-layered vertical stand structure (although seedling- and sapling-size trees are not obvious in this particular photograph), much down wood occurring in a variety of size classes, and the presence of stem decay caused by the Indian paint fungus organism (which causes rust-red stringy rot of heartwood tissue). Two trees near the center have obvious fruiting bodies (conks) indicating their infection by the Indian paint fungus (see white arrows). The softened, well-decayed heartwood resulting from rust-red stringy rot is an important habitat component for cavity-nesting birds (Parks and Flanagan 2001).

Stem decay is just one example of a specialized wildlife niche provided by moist forests – more than 80 species of birds, mammals, reptiles, and amphibians in the interior Columbia River basin use living trees with internal decay, hollow trees, trees with brooms or dead tops, or standing dead trees (snags) for nesting, roosting, denning, or foraging (Bull et al. 1997, Thomas 1979, Wisdom et al. 2000).

For moist forests lacking mature trees with decayed heartwood, several creative methods have been developed to inoculate living trees with wood decay fungi, including shooting them with fungi-inoculated rifle bullets (Baker et al. 1996), or by drilling holes into the stem and inserting fungi-inoculated dowels (Filip et al. 2011). Note that the Filip et al. (2011) study found little benefit from artificial inoculations after 7-14 years (i.e., little stem decay and no excavated cavities), but they did recommend inoculations continue into the future, but with modifications to improve efficacy.

5.7 Root Diseases[2]

Annosus Root Disease *(Heterobasidion annosum)*

TREE SPECIES AFFECTED: Grand fir, subalpine fir, and pines.

DAMAGE CAUSED: Butt decay and tree mortality.

IMPACTS AND EFFECTS: Fairly common in mixed-conifer stands, especially those with a history of repeated partial cutting.

CONTROL MEASURES: Favor tolerant and resistant trees; use short rotations and fewest possible entries; stocking-level control; remove infected stumps or treat them with borax when fresh (immediately after felling).

FOREST INSECT AND DISEASE LEAFLET: No. 172 (Schmitt et al. 2000).

Armillaria Root Disease *(Armillaria ostoyae)*

TREE SPECIES AFFECTED: Douglas-fir and grand fir – severe; pines, spruce, and subalpine fir – moderate.

DAMAGE CAUSED: Reduced growth; butt decay; windthrow; tree mortality.

IMPACTS AND EFFECTS: Widespread in mixed-conifer stands; probably the most damaging root disease of the Blue Mountains.

CONTROL MEASURES: Favor tolerant & resistant species; avoid frequent entries and soil disturbance; sanitize when thinning; reduce density; stump removal in special situations.

FOREST INSECT AND DISEASE LEAFLET: No. 78 (Williams et al. 1986).

Laminated Root Rot *(Phellinus weirii)*

TREE SPECIES AFFECTED: Douglas-fir and grand fir – severe; Engelmann spruce, subalpine fir, and western larch – moderate.

DAMAGE CAUSED: Reduced growth; root and butt decay; increased windthrow susceptibility; tree mortality.

IMPACTS AND EFFECTS: Causes heavy tree mortality on infected sites. The fungus can survive for long time periods in infected stumps and roots.

CONTROL MEASURES: Favor moderately damaged, tolerant, and resistant species; remove infected stumps from disease centers (only used in special situations).

FOREST INSECT AND DISEASE LEAFLET: No. 159 (Nelson et al. 1981).

---

[2] Scientific and common names of insects and diseases in this paper follows Goheen and Willhite (2006).

Three root disease organisms are especially influential for moist forests of the Blue Mountains: annosus root disease, Armillaria root disease, and laminated root rot. Root diseases and other pathogenic fungi are important endemic (site-associated) disturbance processes influencing the species composition of moist-forest ecosystems, particularly by how they affect the ratio of disease-resistant (generally early-seral) and disease-susceptible (generally late-seral) tree species.

Differences in species susceptibility to root diseases (table 10) have an important influence on how silvicultural practices are applied on moist-forest sites – which species are retained during partial cutting, which species are planted following stand-replacing disturbance, and so forth.

Armillaria root disease is commonly associated with moist-forest sites (fig. 34), acting both as an aggressive pathogen and a more passive saprophyte. Host species experiencing stress, including stresses caused by soil compaction or other site damage from previous management practices, can increase the extent of root disease occurrence or its virulence. It has also been observed that planted trees will generally have a higher infection level than natural regeneration.

For this reason, sites with high existing Armillaria incidence or a history of previous infection are often regenerated using natural sources rather than planted stock. In stands with aggressive Armillaria root disease, however, natural regeneration is clearly still susceptible, but obviously less so than planted regeneration (Chapman et al. 2011).

Alternative timber harvest systems have differing potential to cause soil compaction and associated root disease impact. Cut-to-length (CTL) mechanized harvesting systems are popular, and they are effective at minimizing soil compaction by using the limbs removed during tree processing at the stump to create a slash mat in front of the machine.

Using CTL systems can be important for active management because wet soils (soil moisture ranging up to 30%), or those formed from fine-grained materials such as the volcanic ash caps or loess soils commonly associated with moist forests (McDaniel and Hipple 2010, McDaniel and Wilson 2007), are more susceptible to compaction than soils in dry forests or those derived from coarse-textured or gravelly parent materials.

And, research suggests that retaining just a small amount of slash was not effective at minimizing soil compaction because it did not provide “enough cushioning in wet soil to absorb the ground pressure and vibration of the harvesting equipment. Light slash tended to be crushed into pieces and could no longer distribute and absorb the impact of the machine” (Han et al. 2009).

Following adoption of the National Fire Plan in 2000, fuel reduction proponents have advocated more utilization of whole-tree harvesting because it removes most of the fine fuels (foliage and branchwood) to a landing where they can be piled and burned (or perhaps in the future, they could be pyrolyzed to create biochar and other products). "Whole-tree harvesting, however, has high potential for soil compaction and disturbance because skidder travel tends to sweep duff and litter from trails, exposing bare mineral soil (Hartsough et al. 1997)" (Han et al. 2009).

Table 10: Susceptibility ratings for armillaria and annosus root diseases, laminated root rot, and Indian paint fungus, for common conifers of moist upland forest.

Tree Species	Laminated	Armillaria	Annosus	Indian Paint
Douglas-fir (interior)	Severe	Severe	Seldom	Immune
Engelmann spruce	Moderate	Moderate	Seldom	Tolerant
Grand fir	Severe	Severe	Severe	High
Lodgepole pine	Seldom	Moderate	Moderate	Immune
Ponderosa pine	Seldom	Moderate	Moderate	Immune
Subalpine fir	Moderate	Moderate	Moderate	High
Western larch	Moderate	Seldom	Seldom	Immune
Western white pine	Seldom	Moderate	Seldom	Immune

Sources/Notes: Laminated, armillaria, and annosus root diseases: table 5 in Goheen and Willhite (2006); Indian paint fungus: table 5 in Williams et al. (1995). Ratings have the following meaning: High – readily infected and readily killed; Severe – severely damaged; Moderate – moderately damaged; Seldom – seldom damaged; Tolerant – infrequently infected unless growing in association with the most susceptible species, and rarely killed; Immune – species does not serve as a host for the fungus (Thies and Sturrock 1995, p. iii).

Figure 34 – Honey mushroom produced by Armillaria root disease. Armillaria root disease is the most pervasive and important root disease organism affecting moist forests of the Blue Mountains.

Root diseases are best managed by employing a prevention strategy. In situations where root disease activity has already responded to predisposing factors such as soil compaction, future management options may be relatively limited. Under these circumstances, the best option may be to use forest management practices (noncommercial thinnings, etc.) to adjust species composition in such a way that disease-resistant species, preferably those derived from locally-adapted seed sources, comprise a majority of post-thinning stocking levels (Cleary et al. 2008).

Noncommercial thinning to reduce stocking levels and thereby improve vigor of residual trees can have a positive influence on reducing Armillaria root disease impact (Filip et al. 2009), but thinning will not eliminate the disease. And if commercial thinning involving ground-based skidding operations causes soil compaction or residual-tree wounding (Han et al. 2009), either outcome would likely contribute to negative disease effects and could exacerbate root disease activity.

There is evidence that historical fire regimes on a dry end of a moist-forest zone (e.g., fire regime IIIa sites) probably functioned to keep Armillaria root disease activity within moderate levels, i.e., within a range of variation for this pathogen. Fire regime effects were likely related to many factors, including fire's role in maintaining early-seral, shade-intolerant tree species, by keeping stocking levels relatively low, and by favoring fungi antagonistic to Armillaria root disease (e.g. *Trichoderma* spp.; see Filip and Yang-Erve 1997, and Reaves et al. 1984, 1990).

Application of prescribed fire in a manner emulating an historical fire regime (on fire regime IIIa sites, for example) would be expected to provide some amelioration of Armillaria impact, but as was the case for noncommercial thinning, prescribed fire will not eradicate the disease from a moist-forest site (Filip and Yang-Erve 1997). But one caution here: trees affected by root disease are much more susceptible to fire-caused mortality than uninfected trees. Root disease infection can function as a predisposing agent, with fire filling the role of a secondary agent in this example. When fire damages uninfected trees, it then functions as the predisposing agent, allowing root disease to overwhelm the tree's compromised defense systems, become well established, and ultimately cause tree mortality. For these reasons, the presence or absence of root disease is used as an evaluation factor when calculating a predicted survival percentage for fire-damaged trees (Scott et al. 2002).

As described above for fire regime influences, tree species composition has an important influence on the ebb and flow of Armillaria root disease abundance and virulence on moist mixed-conifer sites. When stand-replacing disturbances are followed by regeneration of western larch, lodgepole pine, or western white pine, all of which are tolerant or resistant to Armillaria infection (table 10), the root disease inoculum in soil will diminish substantially. As the early- or mid-seral species eventually decline in abundance and are ultimately replaced by mid-seral Douglas-fir and late-seral grand fir, both of which are highly susceptible to Armillaria infection (table

10), the root disease will resurge, increase in virulence, and then spread to occupy more of the site or move onto adjacent sites.

Annosus root disease is often associated with high-elevation true-fir stands having a history of selective timber harvest (e.g., partial cutting). The primary host trees of this root disease are grand fir and subalpine fir.

Annosus root disease spreads by windborne spores infecting fresh stump surfaces – the fungus colonizes the stump, and then spreads to adjacent hosts when their roots come in contact with infected stump roots (Filip et al. 2006). Because of this spread mechanism, boron-containing products are typically applied to freshly cut stump surfaces within 24 hours of tree felling to prevent stump infection. True fir stumps less than 14 inches in diameter (9 inches for subalpine fir) are generally not treated because they will desiccate before the fungus can spread to adjacent trees.

Boron treatments are not effective for previously-infected trees, so boron is only applied when there is no history of multiple-entry treatments, and no annosus-caused stain or decay is present at the stump surface. A high frequency of annosus root disease is associated with seed-tree or shelterwood seed cuts completed in the 1970s and 1980s, an era before boron was routinely used to treat fresh stump surfaces. For moist grand fir sites where annosus root disease is a management concern, planting or favoring resistant western larch or pines is considered to be an effective strategy (Filip et al. 2006).

Management response to laminated root rot is similar to recommendations for the other root diseases: favor resistant or tolerant tree species whenever possible; manage as though the root disease organism is associated with the site (i.e., it is a long-term site component, not an occasional or transient visitor); and favor natural, site-adapted tree regeneration, rather than planting stock, when circumstances allow this approach.

Additional Information

Schmitt (2001) provides a good summary about root diseases of the Blue Mountains, including potential active management responses to their impact. A broad-scale summary of root diseases affecting forests of eastern Oregon and Washington is provided by Thies (2001). Thies and Sturrock (1995) summarize information about laminated root disease; Filip and Schmitt (1979) describe conifer susceptibility to laminated root disease. Agriculture Handbook 691 (Shaw and Kile 1991) provides a good synthesis about Armillaria root disease. A journal article about interactions between fire, insects, and pathogens (including root disease) is provided by Parker et al. (2006). Filip and Schmitt (1990) describe silvicultural options for diseased true-fir stands in eastern Oregon. Craig Schmitt, retired pathologist, recently summarized many of the interactions between diseases and moist upland forest (Schmitt 2010).

5.8 Mountain Pine Beetle in Lodgepole Pine

Mountain Pine Beetle *(Dendroctonus ponderosae)*
TREE SPECIES AFFECTED: Lodgepole pine and ponderosa pine.
DAMAGE CAUSED: Tree mortality; blue-staining of sapwood.
IMPACTS AND EFFECTS: Killed millions of lodgepole pines in the Blue Mountains during major outbreaks from 1905-15 and 1968-78; it was also active in both the lodgepole pine and ponderosa pine forest cover types following the late 1980s drought.
CONTROL MEASURES: Stocking-level control; chemical insecticides or attractants (pheromones) are occasionally used.
FOREST INSECT AND DISEASE LEAFLET: No. 2 (Gibson et al. 2009).

The mountain pine beetle is a native insect inhabiting pine forests of the Blue Mountains (primarily affecting lodgepole, ponderosa, and whitebark pines, with western white pine affected to some extent). It causes little impact to its host trees at low population levels, but when populations build quickly to an outbreak, the effects can be severe, and they occur at a landscape scale (fig. 35). Adult beetles bore through the bark and into a tree's cambium layer; larval feeding chambers (galleries) in the cambium destroy a tree's capacity to transport water and nutrients.

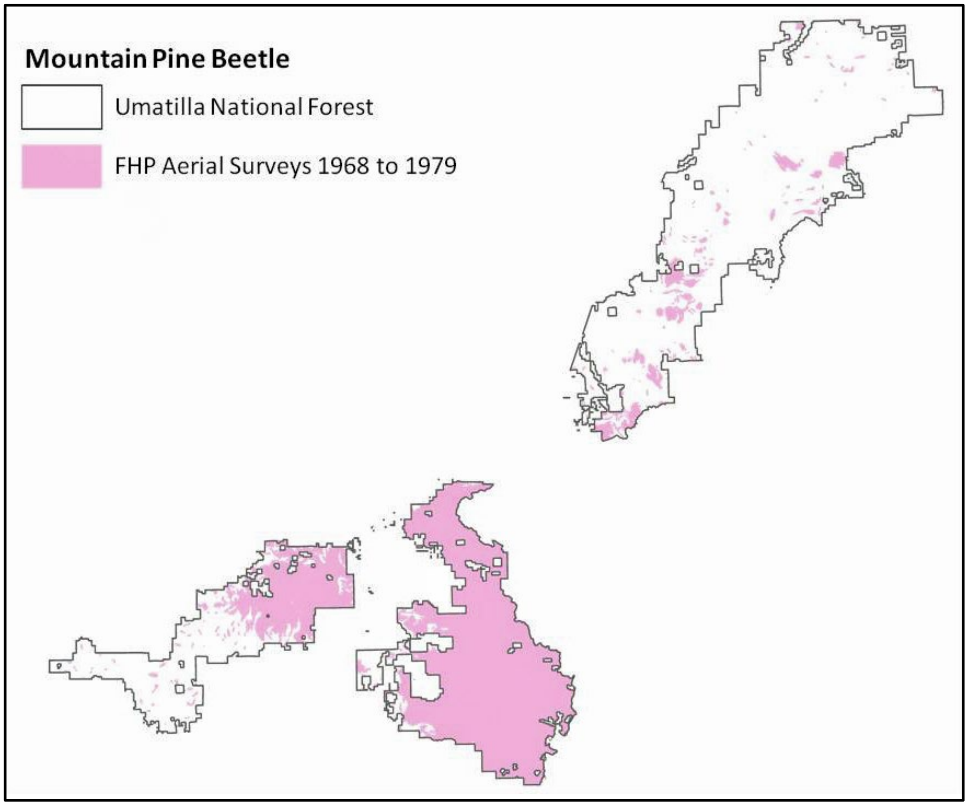


Figure 35 – Mountain pine beetle distribution for the 1970s outbreak on the Umatilla NF. At its height in 1976, the mountain pine beetle outbreak affected more than 375,000 acres of lodgepole pine forest type on the Forest.

The mountain pine beetle, in combination with associated blue-stain fungi, usually attacks and kills the less vigorous lodgepole pine trees, particularly if they are also large-diameter (≥ 9 inches dbh) and growing in relatively dense stands, such as those with a stand density index of 170 or greater (see fig. 6; Cochran et al. 1994, Peterson and Hibbs 1989).

Early Mountain Pine Beetle Outbreak (1905-1915)

An extensive mountain pine beetle outbreak occurred in lodgepole, ponderosa, and whitebark pines in the northern Blue Mountains and Wallowa Mountains in the early 1900s (beginning no later than 1905, according to historical accounts such as Burke 1990). By 1910, direct control measures were being implemented to combat this outbreak; "the first effort [anywhere in the United States] to control an outbreak of mountain pine beetle in lodgepole pine was made in 1910 and 1911 on the Whitman National Forest in northeastern Oregon" (Craighead et al. 1931).

On the Wallowa National Forest, the first serious mountain pine beetle outbreaks were reported in 1905, continued to grow to a peak in 1910, and then collapsed to quiescent levels by 1912. Another outbreak occurred not 25 miles away in the Powder River valley on the Whitman National Forest, beginning in lodgepole pine stands in 1906. It increased rapidly for the next 5 years, apparently culminating in 1912. After it peaked in lodgepole pine, the beetle began working in mixed stands of lodgepole and ponderosa pine, eventually killing considerable numbers of ponderosa pine trees. In 1913, large infestations were also known to exist on the Malheur and Umatilla National Forests, although they had not been studied to the same extent as the Whitman and Wallowa outbreaks (Pernot 1913a).

As of 1913, four bark-beetle control projects had been completed – two on the Whitman National Forest in cooperation with the Bureau of Entomology, and two on the Ochoco National Forest (Badger Creek drainage) under the direction of the Forest Service. The 1911 control project on the Whitman National Forest covered an area of 84,330 acres, cost $23,582, and resulted in treatment of 15,170 lodgepole pines (cut and burned), 8,015 ponderosa pines (cut and treated), and 3,388 ponderosa pines treated by peeling them standing. A gradual cessation of insect damage in untreated portions of the Whitman National Forest suggested that post-treatment declines were due more to natural factors than any direct result of the control operations (Pernot 1913a).

One consistent observation from the northeastern Oregon outbreaks was that mountain pine beetle always became a problem in the lodgepole pine type first, occasionally spilling out from there to infest other adjacent or intermingled pine species. During this historical era, extensive outbreaks were not observed in pure stands of ponderosa pine (Pernot 1913a). Limited tree killing was also noted in whitebark pine in the North Fork John Day River drainage and the Crane Creek area (Pernot 1913b). But overall, the early distribution of mountain pine beetle followed the range of lodgepole pine so closely that it could be considered a lodgepole pine beetle, at least in the Blue Mountains (Pernot 1913b).

In 1913, the mountain pine beetle infestation on the Whitman NF was centered in the Anthony Lakes area, the entire headwaters of the North Powder River, the upper watersheds of the Grande Ronde River, and the upper watersheds of the North Fork of the John Day River, especially in Crawfish Creek, on upper Trail Creek, and on Trout Creek and in the vicinity of Trout Meadows. The Crane Creek and Crane Flats areas, which are southern tributaries of the North Fork of the John Day River, were also heavily infested (Pernot 1913b).

On a sample acre near Anthony Lakes, the mixed stand consisted of lodgepole pine, Engelmann spruce, and subalpine fir, with lodgepole pine comprising 83% of the stand. Of the 194 lodgepole pines per acre in the 6 to 13 inch diameter class, 47% had been killed in the last 3 years, 35% were infested in August of 1913 by the 1912-1913 brood of beetles, and 18% were healthy and not yet attacked (Pernot 1913b).

As would be expected from these historical accounts, this early mountain pine beetle outbreak caused extensive tree mortality, as described here by R.E. Kan Smith (1912): "to ride through the lodgepole forests in the vicinity of Porcupine Ranger Station, T. 5 S., R. 37 E., W.M., that were infested in 1909-10 gives one the impression of an eastern hardwood forest in the dead of winter. The lodgepole all stands dead and bare, with here and there an occasional green tree of other species, such as larch, fir, etc. The earliest-attacked lodgepole is beginning to fall and it is only a matter of a short time when the great mass of it will fall."

The occurrence of bark beetles in whitebark pine was irregular and scattered, but mountain pine beetle was certainly associated with this tree species. Whitebark pines along the summit of the North Fork John Day River–Powder River divide were infested, as were trees near the summit of Bald Mountain (Pernot 1913b).

During the same era as the mountain pine beetle outbreak, Douglas-fir beetle was also active in mixed-conifer forests, especially in the area near White Pine and Austin, Oregon. The western larch borer was considered to be partly responsible for the death of a considerable percentage of the mature western larch in the Blue and Wallowa Mountains of northeastern Oregon. It was noted that dwarf-mistletoe infections were particularly severe in mature larches, and that western larch borer was found most abundantly in recent, mistletoe-killed trees (Pernot 1913b).

Mid-1970s Mountain Pine Beetle Outbreak

Quantified records of mountain pine beetle occurrence and spatial distribution are only available since 1947. Although one source characterizes the 1905-1915 mountain pine beetle outbreak as the largest known one to affect the Blue Mountains (Gast et al. 1991), an outbreak that began in 1968 near Johnson Rock (Carter 1976) grew quickly and impressively from there – by the autumn of 1975, almost 1½ million acres of lodgepole pine in the Blue Mountains had been, or were then, infested with pine beetles.

The 1975 aerial survey detection map showed a large contiguous concentration of mountain pine beetle activity stretching from southeast of Prairie City north to Meacham, Oregon; smaller activity centers were scattered around the periphery of this main region from west of Seneca, Oregon north to Asotin County, Washington. Much of the mature lodgepole pine type in the central Blue Mountains (encompassing what were then the Baker, Long Creek, North Fork John Day, Heppner, and La Grande ranger districts) was killed during this outbreak (fig. 35 shows the Umatilla portion of this outbreak area).

Note from the map depicted in figure 35 that the mountain pine beetle outbreak was concentrated on the southern half of the Umatilla National Forest; considerably less area was affected on the northern half of the Forest. One reason for less impact on the northern half was that many areas on the Pomeroy Ranger District had been affected by a series of large forest fires in 1960 (including lodgepole pine forest in the Clearwater and Godman areas) and, by the early to mid 1970s, the lodgepole pine host type in those areas had not yet had enough time to reach a tree size (8 inches diameter) highly susceptible to mountain pine beetle infestation.

One reaction to more than 1½ million acres of mountain pine beetle infestation was to prepare salvage timber sales to remove dead or dying lodgepole pines (fig. 36). For areas with steeper slopes, a dense road system was developed and the dead trees were removed by using short-span 'Idaho jammer' yarding systems (fig. 37). For flat ridgetop or plateau locations, short-boom tree shears (clippers) were used, but this harvest system required the equipment to approach each tree closely so the shear could grip and then clip the stem (fig. 38).

One result of the tree-shear harvest system was that most of each harvest site was traversed by heavy equipment, causing high amounts of soil compaction. Tens of thousands of acres of young lodgepole pine type on the Unity and Heppner ranger districts were treated by using a winged subsoiler (Tilth unit) to restore soil structure, which not only promotes proper hydrologic function (water infiltration, etc.) but also development of normal tree-root architecture. Trees with normally developed root systems are more stable and less susceptible to future episodes of 'wet noodling' (caused by large height-diameter ratios; see Evaluating Windthrow Hazard section, page 139), or uncharacteristic levels of windthrow.

Addressing Mountain Pine Beetle Susceptibility

This section provides a short review of interactions between stand density, thinning as a treatment to manage stand density, and bark beetles, focusing primarily on mountain pine beetle, a tree-killing insect currently causing unprecedented levels of impact in lodgepole pine forests of western North America (Bentz 2009). Section 5.10, Tree Susceptibility to Bark Beetle Attack, discusses bark beetle susceptibility in a broader context than just mountain pine beetle.

Figure 36 – Tree mortality caused by mountain pine beetle attacking lodgepole pines near Highway 244 east of Ukiah, Oregon (photograph acquired on September 6, 1977).

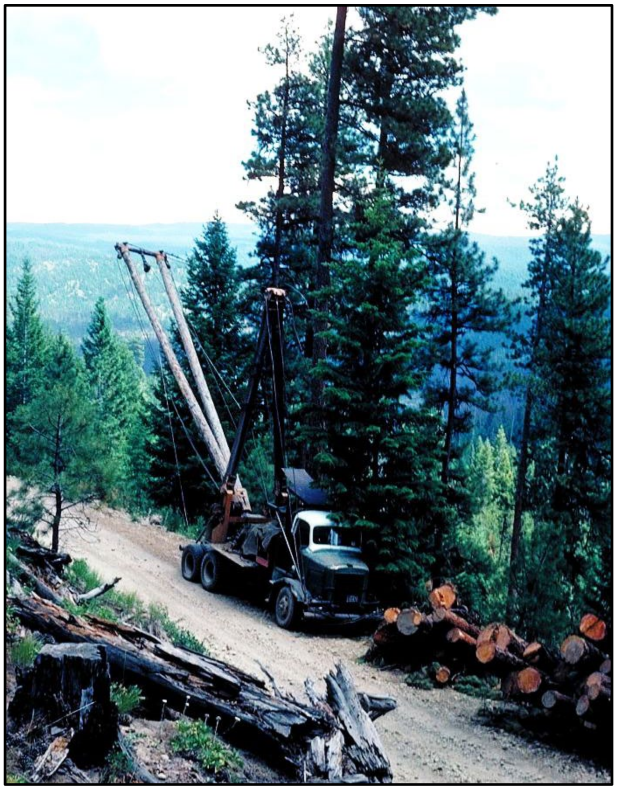


Figure 37 – Idaho jammer yarding system removing ponderosa pines killed by mountain pine beetle, near Johnson Rock fire lookout on the Wallowa-Whitman National Forest, in July of 1976. Although this photograph was taken adjacent to the North Fork John Day RD on the La Grande RD, similar equipment was also used on the Umatilla National Forest during this period. [Note that an Idaho jammer is basically just a mobile crane with tongs attached to the cable; the crane's boom is used to 'throw' tongs out into a cutting unit, where they are used to grasp logs and yard them to a roadside. Idaho jammer systems had a short reach, so a road system developed for their use was quite dense.]

Figure 38 – Tree shear harvest system (clipper) harvesting a lodgepole pine, Umatilla NF. This type of shear was used to harvest much of the beetle-killed timber during the mid-1970s mountain pine beetle outbreak, and because it has a relatively short articulating arm, the shear had to get close to each tree to remove it; this resulted in compacted soils for a high proportion of the harvested sites (Geist et al. 2008). The Heppner RD completed more than 10,000 acres of soil compaction remediation treatments (e.g., sub-soiling) in the late 1980s and 1990s to ameliorate shear-caused soil compaction.

Amman and Anhold (1989) found a negative correlation between SDI and tree mortality caused by mountain pine beetle (i.e., as SDI increased, tree mortality decreased), although this finding was assumed to reflect the fact that less dense stands contain a higher proportion of large-diameter trees with thick phloem, perhaps providing ideal bark-beetle habitat. Anhold et al. (1996) identified a zone of high susceptibility for lodgepole pine consisting of relative densities between 20 and 35 percent of maximum SDI, resulting in rapid growth to a large tree size, and quadratic mean stand diameters greater than 8 inches, reflecting a preponderance of trees with thick phloem. Both studies suggest that very dense lodgepole pine stands are unfavorable for mountain pine beetle, presumably because they contain a high proportion of trees with thin phloem, and these trees provide only marginal habitat for beetle broods.

Several other studies found that tree mortality due to mountain pine beetle was insignificant until a threshold stand density level was reached, at which point mortality could quickly escalate (Cochran 1992, Cochran and Barrett 1993, Mitchell et al. 1983, Oliver 1995, and others). Peterson and Hibbs (1989) analyzed previously published data from both thinned and unthinned stands in the Blue Mountains (see Mitchell et al. 1983), and they concluded that an SDI of 160-170 was the threshold density above which beetle-induced mortality became serious for lodgepole pine.

Thinning lodgepole pine increases tree vigor and resistance to mountain pine beetle attack (Mitchell et al. 1983); studies found that fewer trees were killed in heavily thinned areas when compared with lightly thinned or unthinned stands (Preisler and Mitchell 1993, Schmitz et al. 1989, Whitehead and Russo 2005). Waring and Pitman (1985) noted that the risk of a mountain pine beetle outbreak "can be greatly reduced by periodic thinning," and that improved bark-beetle resistance develops within three years of a thinning treatment. Once trees produce a physiological response to thinning (usually 3-5 years after treatment), their improved vigor promotes production of defensive chemicals enhancing beetle resistance (Christiansen et al. 1987, Franceschi et al. 2005, Kolb et al. 1998, Mitchell and Martin 1980, Perrakis and Agee 2006, Shrimpton 1978).

The relationship between stand density and mountain pine beetle susceptibility is complex, particularly for lodgepole pine ecosystems (Hindmarch and Reid 2001). Some of the thinning studies suggest that a change in stand microclimate (air temperature, light intensity, etc.) is the principal factor influencing mountain pine beetle attack because observed reductions in beetle activity occurred immediately after thinning, and this type of response is too quick to attribute to any tree vigor improvement resulting from thinning. The gregarious nature of bark beetles also suggests that the immediate post-thinning environment could be disrupting auditory cues or pheromone communication between beetles (Amman et al. 1988, Bartos and Booth 1994, Mitchell and Preisler 1991, Waring and Pitman 1985).

5.9 Douglas-Fir Beetle And Fir Engraver

Douglas-fir Beetle *(Dendroctonus pseudotsugae)*

TREE SPECIES AFFECTED: Douglas-fir; western larch (secondary).

DAMAGE CAUSED: Blue-stained sapwood; gray saprot; tree mortality.

IMPACTS AND EFFECTS: Attacks trees weakened by disease, drought, defoliation, fire, or another stressor; active following the early 1970s Douglas-fir tussock moth outbreak, the 1980-92 western spruce budworm outbreak, and the late 1980s drought.

CONTROL MEASURES: Salvage attacked and susceptible hosts; stocking-level control; manage green slash to prevent population buildups.

FOREST INSECT AND DISEASE LEAFLET: No. 5 (Furniss and Kegley 2014).

Fir Engraver *(Scolytus ventralis)*

TREE SPECIES AFFECTED: Grand fir and subalpine fir.

DAMAGE CAUSED: Brown-stained sapwood; top killing; tree mortality.

IMPACTS AND EFFECTS: Attacks trees weakened by drought, defoliation, or root disease; very active following the 1980-92 spruce budworm outbreak and late 1980s drought.

CONTROL MEASURES: Improve host vigor; treat root diseases and other stress-causing agents over which we have some control; stocking-level control; salvage damaged and susceptible hosts.

FOREST INSECT AND DISEASE LEAFLET: No. 13 (Ferrell 1986).

Douglas-fir beetle kills Douglas-firs by girdling them and introducing a bluestain fungus. Trees weakened by fire, disease, drought, defoliation, or other stresses are especially vulnerable to attack. In the Blue Mountains, Douglas-fir beetle caused considerable amounts of tree mortality on the heels of the 1980-1992 spruce budworm outbreak (Gast et al. 1991) – budworm stressed host trees and functioned as a contributing factor, but Douglas-fir beetle actually administered the coup de grace.

Older, dense stands are more susceptible to Douglas-fir beetle attack than younger, open stands, and the underlying geology (parent materials) also seems to exert an influence, particularly when beetles occur at endemic (rather than outbreak) population levels (Garrison-Johnston et al. 2003). As is the case for many bark-beetle species, high levels of stand density function as a predisposing factor for Douglas-fir beetle. "Bark beetles prefer densely-stocked stands" (Filip et al. 1996).

Douglas-fir beetle outbreaks are much more likely in older stands, particularly if radial growth has begun to slow and if intertree competition is severe (Berryman 1982, Shrimpton 1978). Furniss et al. (1981) reported that stands attacked by Douglas-fir beetle were both overstocked and overmature at the time of an outbreak. And, they also found that high levels of intertree competition (overstocking) were commonly associated with somewhat younger trees being attacked.

Fir engraver attacks white and subalpine firs weakened by defoliation, drought, root disease, or other factors causing tree stress. As was the case for Douglas-fir beetle, fir engraver caused tree mortality in the Blue Mountains in the late 1970s in response to the early 1970s Douglas-fir tussock moth outbreak (Wright and Berryman 1978), and in the late 1980s and early 1990s in response to a widespread spruce budworm outbreak (figs. 39 and 40) (Gast et al. 1991).

Filip et al. (1989) examined interactions among fir engraver attacks, stain fungi (*Trichosporium symbioticum*) carried by the beetle, and their grand fir host trees on the Umatilla National Forest. "They reported that resin production, a common defense response of beetle-attacked firs, was significantly greater in high-vigor trees, such as trees that had been thinned" (Filip et al. 1996).

Influence Of Good Versus Bad Rocks On Bark-Beetle Susceptibility

Reducing the amount of overmature Douglas-fir forest growing on 'bad' rock types was suggested as a tactic for reducing future tree mortality for northern Idaho areas experiencing an extensive Douglas-fir beetle outbreak in the late 1990s (Garrison-Johnston et al. 2003). This study found that Douglas-fir root biochemistry was significantly affected by potassium levels (Entry et al. 1991), which were reduced on 'bad' geologic rock types (and for their resulting soils).

Soil nutrient status influences moist forests on deep tephra (volcanic ash) deposits because ash mantles contain 60 to 90% volcanic glass, which has little nutritive value (McDaniel and Wilson 2007, McDaniel et al. 2005). But the nutrient status of ash soils can be quite different (more limited) than for loess soils, even though both are fine-textured and have high water-holding capacity. The growth of Douglas-fir trees established on loess soils, for example, may not be limited by potassium, sulfur, or boron, three important nutrients for forest growth (White et al. 2012).

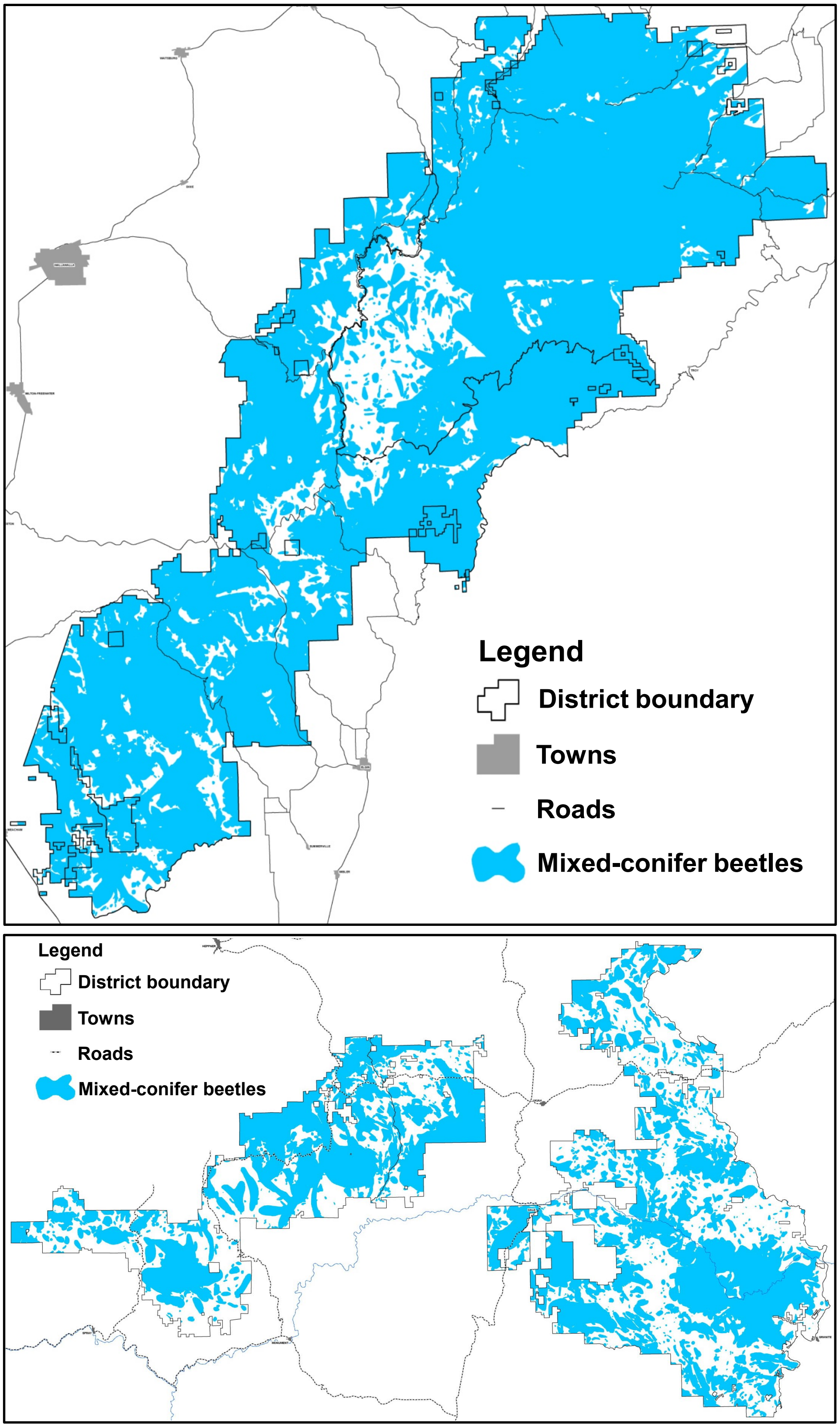


Figure 39 – Geographical distribution of mixed-conifer beetles (Douglas-fir beetle and fir engraver), 1947-2010, for the Umatilla National Forest.

Figure 40 – Grand fir trees killed by fir engraver beetles (from Powell 1994). Defoliation, root disease, drought, dwarf mistletoes, overstocking, and other stressors increase a tree's susceptibility to bark beetle attack (Filip 1994, Filip and Schmitt 1990). Fir engraver and Douglas-fir beetle caused widespread damage in the Blue Mountains during the late 1980s and early 1990s (fig. 39), primarily in response to stress caused by budworm defoliation and drought (Gast et al. 1991, Powell 1994). High-vigor trees are better able to ward off insect and disease attacks by producing resins, phenols, terpenes, and other defensive chemicals (Christiansen et al. 1987, Coyea and Margolis 1994, Filip et al. 1989, Kolb et al. 1998, Langenheim 1990, Mitchell et al. 1983, Pitman et al. 1982, Safranyik et al. 1998, Sartwell 1971, Shrimpton 1978, Vité 1961, Waring 1987).

It was also found that Douglas-fir forests growing on 'bad rocks,' from a site-nutrition perspective, are more likely to contain trees experiencing chronic nutrient stress (Shaw et al. 1998). These high-stress environments then function as long-term refugia for Armillaria root disease. And as might be expected, Douglas-fir trees infected with root disease were more susceptible to successful bark-beetle attack because they were unable to produce sufficient defensive chemicals (resins, etc.) to ward off attack by not only the root disease, but also by Douglas-fir beetle (Garrison-Johnston et al. 2003).

Nutrient limitations, particularly for areas with bad rock types, can exacerbate physiological stresses experienced by Douglas-fir or grand fir forests, so it may be important to keep stocking levels on bad-rock sites at very low levels as a strategy for addressing broad-scale susceptibility to defoliating insects, bark beetles, and other biotic agents (Carlson et al. 1985, Cates et al. 1983, Cochran 1998, Mandzak and Moore 1994, Powell 1999b, Shen et al. 2001, Stoszek 1988b). And, we might expect increased insect or disease activity, as related to warmer temperatures and earlier spring snowmelts predicted by climate change scenarios (Westerling et al. 2006), to manifest first on bad-rock sites. [Note that guidelines relating specific rock types to a 'good rock/bad rock' classification framework have been published for the Blue Mountains: see Garrison-Johnston 2010, and Garrison-Johnston and Johnson 2008.]

5.10 Tree Susceptibility To Bark Beetle Attack

In a properly functioning ecosystem, the distribution of defoliating insects, bark beetles, and pathogenic fungi is typically limited to just a few stressed trees (within an individual stand), or to a few stressed stands within a larger forested landscape. Many different rating systems have been developed to help predict whether a landscape would be expected to sustain healthy forest conditions in the future, or whether it might evolve to a situation where heightened levels of insect and disease activity are common. Perhaps the most common rating strategy involves an assessment of hazard or susceptibility to insect and disease attack.

Systems for rating susceptibility were developed for central Idaho, the Blue Mountains, and other portions of the interior Pacific Northwest (Garrison-Johnston et al. 2003, Steele et al. 1996, Schmitt and Powell 2005). Susceptibility refers to the potential for a disturbance event (wildfire, insect outbreak, disease epidemic, etc.) to occur, and it is determined by evaluating intrinsic stand or site characteristics such as species composition, forest structure, and tree density (Schmitt and Powell 2005).

Since many insects or diseases respond to stand density in one way or another, a characterization of stand density is often included in hazard or susceptibility rating systems. Stand density was one of the rating factors for a variety of forest insects, including Douglas-fir beetle, mountain pine beetle in lodgepole pine, spruce beetle, western pine beetle and mountain pine beetle in ponderosa pine, fir engraver, and western spruce budworm (Steele et al. 1996, Schmitt and Powell 2005).

To preclude serious tree mortality from mountain pine beetle, western dwarf mistletoe, and perhaps western pine beetle, stand density should be maintained below the self-thinning zone (fig. 41) (Barrett and Roth 1985, Cochran et al. 1994). In response to specific concerns about mountain pine beetle susceptibility, adjustments were made to the upper limit of the management zone stocking level for lodgepole and ponderosa pines (Cochran et al. 1994, Powell 1999b). [Note that the upper limit of the management zone corresponds to the lower limit of the self-thinning zone (dashed line) in fig. 41.] Thinning is a silvicultural activity often used to keep a stand's density below the self-thinning zone.

By reducing the number of trees on a site, thinning provides more sunlight, water, and mineral nutrients for the residual trees (Donner and Running 1986). These changes improve the physiological vigor of residual trees, increasing their seasonal energy activity and growth. Trees with increased physiological vigor produce more resin (pitch) and are better able to resist the effects of defoliators, bark beetles, and fungi (figs. 40 and 42) (Coyea and Margolis 1994, Filip et al. 1989, Kolb et al. 1998, Mitchell et al. 1983, Pitman et al. 1982, Safranyik et al. 1998, Sartwell 1971, Shrimpton 1978, Vité 1961). Trees with high vigor also grow faster than trees with low vigor, and slow tree growth has been associated with higher susceptibility to insect and disease damage, including tree mortality (Bleiker and Uzunovic 2004).

Trees respond to thinning by producing more foliage and developing a higher level of photosynthate reserves, both of which improve their capacity to resist or recover from insect and disease attacks (Kolb et al. 1998, Franceschi et al. 2005). Photosynthate refers to carbohydrates created during photosynthesis. Trees allocate photosynthate in an order of precedence, but the rank is not absolute (Loehle 1988, Oliver and Larson 1996, Waring and Pitman 1985, Waring and Running 1998):

1. Maintenance respiration (first).
2. Producing fine roots and foliage.
3. Flower and seed production.
4. Height, branch, and large-root growth.
5. Diameter growth.
6. Insect and disease resistance (last).

After a tree becomes crowded and loses vigor, its total photosynthate production declines, and a higher proportion of the available reserve must be allocated to maintenance respiration. Since insect and disease resistance is the lowest priority for photosynthate allocation, it is the first need to be abandoned when a stressed tree must shift some of its photosynthate to respiration or any other need higher in the hierarchy. "Stress often translates to increased pest problems" (Perry 1988).

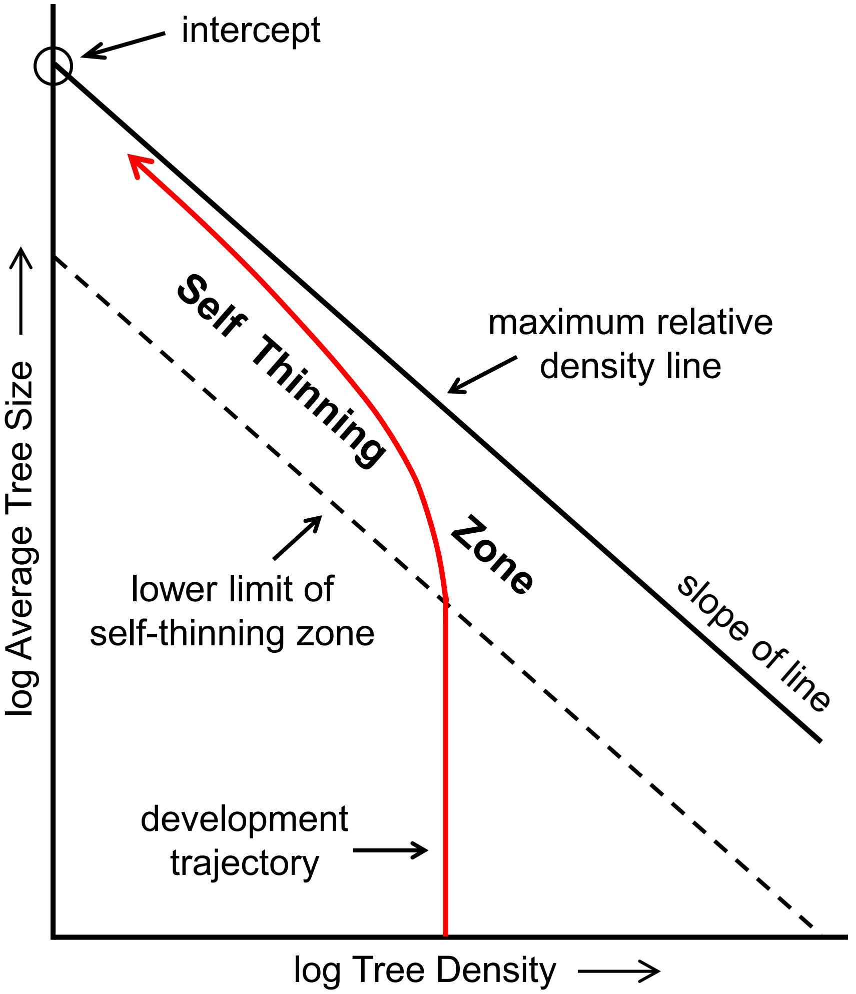


Figure 41 – Schematic representation of important concepts related to self thinning (after Jack and Long 1996). This figure portrays an upper boundary or maximum size-density line which varies not only by tree species, but by plant association for the same species. Variation by plant association can be thought of as 'environment boundary lines' (Sackville Hamilton et al. 1995) because each association represents a slightly different biophysical environment (Powell et al. 2007). The upper-boundary line is a logarithmic relationship with a negative slope (it's sloping downward rather than upward), which means that more trees are associated with a smaller size and less trees with a larger mean size. This negative relationship between mean size and density exists for all self-thinning plant populations, regardless of their life form (tree, shrub, herb) (Westoby 1984).

The red self-thinning trajectory line shows a typical development pattern for an even-aged tree stand eventually experiencing density-dependent mortality. After self-thinning begins, a stand is constrained by the upper boundary and its future trajectory will remain below, but track along, this line. The dashed line shows the lower limit of the self-thinning zone (also known as the upper limit of the management zone); stands beyond this dashed line experience density-related, competition-induced mortality. For many tree species, the lower boundary coincides with 75% of the full-stocking level. Suggested stocking levels moist-forest plant associations of the Blue Mountains are provided by Cochran et al. (1994) and Powell (1999b).

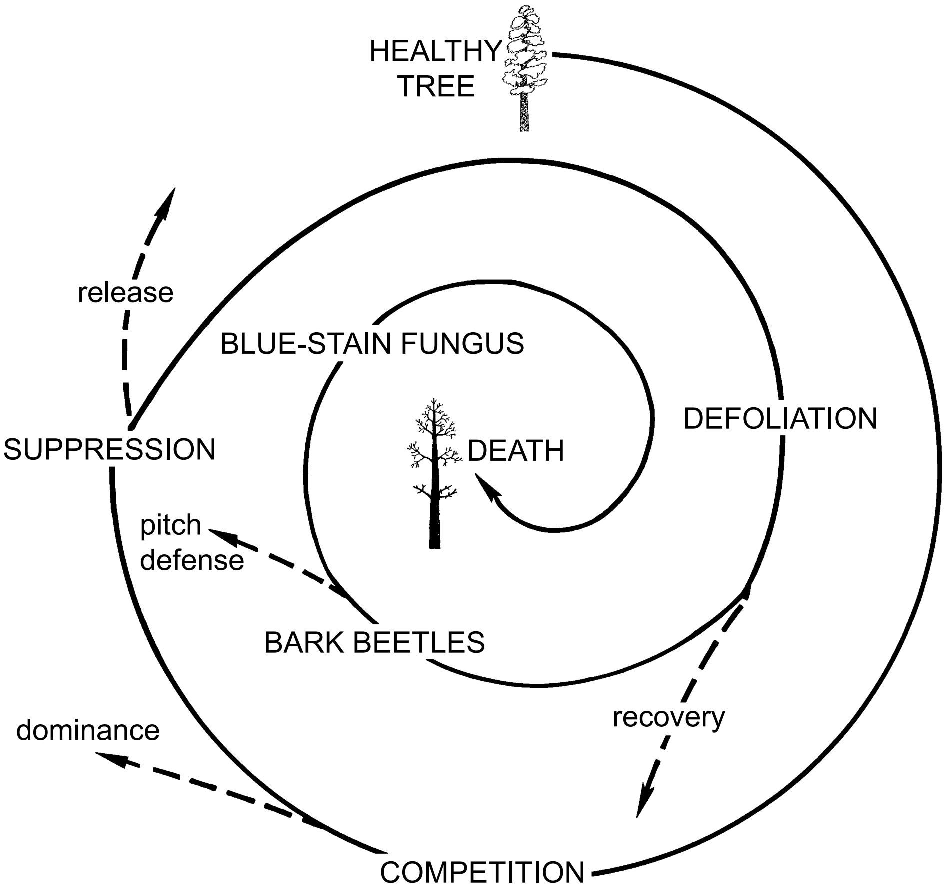


Figure 42 – Death spiral (adapted from Franklin et al. 1987). In this example, a slightly taller tree suppresses a shorter but otherwise healthy tree. If not released from competition, then high stand density functions as a predisposing factor for tree damage or death from defoliating insects. Once partially defoliated, the weakened tree is attractive to bark beetles such as Douglas-fir beetle (Wickman 1978) that carry blue-stain fungus. The fungus blocks water and sap movement in the tree and causes foliage desiccation. In this model of tree decline, suppression is a predisposing factor; defoliating insects and bark beetles are inciting or contributing factors (Pedersen 1998).

Plant Defensive Chemicals And Bark-Beetle Susceptibility

Dense tree stands exist in a sort of perpetual physiological drought because there is not enough soil moisture to meet the water needs of all trees; silvicultural treatments are used to alleviate moisture stress, and allow residual trees to survive and continue growing. Trees in low-density stands exposed to climatic drought would respond favorably to increased soil moisture when precipitation improved, but this outcome would not occur for high-density stands subjected to the chronic, competition-induced stress associated with physiological drought.

“A considerable body of evidence indicates that environmental stress (primarily water stress) renders pines, spruces, and firs more susceptible to bark beetle invasion by a general reduction in oleoresin flow, and, at least in some conifer species, the induced response to bark beetle attack is diminished under stress conditions.”

"Both light and water stresses diminish the ability of grand fir trees to produce oleoresin, thereby suppressing the efficacy of this line of defense" against bark beetles and other attacking organisms (Lewinsohn et al. 1993). And for mixed-conifer forests in western Montana, recent research suggests that a combination of thinning and prescribed fire produced the highest resin flow response in the residual trees (i.e., the thin-and-burn treatment produced a higher resin flow response than the thin-only treatment) (Six and Skov 2009).

According to research conducted in ponderosa pine ecosystems in the southwestern U.S., high-density stands experiencing physiological drought are relatively unable to take advantage of favorable precipitation years because they are already severely stressed from intertree competition (McDowell et al. 2006). A similar response would be expected for northeastern Oregon and southeastern Washington because precipitation conditions are seldom optimum there, particularly as compared to the wet forests of western Oregon, where precipitation is much more abundant. Thus, active management of Eastside forests should attempt to create and maintain low-stress conditions in terms of soil moisture availability, and this strategy will become even more relevant in a warmer and dryer future created by climate change.

Since many bark beetle species are adept at preferentially seeking out stands experiencing physiological drought (Filip et al. 1996), perhaps by interpreting the chemical cues emitted by stressed trees, thinnings could effectively alleviate a physiological drought condition while simultaneously reducing bark-beetle vulnerability by modifying a stand's chemical-cue signature.

Additional Information

A good synthesis paper about the influence of thinning and other density management practices on prevention or control of bark beetle infestations in western conifer forests is provided by Fettig et al. (2007). A similar summary paper about interactions between fire, insects, and pathogens is supplied by Parker et al. (2006). Useful syntheses about relations between fuels, fire behavior, and beetle-attacked forests are provided by Bentz et al. (2010), Hicke et al. (2012), Jenkins et al. (2008, 2012), and Simard et al. (2011).

There is a rich and varied literature examining interactions between bark beetles and active forest management practices (in addition to citations in sections 5.8 to 5.10, the References section of this white paper provides many other sources about this topic). Gast et al. (1991) provides detailed discussions for major bark beetles of the Blue Mountains, including possible management response to their impact. British Columbia's Bark Beetle Management Guidebook (British Columbia Ministry of Forests 1995a) is also a useful reference.

Don Scott, retired entomologist, recently prepared a synthesis document describing interactions between insect disturbance agents and moist forests of the Blue Mountains (Scott 2010).

5.11 Wildfire

> **It may not be too fantastic to consider the day when we shall wonder whether fire, in some form at least, has not been a friend masquerading as a foe.** *W.J. Bloomberg, Fire and Spruce, 1950*

Fire is a major disturbance process shaping the composition, structure, and density of many forest, shrub, and herb communities (Brown and Smith 2000). For at least the last twelve thousand years, wildfire has been a primary disturbance process influencing plant succession in the interior Pacific Northwest (Habeck 1976). Fire regime is a useful concept for understanding fire's historical role in the ecosystem, including the effects of aboriginal burning (Agee and Maruoka 1994). "A fire regime is intended to characterize the features of historic, natural fires that have been typical for a particular ecosystem or set of ecosystems" (Pyne et al. 1996).

Fire regime characterizes a fire environment – how fire functions and its effect on the vegetation. For forested ecosystems, at least six factors are used to describe a fire regime (Agee 1993, Pyne et al. 1996):

- Fire frequency (how often fire occurs);
- Fire intensity (typically expressed as flame length);
- Fire severity (how much of the overstory vegetation is killed);
- Burned area (annual area burned, as a percentage);
- Fire size (typical fire extent, in acres);
- Fire timing (season when fire typically occurs).

Five coarse-scale fire regimes have been defined for the United States. They are used with a national assessment protocol to evaluate how existing vegetation conditions depart from reference conditions. This fire regime condition class (FRCC) protocol defines five fire regime groups by using two factors: fire frequency and fire severity (table 11, Barrett et al. 2010).

Table 11 summarizes selected characteristics for four of the five fire regimes; fire regime V is not included in table 11 because it is uncommon in the Blue Mountains. Figure 43 shows the distribution of historical fire regimes for the Umatilla National Forest in the 1880s. This figure was derived from General Land Office survey notes by interpreting the witness or bearing tree information recorded at section corners and along section lines (Powell 2008c).

Moist upland forests have a mixed-severity fire regime (Fire Regime III in table 11). At the coarse level of the FRCC protocol, fire regime III includes a wide fire frequency range of 35 to 200+ years. At the finer scale of the Blue Mountains, fire regime III is believed to have three variants or sub-regimes: IIIa (fire frequency of 50 years or less), IIIb (fire frequency of 51 to 100 years), and IIIc (fire frequency of greater than 100 years) (Evers 2002). Figure 44 displays fire return interval ranges by fire regime, with fire regime III split into its a, b, and c variants.

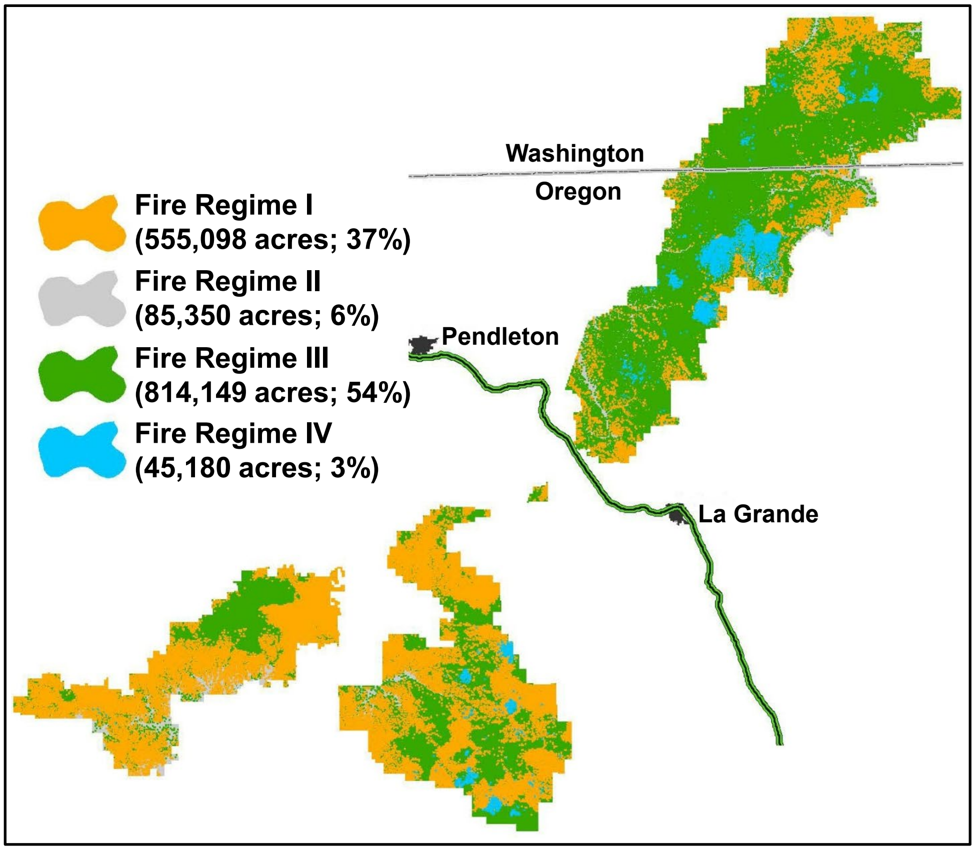


Figure 43 – Historical fire regimes of the Umatilla National Forest in the 1880s (wide green line shows the current location of Interstate 84 between Pendleton and La Grande). This map was derived from General Land Office survey notes collected primarily between 1879 and 1887. Bearing tree information from section corners and quarter-corners were analyzed to determine the presence and density of individual tree species, and by using geostatistical techniques to spatially interpolate between these points. Combinations of tree species were examined by ecologists and assigned to their corresponding ecological systems, a system used by the Nature Conservancy for vegetation classification (Comer et al. 2003). The ecological systems were cross-walked to their respective fire regimes by the author of this white paper. Powell (2019) provides detailed information about how the GLO survey notes were used to prepare an historical vegetation map for the Umatilla NF.

Table 11: Selected characteristics for historical fire regimes of the Blue Mountains.

	HISTORICAL FIRE REGIMES *			
Fire Regime Characteristic	**I**	**II**	**III**	**IV**
Fire return interval (mean; in years)[1]	< 25	< 35	**35-100+**	35-100+
FRCC: fire frequency interval[2]	0-35 years	0-35 years	**35-200+ yrs**	35-200+ years
Fire severity on upper canopy layer[3]	Low	Replacement	**Mixed**	Replacement
Upper canopy layer mortality[3]	≤ 25%	> 75%	**26-75%**	> 75%
FRCC: fire severity name[2]	Low/Mixed	Replacement	**Mixed/Low**	Replacement
Fire intensity adjective[4]	Low	Low-Moderate	**Moderate-High**	High
Fireline intensity (flame length; feet)[5]	< 3	< 3	**3-10**	> 10

	HISTORICAL FIRE REGIMES *			
Fire Regime Characteristic	**I**	**II**	**III**	**IV**
Fuel component driving fire spread[4]	Surface	Surface	**Surface/canopy**	Canopy
Ecosystem example[4]	Ponderosa pine	Grassland/ shrub	**Moist mixed-conifer forest**	Subalpine forest
Historical burned area (percent)[6]	75	5	**15**	5
Estimated fire size (acres)[7]	1-3,000	Unknown	**1-10,000**	1-5,000
Measured fire size (acres)[8]	2,950	Unknown	**900**	Unknown
Fire size variability (acres; min-max)[9]	50-19,960	Unknown	**250-1,940**	Unknown
Fire timing (seasonality)[10]	Summer and autumn	Spring and summer	**Summer and autumn**	Summer and autumn

* **Historical fire regime** (FR) is a characterization of the historical combination of fire frequency and severity under which plant communities evolved and were maintained (Schmidt et al. 2002). Five fire regimes are currently recognized (Barrett et al. 2010):

Fire regime I: 0- to 35 year fire frequency; low or mixed severity on upper canopy layer.
Fire regime II: 0- to 35 year fire frequency; replacement severity on upper canopy layer.
Fire regime III: 35- to 200 year fire frequency; mixed or low severity on upper canopy.
Fire regime IV: 35- to 200+ year fire frequency; replacement severity on upper canopy layer.
Fire regime V: 200+ year fire frequency; replacement severity on upper canopy layer.

<u>Notes</u>: FR V is uncommon in the Blue Mountains and not included in the table. FR III is shaded in gray because moist forests, the subject of this white paper, are assigned to FR III.

[1] **Fire return interval** (years) is the frequency between successive fire events; table data is based on Hall (1976), Heyerdahl and Agee (1996), Maruoka (1994), and Schmidt et al. (2002).

[2] **FRCC** (fire regime condition class) is a process for evaluating whether current conditions depart from historical reference conditions and, if so, the magnitude of the departure; the FRCC frequency and severity names, by fire regime group, are taken from Barrett et al. (2010). When more than one fire severity is shown (Low/Mixed), the first one listed is predominant.

[3] **Fire severity on upper canopy layer** is the effect of fire on dominant plants: no more than 25% of upper canopy layer plants are killed by low-severity fire, whereas 75% or more are killed by replacement fire; mixed-severity fire has survival percentages between these values (the 25% and 75% thresholds were established by FRCC; see Barrett et al. 2010, page 99).

[4] **Fire intensity, fuel component,** and **ecosystem example** were taken from Keeley et al. 2009 (table 1 in that source). When more than one intensity adjective is shown (Moderate-High), the first one is assume to be predominant.

[5] **Fireline intensity** refers to the energy release rate of a fire. Since intensity is generally proportional to flame length, fireline intensity is frequently expressed as a flame length, in feet. Table data were taken from Agee (1996b).

[6] **Historical burned area** is an estimate of annual burned area (percent) for the Blue Mountains area prior to Euro-American settlement; table data were adapted from Agee (1996b).

[7] **Estimated fire size** provides an indication of average wildfire extent (in acres) for the Blue Mountains, as derived using an expert panel approach involving 50 employees from the Malheur, Umatilla, and Wallowa-Whitman National Forests (Johnson 1993).

[8] **Measured fire size** provides an indication of average wildfire extent (in acres) from a Blue Mountains fire history study (Heyerdahl and Agee 1996, Heyerdahl 1997).

[9] **Fire size variability** shows how historical wildfire extent varied (in acres) from a Blue Mountains fire history study (Heyerdahl and Agee 1996, Heyerdahl 1997). Note that the fire size variability characteristic might have been influenced by the number of fires sampled (fire regime I included 210 fires, but fire regime III included only 8 fires), and because the mapped fire extent was truncated at the study area boundary for some of the sampled fires.

[10] **Fire timing** refers to the typical season of fire. Table data were taken from Agee (1996b).

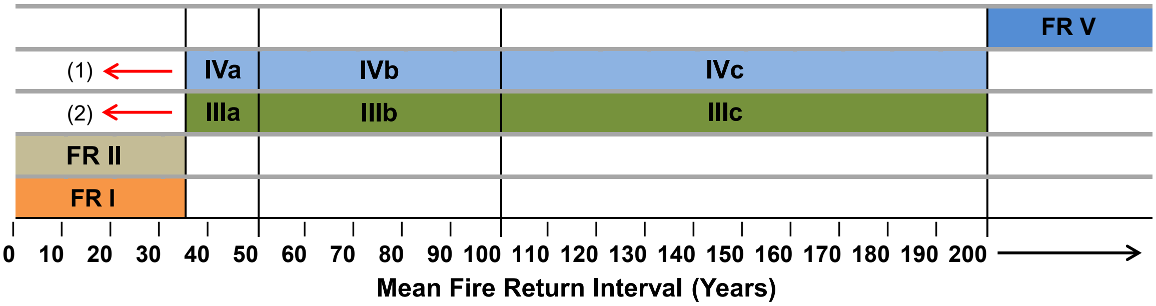


Figure 44 – Distribution of fire return interval ranges for fire regimes (FR) of the Blue Mountains. When Stephen Barrett examined mean fire return intervals (MFRI) for mixed-severity fires of the northern Rocky Mountains, he found that fire regime IIIa had a MFRI of 32 years, fire regime IIIb had a MFRI of 73 years, and fire regime IIIc had a MFRI of 135 years (Barrett 2004b: table 1, p. 33). Barrett compiled a database of published and unpublished fire history studies for the northern Rockies (his study area includes the Blue Mountains), and he found that 60% of the area experienced mixed-severity fire, with only 13% of the area having nonlethal fire and the remainder (27%) with lethal (stand-replacing) fire. **Notes for (1) and (2)**: The range of mean fire return interval for fire regimes III and IV is not shown as extending below 35 years, which agrees with national policy (Barrett et al. 2010). However, it is certainly possible for this to happen, as demonstrated by Barrett's study of the mixed-severity fire regime in the northern Rocky Mountains – his fire regime IIIa samples had a MFRI of less than 35 years (32 years, specifically) (Barrett 2004b).

As its name implies, the mixed-severity fire regime features a mix of fire severities, ranging from low severity (underburns) to patches of replacement severity (crown fire in some circumstances), all occurring as an intricate mosaic within a single fire perimeter. "A single fire may create patches of several types, consuming some stands in intense crown fires, burning others with cooler surface fires, and leaving unburned islands within burned areas" (White et al. 1999).

Conceptually, a mixed-severity fire regime could result from at least three different scenarios (and all three scenarios probably occur somewhere in North America):

1. Many trees are killed by mostly surface fire, but quite a few others of fire-resistant species or large size survive, resulting in upper canopy layer mortality of 26 to 75% (table 11). In some sources, this situation is referred to as a 'moderate' fire regime because it results in moderate amounts of overstory tree mortality. This result occurs not because intensity (flame length) varies across a burn area, causing variable mortality – it reflects varying fire tolerance due to a mix of thick- and thin-barked species, or because trees of widely varying size within a single species (seedlings/saplings to mature veterans) occur in an area.
2. Severity between successive fires on the same area varies temporally between low and mixed. One fire event may cause overstory mortality of less than 25%, whereas the next fire could kill 26 to 75% of the overstory trees. In some sources, this regime is termed 'variable' because it alternates between more than one severity class. Note that this variable fire regime is being found more often than previously thought for ponderosa pine forests of the southern Rocky Mountains

(Colorado Front Range and Black Hills) and interior Pacific Northwest (Baker and Ehle 2001, Lentile et al. 2006, Sherriff and Veblen 2006).

3. More than one severity type occurs within the same fire perimeter. This variant is the classic incarnation of mixed-severity fire defined in the paragraph introducing this section because a mix of fire severity occurs, ranging from low in some areas to patches of replacement in others, and all of the severities occur as an intricate mosaic within a single fire perimeter (and during a single fire event). The primary sources of this variation are driven by local, fine-scale, bottom-up fire behavior factors: topography and fuels.

Variability In The Mixed-Severity Fire Regime

Historically, low-to-moderate severity fires were an important component of the mixed-severity regime (Agee 1993, Brown and Smith 2000) – these are FR IIIa and IIIb sites (figs. 45-47). Research in many portions of the interior Pacific Northwest has often shown that mixed-severity regimes have more variability in fire frequency and severity than previously thought. And, it is interesting that variability was even encountered for some of the coldest and wettest sites in FR III – the FR IIIc sites – as described in the next paragraph for subalpine fir and silver fir sites in the eastern Cascade Mountains of central Washington.

"Fire-return intervals for the subalpine fir zone within the study area were historically more frequent and less severe than those ascribed to this vegetation zone in other regions. Unpublished fire history data from 24 subalpine and silver fir series stands in the Entiat watershed, about 65 miles north and somewhat west of the study area also corroborate that, east of the Cascades Crest, fires were not uniformly infrequent, stand replacing events. Of these 24 stands, 13 had fire return intervals of 100 years or less. Seven had fire return intervals between 100 and 200 years; only 4 showed fire return intervals greater than 200 years" (Camp 1999).

Results from Camp (1999) suggest that, in a manner similar to recently described variability for low-severity fire regimes (item 2 on previous page describes fire-regime variability for ponderosa pine forests), there may have been more variability in fire return interval for mixed- and high-severity regimes (e.g., fire regimes III, IV, and V) than was previously appreciated.

A high amount of variability associated with the mixed-severity regime (FR III) led to this recommendation: "the mixed severity bin is large, spanning fires that range from surface to crown fire dominated. Leaving the existing mixed severity fire class intact probably has limited utility. Instead, it would be useful to managers if fire and landscape ecologists explored the mixed severity fire continuum and erected finer classes reflective of the comparative roles of surface and stand replacing fires, thereby giving managers more insight about how they might vary and distribute management intensities" (Hessburg et al. 2007, p. 21).
[This recommendation makes a compelling case for establishing subregimes for a mixed-severity regime, such as fire regimes IIIa, IIIb, and IIIc shown in fig. 44.]

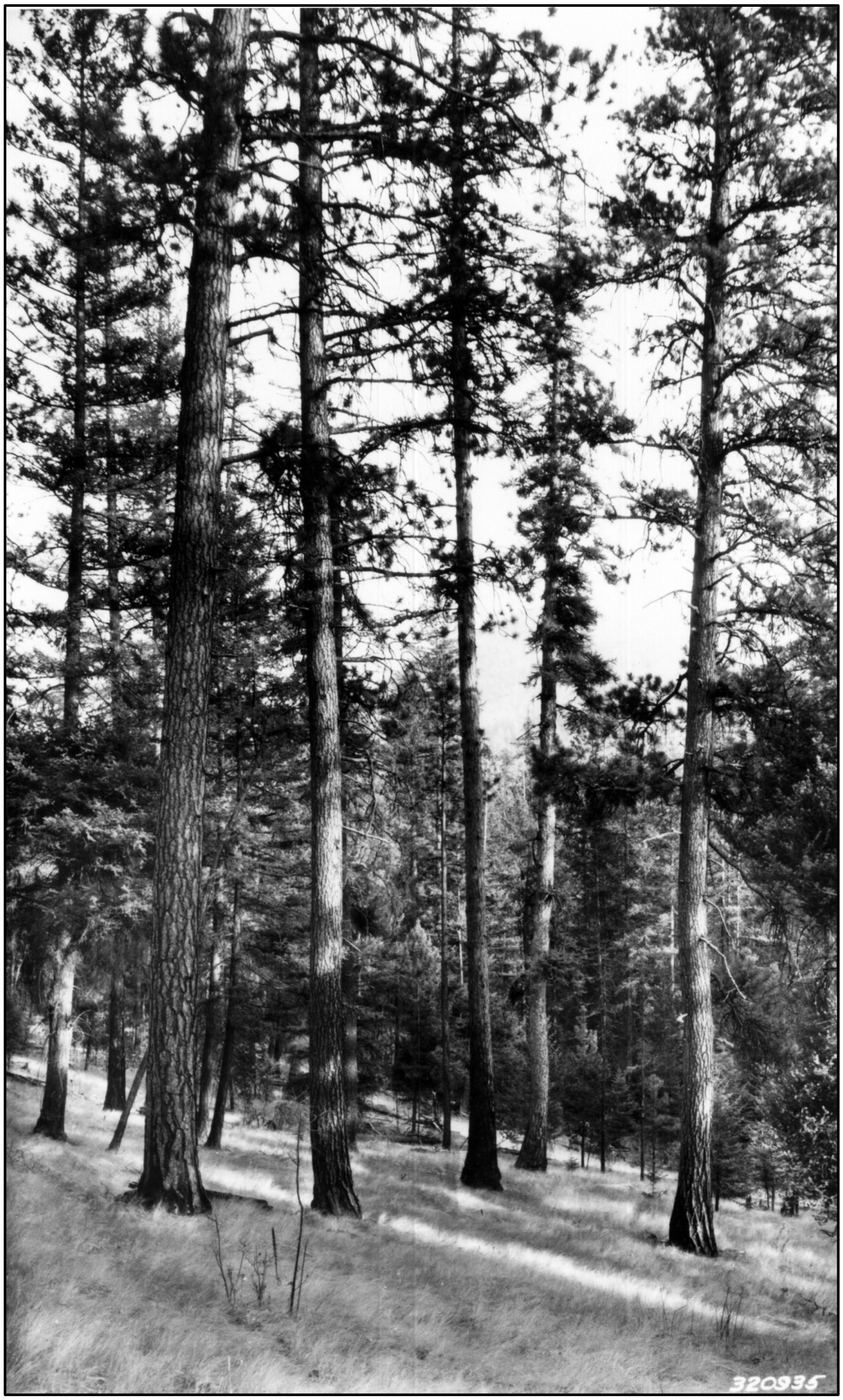
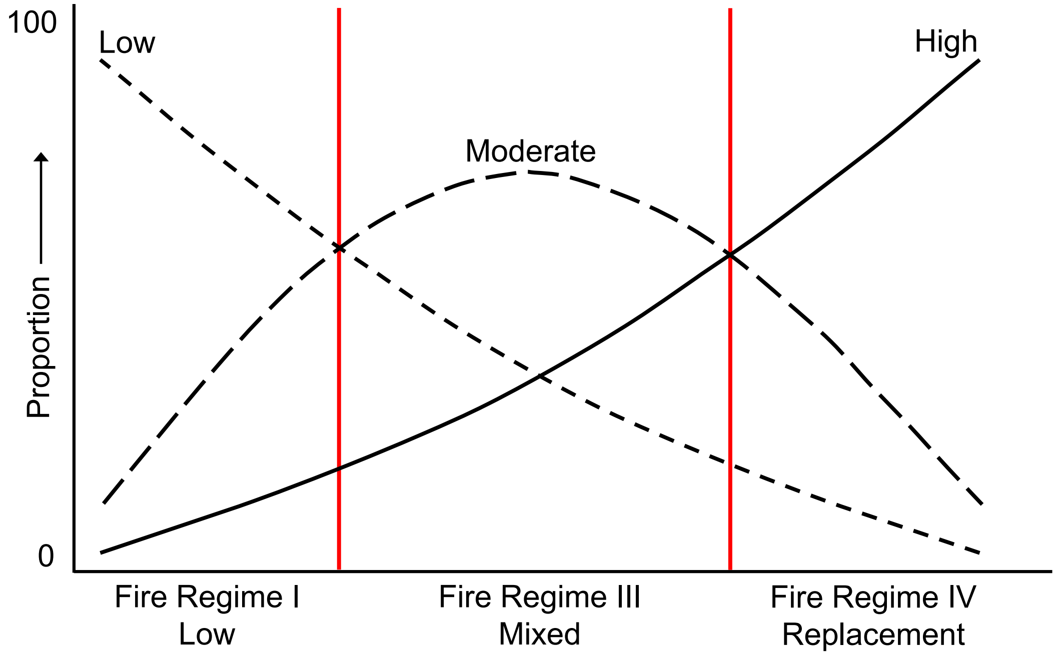


Figure 45 – Open stand of ponderosa pine and western larch (from Cowlin et al. 1942; photograph by Ray Filloon in 1936). Open, mixed-species stands containing western larch and ponderosa pine were fire influenced. Ecologically, stands similar to this one occur at the moist end of the dry-forest zone (fire regime I), or at the dry end of the moist-forest zone (fire regime IIIa). Note that a large-diameter western larch in the middleground (right of center) has obvious brooming caused by larch dwarf mistletoe parasitism.



FIRE SEVERITY PERCENTAGES BY FIRE REGIME

	FIRE REGIME I	FIRE REGIME III	FIRE REGIME IV
Low:	60-90	20-50	10-20
Moderate:	20-60	50-70	20-60
High:	10-20	20-50	60-90

Figure 46 – Predicted percentages of fire severity by fire regime (adapted from Agee 1998, and as reported in Powell 2017). Low, moderate, and high severity are defined in table 11, and they reflect low (< 25%), moderate or mixed (25-75%), and high or replacement (> 75%) mortality effects on the overstory tree cohort. Note: Agee (1996b) defined three fire regimes for the forested portion of the Blue Mountains: low (FR I), moderate (FR III), and high (FR IV). His estimate of the historical occurrence of the three fire regimes, as based on the distribution of historical forest types, is: Low – 80% of forested area; Moderate – 15% of forested area; and High – 5% of forested area.

Other areas in the western U.S. also have short fire return intervals (FRI) for relatively moist, mixed-conifer sites. Consider the Illilouette Creek basin in Yosemite National Park – the basin has a Mediterranean climate with cool, moist winters and warm, dry summers (precipitation averages 100 cm and occurs predominantly as snow).

The historical fire regime consisted primarily of frequent surface fires, with a mean FRI of 6.3 years and a fire rotation (the length of time required to burn a cumulative area equal to the basin's area) of about 25 years. But species composition associated with this fire regime is interesting because forests in Illilouette Creek basin are dominated by Jeffrey pine, white fir, red fir, and lodgepole pine, and most fire ecologists would not expect true firs and lodgepole pine to be associated with a surface fire regime featuring a mean FRI of 6.3 years (Collins et al. 2009).

Variable fire effects associated with a mixed-severity fire regime make substantial contributions to landscape heterogeneity (Hessburg et al. 2007, Perry et al. 2011). Mixed-severity fire studies suggest that stand-replacing fire effects were a component of Blue Mountain moist forests, but at relatively low proportions across the landscape (~5 to 15 percent) (Agee 1996b, 1998) and consisting mostly of small patches (often less than 10 acres or 4 hectares), but with a few large patches (~150 acres or 60 hectares) also created.

Based on mixed-severity fire regime studies, it appears that moist-forest landscapes with active fire regimes included representation of relatively dense, even-aged tree stands and shrub patches (including mixtures of snowbrush ceanothus and Scouler willow), as well as minor representation of open, park-like, multi-age stands such as the mixture of ponderosa pine and western larch depicted in figure 45.

Fire Regime IIIa

As reported for other portions of the western U.S. (Hessburg et al. 2007, Schoennagel et al. 2004), fire history work in the Blue Mountains suggests that some dry forest may have had a variable fire regime (i.e., it may have exhibited variation in fire return intervals) (Heyerdahl 1997, Heyerdahl et al. 2001, 2002, 2008). When Heyerdahl studied fire regimes for four sites oriented on a north-south transect traversing the Blue Mountains, she found that at the southern site, most of the fire return intervals were less than 25 years, whereas only half of the intervals were less than 25 years for the northern site. The maximum interval decreased from north to south: the Tucannon site (northernmost area, and in the maritime climatic regime) had the largest range of fire interval, and the Dugout site (southernmost area, and in the continental climatic regime) had the smallest range (Heyerdahl 1997).

The Heyerdahl (1997) study suggests that Blue Mountains mixed-conifer forests include a wide spectrum of fire frequency and severity, ranging from an area that is obviously a good example of fire regime I (the Dugout site), to an area (the Tucannon site) best assigned to fire regime IIIa (fig. 44). And, her study corroborates that mixed-severity sites (FR III) can experience almost as much surface fire as low-severity (FR I) sites – see the low-severity percentages for fire regimes I and III in fig. 46 (FR I had 60-90% for low severity, and FR III had 20-50% for low severity, so only 10% separates the low end of FR I (60%) from the high end of FR III (50%)).

Fifteen Blue Mountain sites were sampled to survey fire frequency in stands ranging from Douglas-fir to dry grand fir associations (Maruoka 1994). Current stand structure at 80% of the sites had an overstory dominated by ponderosa pine, with Douglas-fir and grand fir as understory dominants. Pulses of Douglas-fir and grand fir establishment occurred after the last recorded fire at 53% of the sites, while establishment pulses occurred between years of recorded fires at 47% of the sites. Fire scar analyses revealed high variability in fire return intervals. Mean fire intervals at each site ranged from 9.9 years to 49 years. Individual fire return intervals ranged from 2 to 119 years, but may be influenced by sampling limitations.

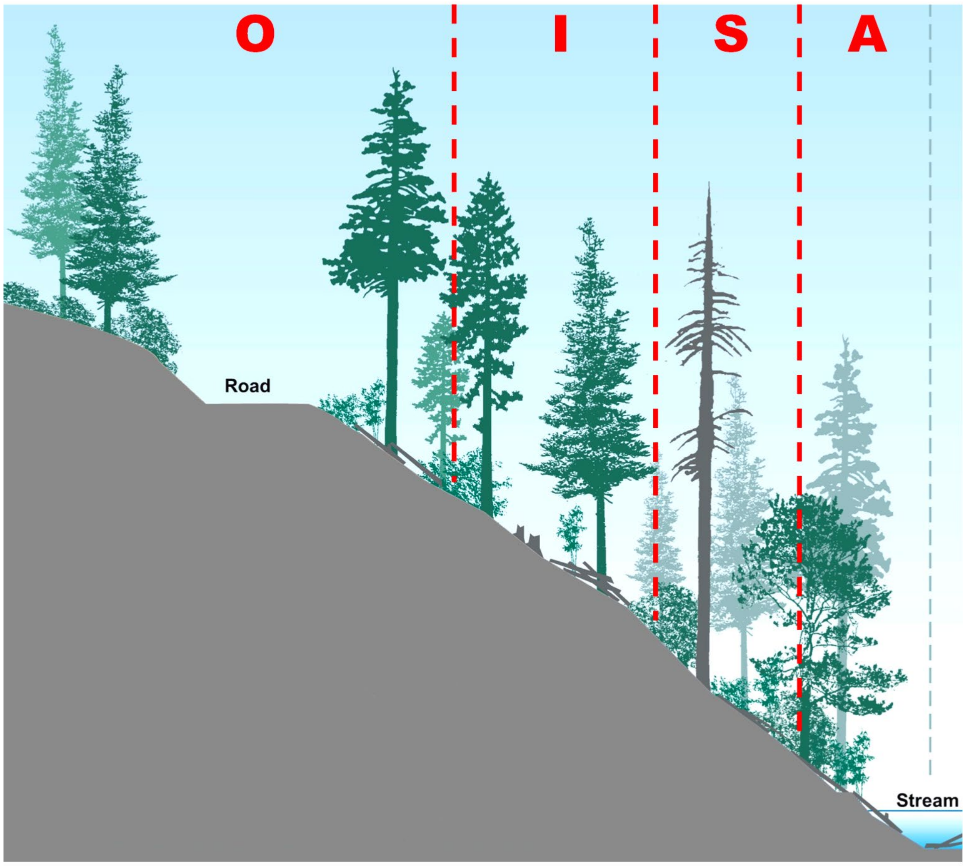


Figure 47 – Illustration of the ASIO model of fire occurrence (concept derived from Angelstam 1998, and Kuuluvainen and Grenfell 2012; diagram adapted from Boise Cascade Corporation 1996). In this model of fire occurrence, the landscape is divided into four zones by using a site moisture and soil fertility gradient. The four zones, moving from right to left, are: A (Absent, where fire theoretically does not occur, or it occurs with an extremely long recurrence interval); S (Seldom, where fire is acknowledged to occur but with a long recurrence interval – perhaps analogous to fire regime IIIc for moist forests of the Blue Mountains); I (Infrequent, where fire occurs with a moderate recurrence interval – possibly analogous to fire regime IIIb for moist forests); and O (Often, where fire occurs with some regularity – probably analogous to fire regime IIIa for moist forests). The ASIO model might function as a template for active management practices designed to emulate the assumed effects of fire frequency and severity on moist-forest species composition, forest structure, and stand density.

The Maruoka (1994) study demonstrates that a predominance of ponderosa pine in the overstory composition does not allow one to automatically assume that fire regime I is present – her sample sites with longer fire return intervals are probably best assigned to fire regime IIIa. And her study, along with the Heyerdahl (1997) results, demonstrate that the model typically assumed for fire regime I sites – frequent, low-severity surface fires occurring on a regular and predictable cycle (Arno 1980, Cooper 1960, Hall 1977) – is not generally applicable to the mixed-severity fire regime, although certain aspects of it may apply to the IIIa portion of fire regime III.

When the drier end of the moist-forest spectrum (e.g., the IIIa variant of fire regime III) was examined in another study, it was found that management-ignited fire tends to be smaller than historical fire, suggesting that the spatial scale of management fires is inappropriate. In other words, prescribed fires need to be larger if one of the objectives is to mimic fires that burned during the historical era on fire regime IIIa sites (Taylor 2000). Many other fuel treatment studies identified fire regime IIIa sites as having a high restoration need (Schoennagel and Nelson 2011): “this experience suggests it is impossible to administer enough small low-intensity burns to reduce accumulations of surface fuel in a dense mixed-conifer forest over a landscape scale of management” (Goforth and Minnich 2008).

Fire Regime IIIb

Fire regime IIIb has longer fire-return intervals than either fire regime I (< 25 years) or fire regime IIIa (≤ 50 years) sites. This means that average fire frequency tends to be intermediate for FR IIIb sites – minor portions may burn at a frequency of less than 50 years, the majority burns at a modal frequency of 51-100 years, and minor portions may burn at a frequency of more than 100 years. Once again, this variability is a hallmark of mixed-severity ecosystems.

The effect of variable fire-return intervals depends on the overstory vegetation and how well adapted it is to survive fire – its ‘fire tolerance’ or ‘fire resistance.’ We can expect forest recovery to be slow in the high-severity portions of a mixed-severity fire, particularly for areas where the pre-fire composition was dominated by species with low fire resistance (table 9). Initially, high-severity areas will support herbaceous vegetation (forbs and grasses) and shrubs. But even in high-severity areas with high amounts of tree mortality, research has shown that regeneration is still effective enough to restore tree dominance by 30 to 40 years after the fire (fig. 48).

Fire Regime IIIc

The long-interval portion of the mixed-severity regime (fire regime IIIc) has characteristics in common with the lower subalpine forest zone (fig. 49). Fires in these environments depend on strong winds, accumulations of dry fuel, low relative humidities, and moderate to high temperatures. These burning conditions may only occur for short periods, not just within one fire season but perhaps only once every few decades, except during a regional-scale drought when severe fire weather may occur for several years in a row. Large fires associated with fire regime IIIc sites tend to have top-down weather controls rather than bottom-up fuel controls (Agee 1998, Bessie and Johnson 1995). [Note that weather is considered to be a top-down control because it exerts a spatially consistent influence across large areas; fuel is a bottom-up control because its influence is spatially variable (Parks et al. 2012).]

Usually when weather and fuel conditions are conducive to fire spread for fire regime IIIc sites, equally dry or drier conditions are present at lower elevations in the fire regime I, II, IIIa, or IIIb areas. High risk at lower elevations results in regional fire danger such that fires are typically suppressed everywhere, even though fires burning under severe weather conditions could be viewed as ecologically desirable, and burning within their historical range of variation, for fire regime IIIc sites (and this would be true even during drought periods). [But as described in the Climate Change section (5.15), progressively earlier snowmelt may extend fire seasons in high-elevation forests by increasing the length of time that fuels are dry enough for sustained fire spread (Karl et al. 2009, Westerling et al. 2006).]

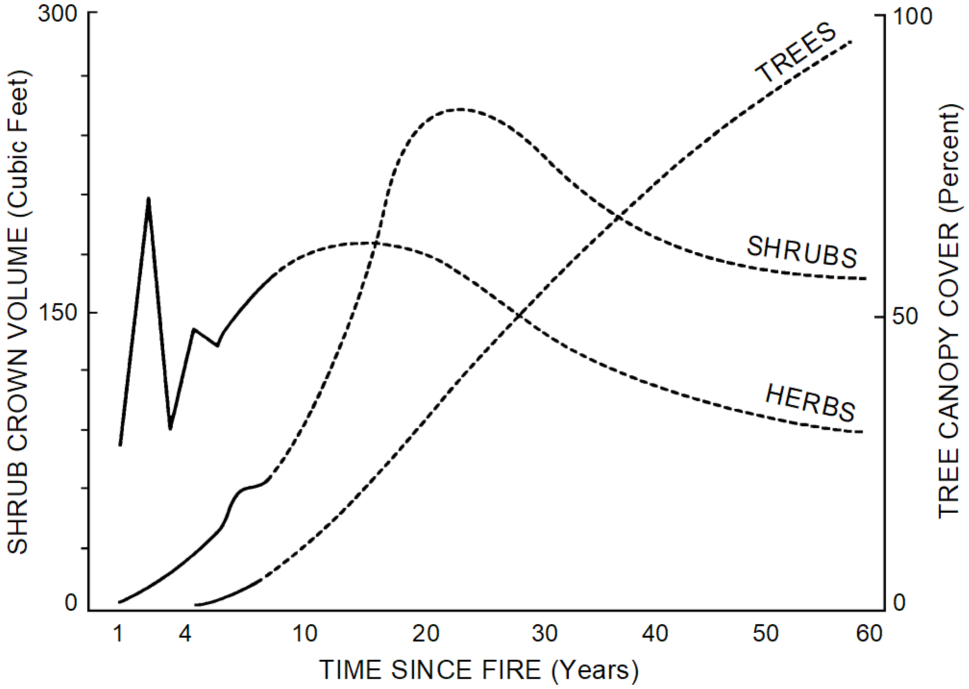


Figure 48 – Plant succession following a stand-replacing fire in south-central Idaho (adapted from Lyon 1971). [Prescribed crown fire occurred on a moist site dominated by Douglas-fir, with associated lodgepole pine, subalpine fir, Engelmann spruce, and quaking aspen. Dominant undergrowth species included Rocky Mountain maple and mountain snowberry.] Herbaceous plants initially dominate a stand-replacing wildfire area. As succession proceeds, shrubs peak by the second decade and trees become dominant between 30 and 40 years later. Note that trees were already established in the post-fire community by the fourth year – a good example of the initial floristics forest development pattern (Oliver and Larson 1996, Powell 2000).

Effects Of Fire Suppression On Mixed-Severity Fire Regimes

In forests with mixed-severity fire regimes, heterogeneity within individual stands, and among stands at a landscape scale, was maintained through variability in fire frequency and severity (fig. 50). Modern fire-fighting technology, in combination with other changes caused by anthropogenic influences, allowed many fires of low or moderate intensity to be suppressed over the previous 75 to 100 years (Arno et al. 2000, Barrett 2004a, Brown et al. 2004, Keane et al. 2002). Some of the fire suppression activity occurred in the moist-forest zone, but the remainder occurred in dry or cold forests, which still influences the mixed-severity regime by suppressing fires that eventually would have burned into moist forests.

Figure 49 – Examples of fire regime IIIb and IIIc sites. The upper photograph shows an example of the grand fir/queencup beadlily plant association; the lower photograph shows an example of the subalpine fir/false bugbane plant association. In a potential vegetation context, the temperature/ moisture regime for both of these plant associations is cool and moist, and they are assigned to the Cool Moist plant association group (PAG). The Cool Moist PAG includes a total of 21 potential vegetation types (consisting of plant associations, plant communities, and plant community types). The Cool Moist PAG is assigned to the Moist Upland Forest potential vegetation group (PVG); PVG is the next higher level (above PAG) in a potential vegetation hierarchy. Since the Cool Moist PAG was assigned to the Moist PVG, this signifies that the potential vegetation types in this PAG are influenced more by moisture than by temperature (Powell et al. 2007). The grand fir/queencup beadlily site (upper photo) is believed to be best assigned to the IIIb fire regime variant; the subalpine fir/ false bugbane site (lower photo) is believed to be best assigned to the IIIc fire regime variant. These assignments of plant association to fire regime are tentative because fire history studies are lacking for these biophysical environments.

One outcome of modern fire suppression activity is that for certain portions of the mixed-severity landscape, the historic fire regime may have been altered due to fire suppression (Amoroso et al. 2011). "A study of fire regimes in Glacier National Park concluded that fire suppression had been very effective in areas that previously had a mixed-severity fire regime, but much less effective in areas of stand-replacement fire regimes (Barrett et al. 1991). A detailed study of the entire inland portion of the northwestern United States also concluded that areas historically under a nonlethal or mixed-severity fire regime have now shifted toward stand replacement regimes (Morgan et al. 1998; Quigley et al. 1996)" (Arno et al. 2000, p. 227).

Figure 50 – Examples of mixed-severity and replacement fire. Proceeding clockwise from upper left: top left image shows mixed-severity fire effects in the Boundary fire, which occurred in 1994 (on the boundary of the Umatilla and Wallowa-Whitman national forests, North Fork John Day and La Grande ranger districts). Top right image shows mixed-severity fire effects at very high elevations of the Summit fire, which occurred in 1996 (North Fork John Day Ranger District, Umatilla National Forest). Lower right image shows mixed-severity fire effects in the Biscuit fire, Rogue River-Siskiyou National Forest, southwestern Oregon. Lower left image shows large areas of replacement fire severity in the Summit fire, depicting a dramatic contrast to the mixed-severity fire effects portrayed in other images. Image credits: T.A. Spies for Biscuit fire (from Halofsky et al. 2011); all others by D.C. Powell.

Observations of recent stand-replacement fires (such as Yellowstone National Park (YNP) in 1988) revealed that they burn in a heterogeneous manner even under extreme conditions (Turner et al. 1994). And note that actual fire environments in YNP during extreme weather conditions (e.g., dry periods resulting in 13% fuel moisture levels for 1000-hour fuels) were observed to be influenced primarily by fuels, not by strong winds (Hargrove et al. 2000).

These fire regime shifts reflect suppression of historically wide-ranging wildfire activity, as evidenced by the Barrett et al. (1997) study identifying 35 large fire episodes between 1540 and 1940, with most of their fire dates recorded across large portions of the interior Columbia River basin. The study also noted that “major fires prior to 1900 burned more area than any fire year since” (but this statement does not reflect fire activity occurring after the mid 1990s) (Barrett et al. 1997).

Perhaps the most important influence of fire suppression activity on the mixed-severity fire regime is its impact on patch size and spatial pattern at a landscape scale. "Stand-replacing fire events historically occurred at the stand scale (10-100 ha), not the landscape scale (>1000 ha), and served the purpose of creating scattered small to medium-sized patches on the larger landscape" (Wright and Agee 2004, p. 455). "On landscapes such as large wilderness areas, the effects of fire exclusion tend to include greater uniformity in stand ages and in stand composition and structure, together with a declining diversity of undergrowth species (Arno et al. 1993; Keane et al. 1996). The intricate, fine-grained landscape mosaic of diverse stand structures and compositions will be replaced by a coarser pattern of even-aged stands" (Arno et al. 2000, p. 227).

For much of the interior Columbia River basin, there has been a dramatic increase in the amount of closed, mid-age forest, including for the warmer and drier portion of the moist-forest biophysical environment (these are fire regime IIIa and IIIb sites). For ecosystems evolving with occasional, fine-scale, replacement fire severity, the recent exclusion of low and mixed severity fires may lead to increased frequency, and perhaps most importantly, more spatially contiguous replacement fire than was experienced historically (Hessburg et al. 2007, Quigley et al. 1996).

One result of fire suppression is that for any individual moist-forest stand selected at random, especially if it features a mid- to late-seral composition and structure, it may not seem to be outside of the range of variation, but a lack of mixed-severity fire for a century or more (in some areas) has certainly resulted in a paucity of young, post-fire vegetation patches at a landscape scale. "But lack of fire is also a major force for change in moist forests where effects are more subtle and have been confused to some extent by aggressive harvesting of seral species plus the impacts of the blister rust fungus" (Harvey et al. 1999, p. 138).

Species composition in moist forests represents a spectrum of fire tolerance and fire resistance (table 9). Changes caused by fire exclusion are not as significant for moist mixed-conifer forest as for dry mixed-conifer forest (see Powell 2014), but fire suppression activities have affected the dry end of the moist-forest zone (these are fire regime IIIa sites) and contributed to increased establishment of shade-tolerant, fire-susceptible species, along with associated increases in landscape homogeneity and fuels connectivity (Arno et al. 2000). "The resulting low fuel accumulations and continuity in mixed-conifer stands repeatedly 'thinned' by low intensity insect outbreaks and surface fires probably prevented or postponed catastrophic stand replacement by high-intensity outbreaks or fires" (Swetnam et al. 1995, p. 24).

Stands are generally comprised of cohorts (Oliver and Larson 1996), and it is typical for moist-forest stands to contain several cohorts, each of which would have originated during a different time period (and therefore has a different age). A common example includes old-forest stands where the upper canopy stratum contains older trees with diameters greater than 21 inches dbh (the upper stratum is often a separate cohort with its own age class), along with younger cohorts occupying lower canopy strata. A multi-cohort stand structure can arise during normal successional processes, but it can also reflect situations where low- or moderate-intensity disturbance processes (such as low- or moderate-severity fires occurring within a mixed-severity fire regime) have killed some of the overstory trees, but not the whole cohort. After growing space is liberated by these partial mortality events, a new tree cohort can get established, contributing to a multi-cohort stand structure.

Studies of lodgepole pine forests growing on cold upland sites (fire regime IV) in the Yellowstone area (Romme 1982, Barrett 1994) "suggest that in systems in which infrequent crown fires dominated the presettlement disturbance regime, recent fire suppression has not fundamentally altered landscape dynamics" (White et al. 1999). However, much of the lodgepole pine forest type in the Blue Mountains occurs on moist upland sites (not cold upland sites), and it is more properly assigned to a mixed-severity, moderate-frequency fire regime (fire regime III). Studies examining species composition trends for moist-forest ecosystems with a mixed-severity fire regime have often noted replacement of early-seral species (e.g., lodgepole pine, quaking aspen) by late-seral species in the absence of fire (Wadleigh and Jenkins 1996).

As is true for disturbance processes other than wildfire, and for biophysical environments other than moist forest, the spatial context of a process or environment can influence the ultimate outcome. Often, moist-forest (FR III) patches occur within a larger matrix dominated by either dryer (FR I) or colder (FR IV) biophysical environments. These moist-forest inclusions tend to have fire frequencies and severities much like those found in the surrounding matrix – in other words, characteristics of the matrix can overwhelm characteristics of the inclusions (Arno 1980). Small inclusions of FR III located within a much larger expanse of FR I, for example, will tend to burn as though they too are FR I. Similarly, inclusions of FR IV within a larger matrix of FR III will tend to burn as though they are FR III.

Are Fire Regimes Similar For Upland And Riparian Forests?

"Riparian forests exist within a matrix of sideslope forests and the role riparian forests play in the propagation or suppression of fire disturbance on the landscape is poorly understood. Riparian areas with heavy fuel loading and fuel continuity may serve as conduits (disturbance corridors) for the rapid spread of fire (Agee 1993). Conversely, the more mesic riparian areas may serve as fire breaks in surface fires. The disturbance relationship between riparian and sideslope forests can be anticipated to change in different landforms and plant associations. We know there are many feedback loops between riparian and sideslope forest systems but we do not understand how closely the fire disturbance regimes of riparian and sideslope forests are intertwined" (Everett et al. 2003).

"Pre-settlement forest structure and composition along first and second order streams in the Mixed Conifer zone resembled upland forests in the region. Given the historic continuity of fire disturbance between riparian forests and adjacent uplands (Everett et al. 2003, Olson and Agee 2005), it may be beneficial to permit partial harvest treatments and prescribed fire in some riparian areas to allow restoration of desirable characteristics of pre-settlement forest structure and composition. Treatments may include the creation of large canopy gaps, untreated 'islands', clumps, and irregularly spaced trees. Because most riparian forests have not burned for 70-100 years, many trees that would have been killed by low- or moderate-severity fires are now too large to be killed by low-severity prescribed fires" (Messier et al. 2012).

Everett et al. (2003) came to a similar conclusion when they noted "our historical cohort information indicates reduced fire effects since the early 1900s have altered the number and age structure of cohorts in both sideslope and riparian forests, making them more similar. Increased homogeneity (reduced patchiness) has negative attributes of increased continuity in fuels and insect hosts that create significant problems in the management of sustainable forests" (Everett et al. 2003, p. 45).

An observation that composition and structure along low-order streams may not vary much from adjacent upland conditions (Dwire and Kauffman 2003, Messier et al. 2012) has also been noted for the Blue Mountains. "Olson (2000) found fire occurrence in riparian zones to be only slightly less frequent than on adjacent uplands in similar forest types in the Blue Mountains in Oregon" (Wright and Agee 2004: 454).

As Olson noted in her thesis: "Keeping fire out of the ecosystem will not only continue to alter the structure and vegetational composition of these riparian forests, but will also allow the buildup of fuels that could result in unprecedented fire intensities, and subsequently higher fire severities, than were present in the system historically. If the goal of forest management is to restore historical disturbance regimes to these forests, results from this study indicate riparian forests should be managed according to the historical fire regime of the forest type rather than distance from a stream" (Olson 2000, p. 78) (in this context, "distance from a stream" refers to a process of using designated buffer widths (in feet), varying by stream class, to establish riparian habitat conservation areas).

Results from the Everett et al. (2003) research "indicate the percentage of shared fire disturbance between the riparian forest and the sideslopes is, in part, the combined result of the riparian forest plant association group, the sideslope plant association groups, and the topography of the site" (Everett et al. 2003, p. 41). And a study examining Sierra mixed-conifer riparian vegetation (Russell and McBride 2001) "showed that proximity to water exerted a greater influence on vegetation composition than did historical fire occurrence" (Kobziar and McBride 2006).

Vegetation Response To Mixed-Severity Fire

For at least the last several thousand years, fire has been an important initiator of forest succession in the interior Pacific Northwest. Fire severity in these forests varies from light surface fires to intense, stand-replacing crown fires. A stand-replacing crown fire, which typically also includes a coupled surface fire, constitutes one of the most severe disturbance events a moist-forest ecosystem experiences. Due to its severity, crown fire has occasionally been termed holocaustic.

A holocaustic fire is one that: 1) kills the coniferous tree overstory, 2) reduces the tree-shrub understory and herb layers to ground level, and 3) consumes all of the dead organic material on the forest floor down to the mineral soil surface. Although holocaustic fire incinerates the above-ground portion of the forest community, the below-ground portion can remain intact and essentially undisturbed. Plants comprising the initial community following holocaustic fire have been classified as survivors, residual colonizers, and offsite colonizers (table 12; Stickney 1990).

A mixed-severity fire often affects a large area supporting a wide diversity of plant species. Plants have varying degrees of fire resistance. A plant's response to fire depends on factors such as the moisture content of soil and duff at the time of burning, the physiological stage of the plant (immature, mature, etc.), and the fire's severity, particularly regarding the amount of heat that permeates the litter, duff, and upper soil layers (Crane and Fischer 1986, Stickney 1990). An important factor affecting a plant's fire resistance is whether it regenerates vegetatively (survivor plants) or from off-site or buried seed (colonizer plants) (table 12) (Stickney 1990).

Table 12 provides information about the fire response mode and seedling competition risk associated with 29 shrubs and herbs commonly associated with the mixed-severity fire regime in the Blue Mountains. Note that seven of the plants in table 12 pose a high risk of competing aggressively with planted conifer seedlings, and are collectively referred to as 'competing vegetation.'

Historical Proxies For Fire Regime Characteristics

Using historical proxies to estimate mean fire intervals or other attributes of the historical disturbance regime can be problematic for regimes featuring mixed-severity fire as a dominant process. It can be difficult to parse out the relative proportions of moderate- and high-severity burning in mixed-severity regimes because one of the legacies of large but infrequent high-severity burns is that they remove much of the evidence of previous low- or moderate-severity fire (Barrett et al. 1997). And even when excellent fire history information exists for specific sites within a mixed-severity regime, information may be lacking or missing for a larger landscape scale.

Disturbance frequencies based on charcoal deposits in lake sediments are available for several Blue Mountain locations (Hansen 1943, Mehringer 1997), but they do not account for important disturbance processes such as mountain pine beetle outbreaks (unless an outbreak was followed by fire). Consequently, charcoal analyses tend to underestimate overall disturbance frequency by not being able to detect insect outbreaks (Wong et al. 2003), and forest insects have affected substantially more area than wildfire (see fig. 18), at least during the modern era. But even with these caveats about proxies, there is a pressing need to consider long-term proxy records as a way to provide a long, temporal perspective (Willis and Birks 2006).

Table 12: Fire response mode and seedling competition risk ratings for common plants of the mixed-severity fire regime.

PLANT SPECIES	RESPONSE MODE	COMPETITION RISK
Bearberry *(Arctostaphylos uva-ursi)*	Survivor	Moderate
Birchleaf spiraea *(Spiraea betulifolia)*	Survivor	Low
Bracken fern *(Pteridium aquilinum)*	Survivor	High
Bull thistle *(Cirsium vulgare)*	Offsite Colonizer	High
Canada milkvetch *(Astragalus canadensis)*	Residual Colonizer	Low
Canada thistle *(Cirsium arvense)*	Survivor	High
Common chokecherry *(Prunus virginiana)*	Survivor	Low
Common snowberry *(Symphoricarpos albus)*	Survivor	Moderate
Dwarf rose *(Rosa gymnocarpa)*	Survivor	Low
Elk sedge *(Carex geyeri)*	Survivor	High
Fireweed *(Epilobium angustifolium)*	Offsite Colonizer	Moderate
Heartleaf arnica *(Arnica cordifolia)*	Survivor	Low
Low Oregongrape *(Mahonia repens)*	Survivor	Moderate
Miners lettuce *(Claytonia perfoliata)*	Residual Colonizer	Low
Northwestern sedge *(Carex concinnoides)*	Survivor	Moderate
Oregon boxwood *(Paxistima myrsinites)*	Survivor	Low
Pearly everlasting *(Anaphalis margaritacea)*	Offsite Colonizer	Low
Pinegrass *(Calamagrostis rubescens)*	Survivor	High
Red fescue *(Festuca rubra)*	Survivor	High
Ross sedge *(Carex rossii)*	Survivor	High
Scouler willow *(Salix scouleriana)*	Residual Colonizer	Moderate
Showy aster *(Aster conspicuus)*	Survivor	Low
Snowbrush ceanothus *(Ceanothus velutinus)*	Residual Colonizer	High
Sticky currant *(Ribes viscosissimum)*	Residual Colonizer	Moderate
Tailcup lupine *(Lupinus caudatus)*	Residual Colonizer	Low
Watson willowherb *(Epilobium ciliatum)*	Residual Colonizer	Low
Western hawkweed *(Hieracium albertinum)*	Offsite Colonizer	Low
Western yarrow *(Achillea millefolium)*	Offsite Colonizer	Low
Woods strawberry *(Fragaria vesca)*	Survivor	Low

Sources/Notes: Plant Species include those observed to be abundant in post-fire plant communities after moderate- or high-severity burns in the central Blue Mountains; Response Mode is based on Strickler and Edgerton (1976) and other sources; Competition Risk is based on local experience. For the Response Mode item, survivors are sprouters and other plants capable of regrowth; colonizers are post-fire plants established from seed, with residual colonizers originating from onsite (burned) sources and offsite colonizers originating from non-burned sources (Stickney 1990). Species with high competition risk are capable of directly killing conifer seedlings; species with moderate risk may cause limited seedling mortality, but more commonly cause growth losses; plants with low risk cause limited growth losses and no seedling mortality. Plant nomenclature follows Botanical Resources Group (2002) and other sources.

Atmosphere-Ocean Influences On Temporal Fire Patterns

A recent increase in the literature dealing with fire climatology relates to improved scientific understanding of broad-scale interactions between the ocean, atmosphere, and regional or continental precipitation and temperature regimes (Heyerdahl et al. 2002, 2008). And superimposed on improved atmosphere-ocean understanding is mounting evidence of anthropogenic warming trends caused by greenhouse gas emissions, and leading to recent increases in area burned and fire season length (Westerling et al. 2003, 2006).

Temporal variations in historical climate records show an oscillating pattern between warm-dry and cool-moist periods over much of the Pacific Northwest (Dello and Mote 2010). Many of these variations are related to broad-scale patterns of climatic variability referred to as the El Niño/Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO). During El Niño years (the warm phase of ENSO), the Pacific Northwest generally experiences drier and warmer than average winters and drought conditions during the growing season; La Niña years (the cool phase) typically bring cooler and wetter conditions than normal.

The Pacific Decadal Oscillation is a longer-term fluctuation in weather patterns, tending to occur on a cycle of 15 to 35 years. Positive (warm) PDO events tend to divert the jet stream northward and southward of the Pacific Northwest, causing storms to bypass this area. During negative (cool) PDO events, the jet stream is aligned with the Pacific Northwest, directing storms to the area. The PDO was in cool phases from 1900-1925 and 1945-1975, and in warm phases from 1925-1945 and 1975-2007 (Pohl et al. 2002).

ENSO tends to occur on much shorter temporal scales than PDO, but when PDO and ENSO are in warm phases simultaneously, they can have a reinforcing effect and winters in the Pacific Northwest can then be exceptionally warm and dry. When PDO and ENSO are in cool phases together, Pacific Northwest winters are especially cool and wet (Dello and Mote 2010). It is noteworthy that both ENSO and PDO appear to be increasing in intensity, and reversing phases more frequently, ostensibly in response to a 20th-century global-warming trend (Pohl et al. 2002).

Very long temporal climate patterns (much longer than ENSO or PDO) have also affected fire patterns for mixed-severity fire regimes. When a large watershed in western Alberta was studied, it was found that climatic conditions were warmer and dryer before 1730, a period when the fire cycle averaged 50 years. Climatic conditions were cooler and moister after 1730, and the fire cycle then lengthened to an average of 90 years (Johnson and Larsen 1991).

In the mid 1980s, the western United States experienced a shift to more frequent and longer-burning fires, a shift which apparently corresponded to earlier snowmelt dates (Westerling et al. 2006). With predicted overall drying, continued earlier snowmelt, and warmer temperatures resulting from climate change, western forests may experience increases in fire size and frequency, and in the amount of land area experiencing high to extreme fire danger (Hurteau and North 2008). [And during the past decade, the western U.S. has seen at least 60 fires of 100,000 acres or more.]

The factors influencing wildfire occurrence and severity can be thought of as being controlled by bottom-up and top-down factors (perhaps in a similar conceptual framework to the stand-maintaining disturbance processes described in fig. 13) (Parks et al. 2012, Pierce et al. 2004). Fuel type, fuel abundance, and fuel distribution are all bottom-up controls, and they are amenable to modification by active management practices. Climate, however, particularly the broad-scale climatic trends represented by the ENSO and PDO patterns described above, are top-down controls over which we have no control (except for anthropogenic production of greenhouse gases and its effect on global climate change).

Even in forests where restoration or climate mitigation activities have been accomplished, fires will still burn under extreme weather conditions, given an ignition source (Bessie and Johnson 1995, Parisien et al. 2011). But if restoration or mitigation activities have properly prepared an ecosystem for severe-weather wildfire, then we can reasonably assume that critical ecosystem components (such as compositional and structural attributes) will persist as inter-generational ecosystem memory.

Native Americans And Fire

Often, it is assumed that when Europeans arrived in the New World, American Indians sparsely occupied the land, the impacts of native peoples were minor, and landscapes were pristine. Although consensus about the pervasiveness of human intervention is lacking, evidence suggests that effects of human management were extensive during the presettlement era (Anderson 2006, Bonnicksen et al. 1999, Botkin 1995, Cronon 1996, Kay and Simmons 2002, Mann 2006, Stewart 2009).

It is entirely possible that Blue Mountain forests were more primeval at the time of Euro-American settlement than before that era. When Columbus landed in 1492, it is estimated that North America (exclusive of Mexico and central America) supported at least 3.8 million Native Americans. By 1800, their numbers had been reduced to a million or less by measles, smallpox, cholera, influenza, and other European diseases (Denevan 1992, Mann 2006). [But note that historical records for the Pacific Northwest, as compiled by Robbins and Wolf (1994), could be interpreted as contradicting this conclusion, perhaps suggesting that the most significant population declines occurred in the American southwest and central America.]

Recent investigations indicate that American Indians were far from the passive hunters and gatherers often depicted in western movies and novels. Their actions had a profound influence on the structure and composition of western ecosystems, a not unexpected result when considering that they used hundreds of plants and animals for food, fiber, shelter, forage, and medicine (Daubenmire 1975, Moerman

2010). Fire was often their main tool for creating and maintaining the habitats required by these plants and animals (Anderson 2006, Boyd 1999, Denevan 1992, DeWalt 1994, Kay 1994, Robbins 1997, Shinn 1980, Stewart 2009).

Even though their populations were already declining due to introduced diseases (Cook 1955), Native Americans of the interior Pacific Northwest may have expanded their use of fire in the early 1700s, perhaps to support evolving lifeways associated with their recent acquisistion of horses (Barrett et al. 2005, Habeck 1987, Haines 1938, Humphrey 1943, Mosgrove 1980, Stewart 1951).

The influence of Native American burning on western forests was frequently noted in early reports and exploration journals (Boyd 1999, Vale 2002). For example, "seven of the ten fires witnessed by explorers Lewis and Clark between 1805 and 1806 in the northern Rockies were attributed to Indian ignitions" (Barrett and Arno 2002, p. 51). The following excerpts provide examples of Native American burning:

> "The assumption is that our forests today, having been untouched by man and exposed to the same factors of their surroundings since times immemorial, must represent more or less exactly the same character they had 100 or 1,000 years ago. But we have practically no genuinely virgin forests; in the great majority of commercial accessible stands, man has for centuries practiced some kind of primitive forestry by setting fires. This 'Piute forestry' has changed the aspect of many stands so completely that the term 'virgin forests' is far from being correctly applied. At best, one can speak of scattered virgin stands here and there" (Meinecke 1916).
>
> "On the way, they met an old squaw, with a large firebrand in her hand, with which she had just set the grasses and bushes on fire; when surprised, she stood motionless, and appeared to be heedless to any thing that was passing around her…there were no other Indians in sight" (Wilkes 1844, p. 251).
>
> "Destroyer by instinct, the savage will never be a producer[3]. How many times has the eye not been saddened, along the riverbanks or in the interior, by the traces of immense forest fires stretching for several miles; he (the savage) has destroyed them (the forests) exclusively to drive out game or his horses that he believed took refuge there, or in order to open up access for himself, when he will go there to harvest his grain (seeds). At summer's end, he sets fire with even greater ease to the entire prairies; but one knows that here he has a useful goal: several weeks afterward, a new, green tasty grass has regrown, richer in nutritional value and preferred by the livestock. It is not the same for the forests" (Saint-Amant 1854, translated from the French, commenting about the Umatilla River region in northeastern Oregon; quote taken from Stewart 2009, p. 228-229).

The quote from the Wilkes Expedition is especially interesting because it demonstrates that much of the Native American burning expertise resided with women, perhaps because they were the primary stewards of the 'women's foods' portion of first foods by promoting and sustaining root plants (bitterroot, cous, camas, etc.) and

---

[3] "Only to the white man was nature a 'wilderness' and only to him was the land 'infested' with 'wild' animals and 'savage' people." Luther Standing Bear (1868-1939) (quote from Stewart 2009).

fruits or berries (huckleberries) (Hunn 1981), some of which required periodic fire to maintain their vigor and productivity (Mack 2001, Turner 1999, Turner et al. 2003).

A variety of native plant species, primarily berries and roots, were managed by using aboriginal burning practices. Burning maintained and promoted shrubby fruiting species (serviceberry, big huckleberry, chokecherry, thimbleberry, currants and gooseberries, elderberry, buffaloberry, and others; fig. 51), herbs with aboveground edibility such as strawberry, and many species of onion, and herbs with edible underground parts such as camas, bitterroot, cous and other biscuitroots, yampa, spring beauty, several lily species, and onions (Stewart 2009, Turner 1999).

"Possibly increased growth would result from an increased supply of available nutrients near the surface of the ground following burning. Fire increases the pH of the soil through release of alkaline ions such as phosphorus, potassium, calcium, and magnesium, and studies show that these nutrients are more readily accessible to plants after a fire. The amount of nutrients released varies with the type of soil and intensity of the burn" (Turner 1999). These observations about aboriginal burning practices illustrate that since the mid-1990s, ecosystem thinking has broadened to incorporate humans not just as a source of outside disturbance, but as integrated within other biological and physical processes and structures (Cronon 1996).

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

Note, however, that there is not universal consensus about the pervasive, widespread use of fire by Native Americans. "Lightning fires, including onsite ignitions and fires spreading from other areas, were well capable of maintaining most fire regimes in the West" (Barrett et al. 2005, p. 32). Much of the uncertainty about anthropogenic burning involves the mixed-severity fire regime because "most early-day accounts suggest that Indian fire use occurred largely in grasslands and adjacent dry forests" (e.g., the low-severity fire regime) (Barrett et al. 2005, p. 33).

But regardless of uncertainty about the scope and influence of Native American burning, it is clear that increased consideration of native knowledge (sometimes referred to as traditional ecological knowledge) would provide helpful insights supporting moist-forest management, particularly in a collaborative context where divergent viewpoints are pursued and valued (Anderson 2006, Berkes 2008, Berkes et al. 2000, Charnley et al. 2007, Kimmerer 2000, Pierotti and Wildcat 2000).

Figure 51 – Grand fir stand with a dense shrub layer dominated by big huckleberry (*Vaccinium membranaceum*), an important Native American first foods species (Moerman 2010). [Photo by D.C. Powell in South George planning area, Pomeroy Ranger District.] Note that thinning the tree overstory to a moderate canopy cover would be expected to increase the productivity (berry production) and vigor of big huckleberry (Hedrick et al. 1968).

Interactions Between Fire And Other Disturbance Processes

Fire regimes interact with other disturbance processes in synergistic and interesting ways (Romme et al. 2011). Perhaps some of the best examples of disturbance-process interactions involve lodgepole pine, a common and important forest cover type associated with moist forests and the mixed-severity fire regime (see table 17 in section 6), and mountain pine beetle, a cambium-feeding invertebrate causing extensive tree mortality across millions of acres of lodgepole pine forest in western North America (Bentz 2009, Woods et al. 2010).

One or more decades after a mountain pine beetle outbreak, when beetle-killed trees topple over and a new cohort of tree regeneration has gotten established (fig. 52), the risk of crown fire (in the regenerated seedling/sapling cohort) may be as great as it was before the bark-beetle outbreak (Gara et al. 1985, Roe et al. 1971). But contrary to suppositions from some of the historical research, crown-fire potential can be low during or soon after a bark-beetle outbreak when dead trees are in the ephemeral ‘red crown’ or ‘gray crown’ stages (Simard et al. 2011) (but even this conclusion has been disputed – see Jolly et al. 2012). This research demonstrates that time-since-beetle-outbreak can be a critical factor for understanding the relationship between mountain pine beetle outbreak, stand structure, fuel dynamics, and fire risk (figs. 6 and 52) (Hicke et al. 2012; Jenkins et al. 2008, 2012).

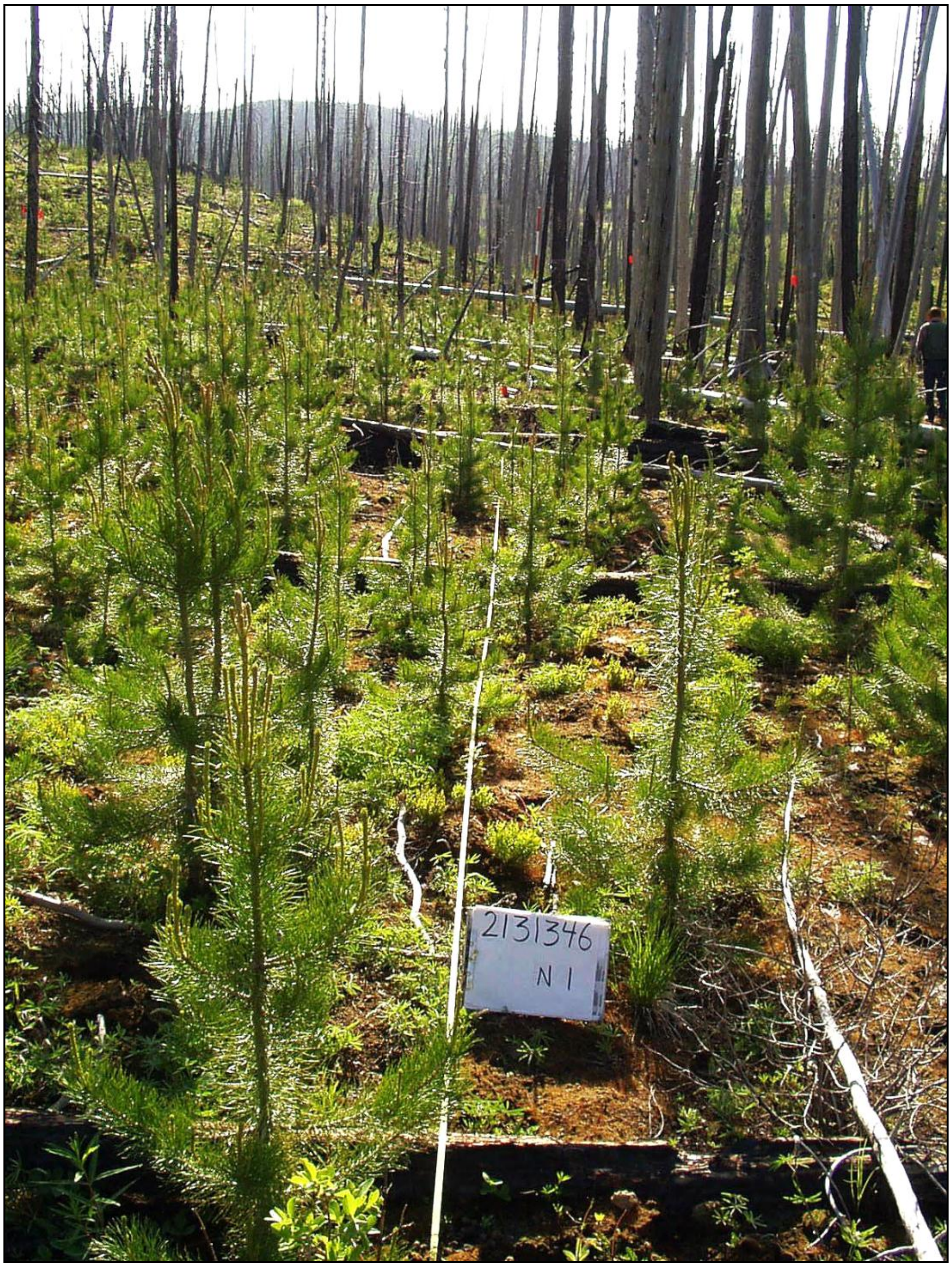


Figure 52 – Dense lodgepole pine regeneration following stand-replacement fire on a moist-forest site; Tower wildfire area (this 50,800-acre fire burned in August-September 1996). Seedling density in this area is about 20,000 per acre. The northern third of the Tower fire is a good example of how one disturbance process (a 1970s-era mountain pine beetle outbreak in lodgepole pine – see section 5.8) functioned as a predisposing factor for a subsequent disturbance process (high-severity wildfire, both with and without active crown fire). Fuel accumulation resulting from a bark beetle outbreak can affect subsequent fire behavior (Roe et al. 1971), as it did for the Tower Fire. Although interactions between stand development, tree growth, stand density, phloem thickness, and bark-beetle susceptibility can be complex (figs. 10 and 53), it is possible to thin post-fire lodgepole pine stands and thereby maintain low levels of crown-fire susceptibility (Powell 2010).

Studies described in this section relate to how mountain pine beetle (MPB) activity influences subsequent fire occurrence and severity (or for some studies, whether MPB even has an influence on fire). But we should also remember that interactions between these two disturbance agents are reciprocal – they also function in the other direction (e.g., fire effects can influence subsequent MPB activity). “Recent research has shown that structurally and compositionally diverse stands, with variable but generally low densities of mature trees, are less susceptible to mountain pine beetle outbreaks. Such stand attributes are consistent with a mixed-severity fire regime” (Amoroso et al. 2011) (this quote is especially germane to the historical fire effects associated with fire regimes IIIa and IIIb).

Dwarf mistletoes, parasitic seed plants, are often considered to be the most serious disease agent in lodgepole pine forests (Hawksworth and Dooling 1984). Just like mountain pine beetle, lodgepole pine dwarf mistletoe can also cause profound effects on forest structure and function, although they occur more slowly than with bark beetles. “Data show that chronic increases of dwarf mistletoe are partly due to the exclusion of fire (Zimmerman and Laven 1984) because fire is the natural control of dwarf mistletoe and has played a major role in the distribution and abundance of current populations and infection intensities (Alexander and Hawksworth 1975). As the frequency and extent of fire have decreased in lodgepole pine stands over the last 200 years, dwarf mistletoe infection intensity and distribution are clearly increasing (Zimmerman and Laven 1984)” (Ferry et al. 1995).

Depending on perspective, fire could also be viewed as a “predator seeking out and consuming plants as its energy source, while they either avoid it in the sanctuary of ‘escape terrain’ or respond on their home ground by erecting defenses and safely sequestering their viable resources.” But “certain plants can be considered active searchers after fire, tracking down and scavenging burned-over ground whenever it appears, moving quickly and efficiently to profit from newly available resources” (Rowe 1983).

Perhaps our best example of an ‘active searcher’ tree species is lodgepole pine. Lodgepole pine, a thin-barked species, has low resistance to fire, regardless of whether a fire has low or high fireline intensity (flame length). But since lodgepole pine regenerates prolifically after fire, often in response to high levels of seed-banking related to cone serotiny (as described in fig. 54), its long-term resilience to fire is actually quite high (fig. 52).

Moderate-severity fires in mature lodgepole pine ecosystems tend to produce a much higher post-fire seedling density than high-severity burns (fig. 52), and high-density lodgepole pine communities develop differently than low-density ones (Anderson and Romme 1991). But regardless of fireline intensity, a common conclusion of studies examining relationships between lodgepole pine and wildfire is that exposure of mineral soil is an important outcome affecting regeneration success and seedling density (fig. 53; Axelson et al. 2009).

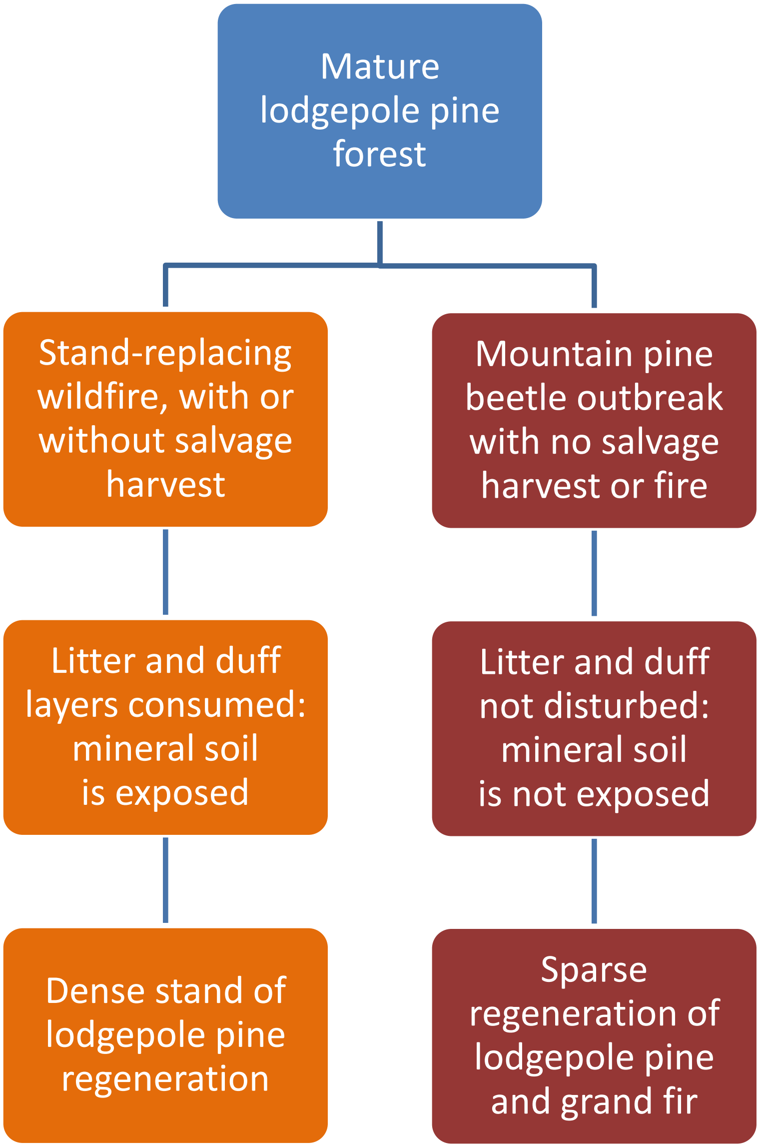


Figure 53 – Alternative response trajectories for a mature lodgepole pine forest experiencing either a stand-replacing wildfire (left) or a mountain pine beetle outbreak (right) (concepts derived from Axelson et al. 2009). Stand-replacing wildfire and mountain pine beetle are the two main disturbance agents that maintain forests of lodgepole pine in various structural stages (see fig. 10) across moist-forest landscapes. In the left trajectory, a mature forest experiences a high-severity wildfire that consumes the soil's O horizon (the litter and duff layers), exposing mineral soil and quickly resulting in a new cohort of dense lodgepole pine regeneration. The right trajectory shows development following a mountain pine beetle outbreak where the dead trees are not removed for economic purposes (salvage harvest). If an outbreak is not followed by wildfire (and not all of them are), then the right-side trajectory shows that a lack of mineral-soil exposure results in a sparse post-disturbance species mixture containing both lodgepole pine and late-seral species such as grand fir or subalpine fir. Partial, stand-maintaining disturbances (see fig. 13) such as low-severity fire, fungal pathogens, and dwarf mistletoes, open the canopy and promote establishment of shade-tolerant spruce and fir.

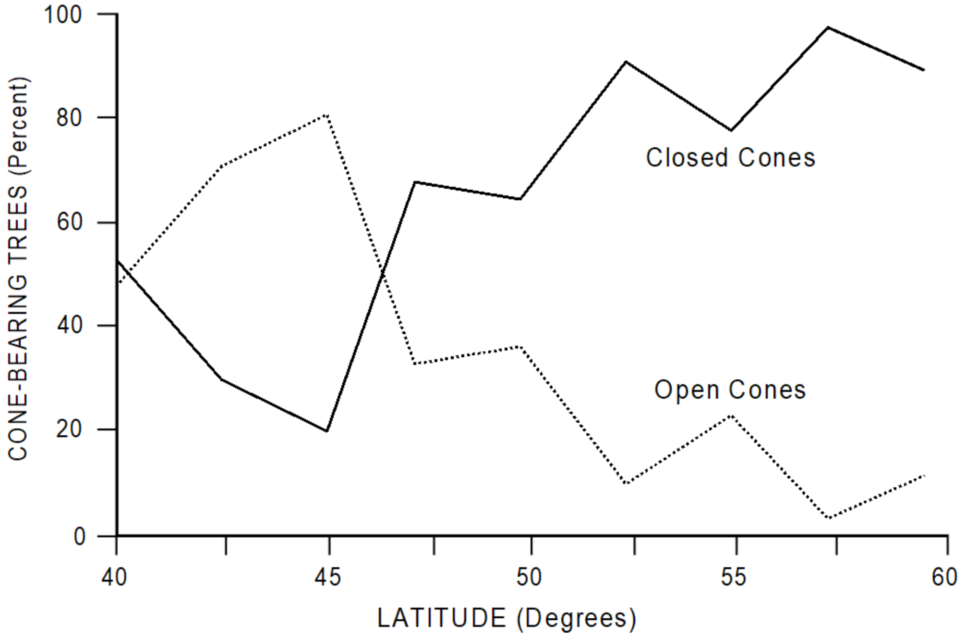


Figure 54 – Variations in cone serotiny for lodgepole pine (from Koch 1996). Lodgepole pine serotiny (the 'closed cones' trait above) varies with latitude. Note that the lowest percentage of closed cones in western North America (20%) occurs at latitude of 45 °North, which coincides with the central Blue Mountains. Since serotiny is presumed to represent an adaptation to fire (Lotan 1976), the trends displayed in this figure suggest that crown fire exerted less genetic selection pressure in the Blue Mountains than in other parts of lodgepole's range. Serotiny also varies with stand age – the closed-cone trait is more common in old trees or stands than in young trees or stands (Lotan 1976, Mason 1915). Variations in the percentage of serotinous trees can have an important influence on landscape heterogeneity (Turner et al. 1997). [Note that serotiny is typically viewed as an ecological adaptation designed to disperse copious amounts of seed from standing trees. But research shows that it also contributes to seed banking in the forest floor – the highly resinous cone-scale bonds are resistant to decomposition, so serotinous cones falling to the forest floor can persist there for long periods (Teste et al. 2011b).]

Lodgepole pine populations in the Blue Mountains coincide with the portion of lodgepole pine's west-wide range having relatively low amounts of cone serotiny (fig. 54). Even though lodgepole pine has evolved with wildfire as the primary ecological cue for breaking serotinous cone-scale bonds and then dispersing copious amounts of seed (Lotan 1976), mountain pine beetle has also been found to provide a weak cue for seed dispersal, although at much lower levels than is associated with fire (Teste et al. 2011). Seedling establishment after bark-beetle outbreaks is typically curtailed because a favorable regeneration substrate (e.g., exposed mineral soil) is not created by the disturbance process (Axelson et al. 2009). [Note that lodgepole pine's relatively low amount of cone serotiny is not the only factor distinguishing Blue Mountain populations from other lodgepole areas in the western U.S. – see Box 5.]

Box 5. Is lodgepole pine a cold or moist type in the Blues?

After a major wildfire event affected the greater Yellowstone ecosystem (which is centered on Yellowstone National Park) in 1988, much scientific literature has been published about fire ecology and fire effects for lodgepole pine. This work, involving investigators such as William Romme, Monica Turner, Tania Schoennagel, Daniel Tinker, and others, is useful for that particular geographical area, and for those ecological settings. However, the Yellowstone studies almost always pertain to cold, continental climates, and **it is not appropriate to extrapolate Yellowstone lodgepole pine research to lodgepole pine forest growing on moist sites in the Blue Mountains**. This caveat is not surprising because it reinforces a central tenet of forest ecology: lodgepole pine is not created ‘equal’ everywhere, just as ponderosa pine and other tree species express considerable variation across their ranges.

Unlike Rocky Mountains populations to the east, lodgepole pine in the Blue Mountains frequently occurs on moist-forest sites (rather than cold forest exclusively or predominantly), and this ecological distribution affects serotiny, wildfire, and bark-beetle relationships. This also means that some of the Blue Mountains lodgepole pine type is best assigned to a mixed-severity fire regime (FR III; lodgepole pine on moist-forest sites), rather than the replacement fire regime (FR IV; lodgepole pine on cold-forest sites).

In a potential vegetation context, lodgepole pine is seldom the climax tree species, so lodgepole pine stands are almost always classified using lodgepole pine plant community types (PCTs; these are seral or successional stages of a plant association typically occurring in the grand fir or subalpine fir series). A total of 507 potential vegetation types have been described for the Blue Mountains; this total includes 16 plant community types and 1 plant association dominated by lodgepole pine and assigned to an upland forest potential vegetation group (PVG). Of the 17 upland-forest lodgepole pine types, 9 were assigned to the Moist Upland Forest PVG, and 8 were assigned to the Cold Upland Forest PVG (Powell et al. 2007).

The Blue Mountains include three regional climatic zones: the maritime zone (northern Blues), the mixed zone (central Blues), and the continental zone (southern Blues) (Caraher et al. 1992). As one might expect, most of the lodgepole pine occurring in the continental zone, along with some of the lodgepole type in the mixed zone, is properly assigned to the Cold Upland Forest PVG (and to fire regime IV). The balance of the mixed-zone lodgepole pine type, along with most of the lodgepole type occurring in the maritime zone, is best assigned to the Moist Upland Forest PVG (and to fire regime III).

Reburn As A Postfire Phenomenon

Following severe wildfires, standing dead trees are abundant on the landscape and they will eventually fall. In some circumstances, the accumulation of this down wood or coarse woody debris (CWD) is sufficient to support another relatively severe wildfire on the same area (Odion et al. 2004, Passovoy and Fulé 2006). A second fire occurring relatively soon after the first one is called a reburn (or in older literature, a ‘double burn’). Reburns, a common postfire phenomenon (table 13), have been described for many areas of the western United States (Hofmann 1917). The ecological role of reburning, and whether or how to implement active management practices following a wildfire to influence the potential effects of a reburn on the same area in the future (practices such as salvage timber harvest or tree planting), have been particularly controversial (Donato et al. 2006, Hudec and Peterson 2012, Thompson et al. 2007, and many other citations in the References section).

Table 13: Reburn analysis for vicinity of 1996 Tower wildfire area, Umatilla National Forest.

INFORMATION FOR ORIGINAL FIRE				
Fire Name	**Total Acres**	**Original Year**	**Year of Reburn**	**Reburn Fire Name**
Big Creek	74*	1988	1996	Tower
Big Creek	91	1993	1996	Tower
Big Creek	153	1994	1996	Tower
Cable Creek	553*	1986	1996	Tower
Long Meadows	167*	1986	1996	Tower
Lost Lake	2,804	1986	1996	Summit
Placer	379	1993	1996	Tower
Ryder Creek	13,610	1987	1996	Tower
			1996	Bull
			2001	Big Creek
Saddle Camp	64*	1986	1996	Summit
South Fork	319	1986	1996	Summit
Squaw Creek	310	1986	1994	Boundary
			1996	Tower
Sulphur	63*	1993	1996	Tower
Switchback	8*	1993	1996	Tower
Tower	51,483	1996	2001	Big Creek II
Trail	81*	1986	1996	Tower

* These acreages were entirely reburned by subsequent fires; acreages without an asterisk were involved in a subsequent reburn, but varying proportions of the original acreage were reburned.

Source: Summarized from digital fire atlas data available in the Blue Mountains province geographical information system.

In May 2000, a prescribed fire in northern New Mexico escaped the project area and then burned 42,858 acres and 235 residences near Los Alamos, New Mexico. Following this Cerro Grande Fire, the Federal Emergency Management Agency (FEMA) commissioned a study to predict changes in fire hazard through time. FEMA's objective was to estimate fuel dynamics and determine when future fire hazard would become greater than it had been prior to the Cerro Grande Fire (Greenlee and Greenlee 2002). The Cerro Grande fire hazard study predicted that there would be an increase in fire hazard after fire-killed trees fall, particularly for high-severity areas experiencing high amounts of tree mortality. Using a four-foot flame length and a spread rate of one mile per hour or more as criteria for high fire hazard, it was found that woody debris and down log accumulations increased the fire hazard after the sixth post-fire year (Greenlee and Greenlee 2002).

Local experience on the Umatilla NF corroborates the Cerro Grande study because reburn fires have occurred relatively soon after the initial fires (table 13).

When the Tower Fire (1996) reburned previous fires, for example, intense combustion of down logs and coarse woody debris resulting from the initial fires caused dramatic fire effects: tree seedlings that regenerated after the initial fire (generally averaging 3-5 feet in height) were not only killed during the reburn, they were consumed clear down to the soil surface (Powell 1998).

"When a large and unusually severe fire occurs in a wilderness environment, it ultimately creates a correspondingly large mass of heavy fuels, starting 12 to 15 years after the fire when much of the dead timber has fallen (Lyon 1984). This becomes incorporated into a new dense fuel bed with small conifers and large shrubs, which can readily support another severe wildfire, or 'double burn' (Barrett 1982; Brown 1975; Wellner 1970)" (Arno et al. 2000, p. 227). "Patterns laid down by previous fires can play a significant role in shaping future fires" (Perry et al. 2011).

Additional Information

There are hundreds of fire-related citations in the References section of this white paper, and only a few dozen of them are cited in this 5.11 wildfire section. General information about wildland fire ecology, mixed-severity fire regimes, fire management, and similar topics is provided by sources such as Agee (1993), Brown and Smith (2000), Keeley et al. (2009), Pyne et al. (1996), and Taylor et al. (2005).

At least 15 references provide place-based fire and fuels information for the Blue Mountains: Agee (1996b), Droske (2012), Hall (1980), Heyerdahl (1997), Johnson (1998), Marouka (1994), Mehringer (1997), Mutch et al. (1993), Olson (2000), Ryan and Pickford (1978), Thies et al. (2005, 2006, 2008), Thies and Westlind (2012), Williamson (1999), and Williamson and Agee (2002).

An excellent journal article synthesizing the ecological aspects of mixed-severity fire regimes is provided by Perry et al. (2011). A good synthesis paper about objectives and considerations for fuel treatments in forests of the interior West is provided by Reinhardt et al. (2008); it:

(1) summarizes objectives, methods, and expected outcomes of fuel treatments;
(2) highlights common misunderstandings and areas of disagreement; and
(3) synthesizes relevant literature to establish a common baseline for future fire and fuels planning.

The mountain pine beetle discussion (sections 5.8 and 5.10) describes potential interactions between insect outbreaks and wildfire (and see figs. 6 and 48 in particular). Crane and Fischer (1986) provide a useful discussion about relationships between lodgepole pine, wildfire, and mountain pine beetle. A recent paper by Hicke et al. (2012) summarizes the existing state of knowledge about interactions between bark beetle-caused tree mortality and wildfire. Other useful syntheses about relations between fuels, fire behavior, and beetle-attacked forests are provided by Bentz et al. (2010), Jenkins et al. (2008, 2012), and Simard et al. (2011).

Parker et al. (2006) synthesize current information about interactions among fire, insects, and pathogens. Interactions between fire and defoliating insects are summarized in Fleming et al. (2002), and in Lynch and Moorcroft (2008). Hummel and Agee (2003) modeled potential fire behavior associated with budworm-caused defoliation in mixed-conifer stands of the eastern Cascades in Washington. Many of the interactions between fire and coarse woody debris or down wood are summarized in Laudenslayer et al. (2002) (and also see Agee 2002a for this topic).

5.12 Wind

Wind is an important ecological process (Ennos 1997), not just for moist-forest ecosystems but for cold and dry forests as well. As a disturbance process, wind often functions in a dispersed fashion, creating small centers of tree mortality in a gap-phase pattern. Ecologically, a distinction should be drawn between endemic wind damage, which tends to be associated with unstable stands or portions of stands, and what can be termed episodic wind events affecting stable, resistant stands distributed across relatively large spatial areas (fig. 55) (Mitchell 1995).

Endemic wind generally creates small gaps within existing stands, and they are often centered on unstable conditions of either biotic (stem-decay, root disease, trees with high slenderness coefficients (Wang et al. 1998)) or abiotic (shallow or rocky soils) origins. Episodic wind events create openings large enough to function as new forest stands, and they tend to be associated with regional-scale windstorms occurring relatively infrequently. The Columbus Day windstorm of 1962 is a good example of a large, infrequent wind event, and although its impact was most severe for the coastal Northwest (Lynott and Cramer 1966), limited damage also occurred in the Bone Springs and Skyline Road areas of the northern Blue Mountains.

Even though well-quantified information about the historical disturbance regime for wind is lacking for the Blue Mountains (i.e., frequency, intensity, severity, and other disturbance characteristics included in table 3, page 17), existing data suggests that wind is a continuing source of tree death, is highly variable in annual recurrence rates, and causes variable types of damage (e.g., stem-snapped versus uprooted trees). It would be particularly helpful to understand the disturbance regime for episodic wind occurrence – how often do broad-scale wind events occur, and are they associated with certain physiographic positions or biophysical environments?

Although it is likely we lack sufficient data to place the episodic wind disturbance process in an appropriate temporal context, several major windstorms occurred in mid winter of 1989-1990 (fig. 55), leading to a suite of small salvage sales to harvest some of the windthrown timber (specifically, the Aim, Buck, Coyote, Jemina, Macwind, Pinbow, Prairie, Spout, Sugar, Trailhorse, Whiskey, and Windsum sales). The spatial pattern depicted in figure 55 provides useful clues about the size and dispersal of patches created when wind's influence occurs at a magnitude greater than the endemic (sub-stand) level.

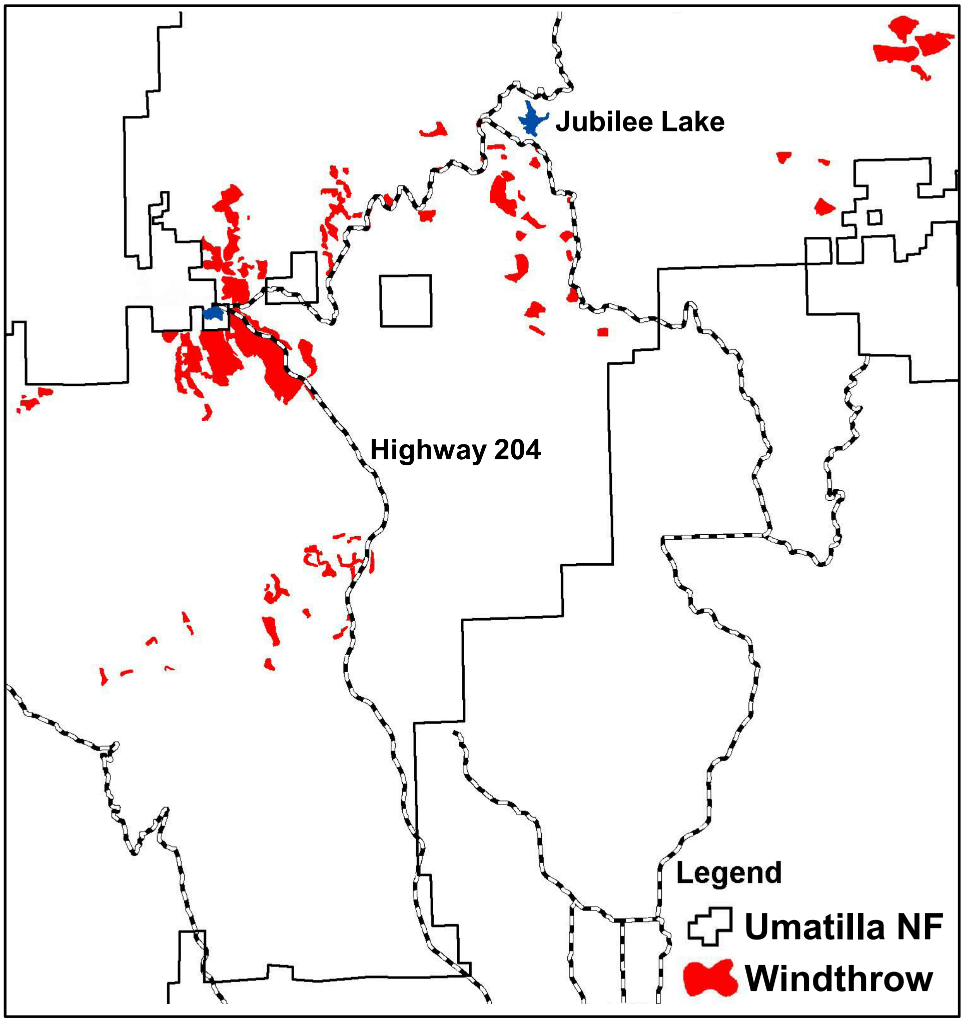


Figure 55 – Distribution of mapped blowdown areas associated with several large windstorms during the winter of 1989-1990 (source: unpublished records available at the Walla Walla Ranger District). Oregon Highway 204 and Jubilee Lake are labeled for orientation purposes. Total windthrow occurrence depicted in this image (red polygons) comprises 3,718 acres. This relatively severe windthrow episode occurred during one winter season (much on January 7-8, 1990), and resulted in a series of small salvage sales to harvest some of the windthrown timber (Aim, Buck, Coyote, Jemina, Macwind, Pinbow, Prairie, Spout, Sugar, Trailhorse, Whiskey, and Windsum sales). The spatial pattern depicted in this figure can provide useful clues about size and dispersal of patches created when wind's influence occurs at a magnitude greater than the endemic (sub-stand) level. Wind is often considered to be stochastic because as a disturbance process, many of its disturbance characteristics are unpredictable (i.e., size of area affected; spatial patch distribution on the landscape; frequency and return interval; and intensity or severity). Despite uncertainties related to windthrow, we should acknowledge that susceptibility to wind damage is predictable and related to site factors such as soil depth and texture (fig. 57), topographic or physiographic factors like slope position, and stand factors including mean tree height or stand density (fig. 58). Windthrow also has synergistic interactions with other disturbances such as spruce beetle (Holsten et al. 1999), root disease, and stem decay.

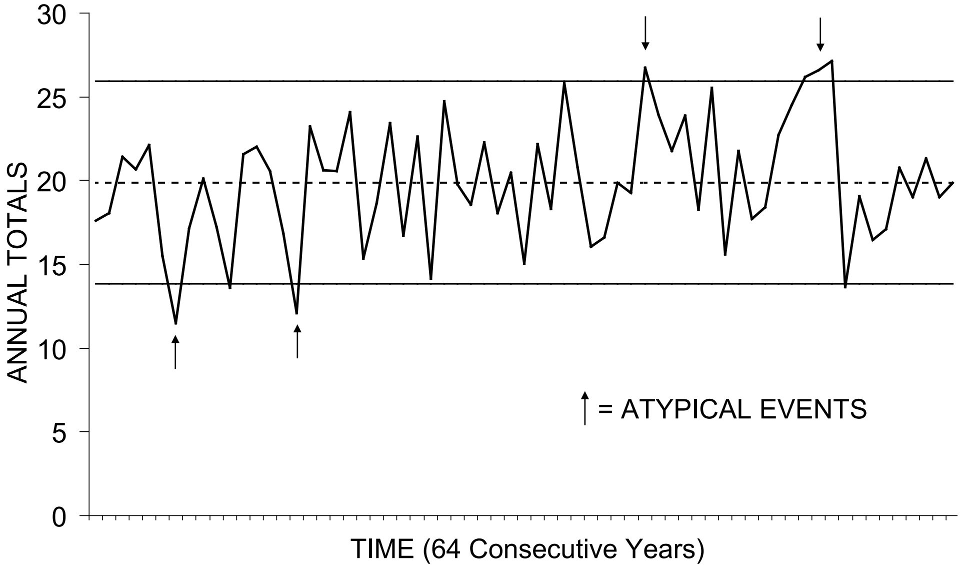


Figure 56 – Extreme or unusual environmental events have an important influence on ecosystems (taken from Powell 2000). Action of the environment on an individual plant or its overall community is neither uniform nor consistent because unusual events are quite normal (episodic wind storms, multi-year droughts, etc.) (Taylor 1934). This figure illustrates the concept by using the precipitation record from a weather station in the Blue Mountains. Two lines delineate a zone containing 90% of the variation around the record's 64-year mean (the dashed line midway between the two lines is the mean). Since the 90% was based on statistics (standard deviation), a different zone could have been established by selecting another value (67%, 80%, 95%, etc.). By selecting 90%, it was assumed that on average, 9 years out of 10 would have normal precipitation and the other year would be abnormal (either unusually high or low). In this figure, 4 periods escaped the 90% threshold zone and, by definition, would be considered unusual (arrows denote the unusual periods). Plant species exposed to this precipitation regime would need sufficient ecological amplitude to survive the unusual periods; otherwise, they may successfully colonize an area in the short term, but would not necessarily persist over the long term. Note that 90% was selected only to illustrate this concept; in actuality, each species would have its own particular definition of what constitutes extreme or unusual circumstances.

Although large windstorms and similar stochastic disturbance events are unpredictable, often tending to be characterized as 'extreme' in a disturbance ecology context, they have a very important influence on ecosystem development (Quine and Gardiner 2007). "Action of the environment on the organism or the biotic community is not uniform or consistent. Unusual features are quite 'normal.' Extreme conditions are probably more significant than are the more ordinary environmental relationships" (Taylor 1934). Figure 56, above, discusses the influence of extreme or unusual events on ecosystem development.

One of the primary concerns about wind damage involves partial cutting silvicultural systems, and how the marking guides for partial-cutting prescriptions should account for site, species, and stand factors affecting the windfirmness of residual trees. But this need has been complicated by the large number of factors contributing to windthrow risk, the complexity of interactions between factors, and the unpredictable nature of wind events (Mitchell 1995). Concern about partial cutting and the associated risk of wind damage has been long-standing, beginning with early accounts involving ponderosa pine timber sales in the Blue Mountains.

As of 1915, about 6,000 acres of western yellow (ponderosa) pine forest in the Blue Mountains had been cut over (Weidman 1921) by using what was referred to as the maturity selection system (O'Hara et al. 2010); all of the timber sales had been sold since 1910. The marking on maturity selection sales emphasized leaving about 20% of the thriftiest, soundest, and youngest trees distributed evenly across the sale area. On May 26, 1913, a windstorm blew down 792 yellow pines on 880 acres of cut-over units on the W.H. Eccles timber sale area. Then, on September 18, 1914, another severe storm blew over 808 yellow pines on the 1,624 acres cut over up to that time. This meant that in aggregate, almost 18% of the reserve stand had been blown over in just two years.

In response to the windthrow events, the Forest Service initiated an administrative study to examine windthrow occurrence in relation to site factors (landform, aspect, soils, etc.) or tree characteristics (height, crown size, crown density, crown class, root systems, stem factors, etc.). Study results were summarized in a letter from the District (Regional) Forester to Forest Supervisors in which he stated that marking guides would not be changed as a result of the study because the "information was collected in a single locality, and in other timber sales in yellow pine throughout the District the loss by windfall has not been excessive" (Smith and Weitknecht 1915).

Evaluating Windthrow Hazard

In addition to resistance ratings at the tree species level, managers generally consider topographic, soil, and stand characteristics when evaluating windthrow susceptibility (figs. 57-58). Stand metrics such as slenderness coefficient can also serve as good predictors of wind risk (Wang et al. 1998). [Slenderness coefficient is a ratio comparing tree height and diameter (in the same units such as inches or feet). Trees with an unbalanced height to diameter ratio are likely to become 'wet noodles' that cannot adequately support themselves; if neighboring support trees are removed or die, wet noodles tend to experience windthrow, snow breakage, or similar damage. In the interior Pacific Northwest, trees with a high slenderness coefficient (a height to diameter ratio of 80 or more) will generally behave as wet noodles, ultimately experiencing increased wind or ice damage (O'Hara and Oliver 1999, Wonn and O'Hara 2001).]

Considering a wide variety of resistance and susceptibility factors is important when evaluating windthrow risk because many interactions exist between wind and other disturbance processes (Gardiner and Quine 2000). It has long been recognized, for example, that trees harboring root disease (Furniss 1962) are more prone to windthrow, and that windthrown trees provide excellent habitat for bark beetles or wood borers, including Douglas-fir beetle (Furniss 1962) or spruce beetle (Schmid and Frye 1977).

A helpful, site-level indicator of previous windthrow activity is the presence of pit-and-mound microtopography, which results when windthrown trees create pits (the holes previously occupied by a root plate) and mounds (the soil hummocks created as a root plate decomposes and sheds the entrained soil). Windthrow on gentle topography produces root plates that are nearly vertical, which then creates horizontal mounds along one lip of a pit following root decomposition. Root plates on steep slopes, however, are leaning at such an angle that when they deteriorate, they fall over onto undisturbed ground and basically cause an inverted soil profile, with the surface soil underneath and the subsoil on top (Schaetzl 1986). These fine-scale wind effects on soils led to speculation that windthrow could actually stimulate organic matter decomposition for coastal forests by mixing the soils and improving aeration, but when this hypothesis was tested experimentally, positive effects were not obtained (Prescott et al. 2000b).

In addition to functioning as a macro-scale disturbance agent, a situation where wind causes tree mortality at either the gap or stand scales, wind also functions at a micro-scale where changes in forest stand dynamics, rather than tree mortality, is the ultimate result. When examining a suite of life-history traits for common tree species of the Blue Mountains, it is likely that one or more of them evolved in response to wind and how it influences inter-species competition in mixed forests.

One life history trait with a strong influence on inter-species competition is branch stiffness – once an overstory tree cohort gets relatively tall, there is often a period of crown shyness caused by tall trees abrading each other in the wind. Stiff-branched species tend to abrade their limber-branched neighbors, while also knocking the tops out of subordinate trees as they grow upward and attempt to move into a higher canopy position (Oliver and Larson 1996, Putz et al. 1984). In mixed stands of Douglas-fir and grand fir, wind provided grand fir with a competitive advantage because it has stiff branches, easily abrading Douglas-fir trees (Larson 1986).

As tree collide with each other in windstorms, this damping of wind energy may prevent them from toppling over, but it often comes at a high cost in terms of branch breakage, twig and foliage loss, and loss of buds and associated future growth. Although the worst damage often occurs during strong but infrequent storms, chronic wind action is thought to have the most inhibitory effect on lateral crown growth (Rudnicki et al. 2003). Crown abrasion might have more influence on stand structure and forest succession in boreal forest than temperate forest because there is increased branch brittleness as air temperature declines below freezing during the long boreal winters (Lieffers et al. 2001). Brittle branches have increased potential for crown abrasion (Rudnicki et al. 2003).

Figure 57 – Windthrow in spruce-fir forest of the northern Blue Mountains. Engelmann spruce is a shallow-rooted species, which increases its susceptibility to windthrow for almost any site condition. But abiotic factors can also contribute to increased wind damage because this site has an ash or loess layer above residual soils derived from Columbia River basalt or lacustrine (lakebed) sediments. These soils (typically Andisol soil types) have high water-holding capacity, an important feature of Engelmann spruce habitat. Flat areas and north-facing slopes have well-developed vegetation inhibiting soil erosion, allowing the ash to be retained on site. However, a relatively thick mantle of loamy, fine-textured tephra (ash) or loess lacks structural integrity, providing poor anchorage for tree roots. Silvicultural options may be limited for wind-susceptible sites like these: avoid tree harvest altogether; salvage windfall as storms continue to unravel the forest; or gradually open the canopy by using intermediate cuttings or the preparatory cut of a 3-step shelterwood (Alexander 1987, Roe et al. 1970).

Branch stiffness and crown abrasion are not the only traits that apparently have evolved in response to wind as an abiotic factor. Trees can bias the release of their seeds to capitalize on certain wind conditions, and some modeling and experimental evidence suggests that trees may respond to wind in ways that produce sustained updrafts in their immediate surroundings. This phenomenon would qualify as an important evolutionary adaptation because most seeds within a wind-dispersed species can be uplifted and are designed for long-distance dispersal: “seeds that fall within the canopy have no chance of long-distance dispersal; seeds that rise above the canopy do” (Horn et al. 2001).

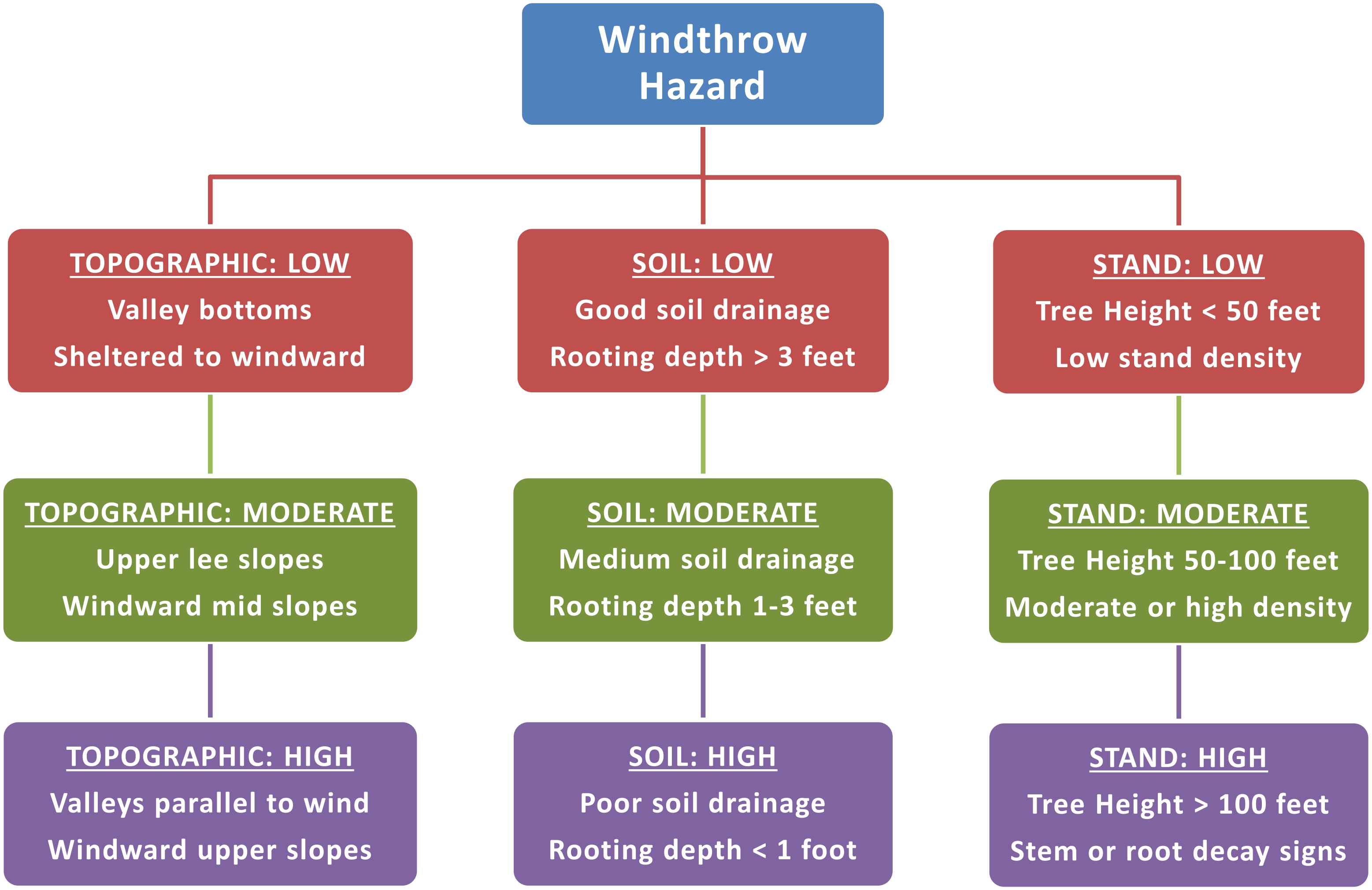


Figure 58 – Windthrow susceptibility, characterized as low, moderate, and high categories, and as related to topographic, soil, and stand factors (adapted from Mitchell 1995 and Wang et al. 1998).

5.13 Timber Harvest

Timber harvest is used to provide the wood products desired by human society. Forest practices have been used in the Blue Mountains for more than a century, beginning with the mining and settlement era in the 1860s and 1870s. The first commercial timber harvest in the northwestern pine region (eastern Oregon and eastern Washington) began around 1890 (Weidman and Silcox 1936). Early harvests were concentrated on private lands and tended to remove high-value species like ponderosa pine, Douglas-fir, and western larch. On dry-forest sites, much of the early harvest was directed at removing high-risk trees for western pine beetle attack; this approach was referred to as maturity selection (Keen 1950, O'Hara et al. 2010).

Beginning in the early 1940s, national forest tree harvests increased to meet a heightened demand for wood products during World War II, and to contribute raw materials for new housing. After the war, ponderosa pine and other species were intensively harvested to provide lumber for home construction, including window sash, trim and molding material, and door stock. A surprising amount of the ponderosa pine volume was used to produce many different types and sizes of boxes for fruit growers in the Yakima, Walla Walla, and Hood River valleys.

In the late 1950s, most every rural community supported at least one sawmill, including communities as small as Troy (fig. 59). In eastern Oregon, 49 communities had timber-processing mills – 33 communities had one mill, 10 communities had two, and 6 communities had three or more (Gedney 1963). Existing mill capacity for processing timber to manufacture wood products such as dimensional lumber is greatly reduced from historical levels. This trend has generated concern about the long-term viability of wood processing infrastructure in the Blue Mountains, and its associated ripple effects on economic and social resilience (Rainville et al. 2008).

Timber harvesting began in earnest in the mid 1950s (fig. 60), initially with partial cutting where only the most valuable trees were removed, and eventually evolving to small clearcutting sales on the north half of the Umatilla National Forest. Clearcutting accelerated in the 1960s. The clearcutting era represented a systematic harvesting program where the management cycle consisted of clearcutting, broadcast burning, grass seeding, and tree planting (Rollins 1982; fig. 61). These early clearcuts, now supporting second-growth stands of mixed conifers, have been thinned several times since the 1960s and 1970s.

Post-harvest seeding was not a universal practice, but on many areas grass seed was broadcast from helicopters, at a rate of 8 to 10 pounds per acre, to provide forage for livestock grazing, for soil erosion control, and to contribute to aesthetics. Tree planting occurred immediately after timber harvest, although the tree seed used to produce the planting stock may not have come from the same national forest on which the cutting occurred (cuttings on the Umatilla National Forest may have been reforested by using a seed source from the Wallowa-Whitman or Malheur national forests, for example) (Rollins 1982).

In the late 1960s, timber management practices for moist upland forests evolved from clearcutting to seed-tree or shelterwood cutting (note that all three of these practices are illustrated in fig. 14). By the mid 1970s, most of the moist-forest regeneration treatments were shelterwood seed cuts because they offered at least two advantages: (1) survival of both planted and natural seedlings was improved; and (2) increased survival allowed more cutting units to meet National Forest Management Act stocking standards within the required 5-year timeframe (Rollins 1982).

As described in the defoliating insect and mountain pine beetle sections, impacts from both of these disturbance agents caused unplanned changes in the timber management and reforestation programs by the late 1970s and early 1980s. For the tussock moth areas, timber managers examined defoliated areas and determined a tree mortality prediction (table 14), which was used when deciding whether salvage harvest would be attempted. During an accelerated salvage program, 40 sales were sold over a 2-year period. Reforestation of the salvaged areas (figs. 61 and 62), along with tree planting for unsalvaged areas where tussock-moth damage was substantial, caused the reforestation program on the Walla Walla Ranger District to increase from 600 acres annually in 1976 to 3,800 acres annually by 1980 (Rollins 1982).

Table 14: Relationship between defoliation caused by Douglas-fir tussock moth and tree survival probability.

Defoliation Status	Predicted Tree Mortality Outcome
0-30% missing foliage	Very high probability of survival
30-50% missing foliage	Moderate to high survival probability
50-75% missing foliage	Trees rarely die from defoliation alone, but moderate losses occur from bark beetles; some topkilling begins to occur
75-90% missing foliage	Low survival probability from defoliation directly; high losses from bark beetles are also expected
90-100% missing foliage	Very low survival probability from defoliation: 90% of trees in this defoliation class will die from defoliation effects alone

Source: Mason and Wickman (1984).

Timber harvest levels continued at high levels throughout most of the 1970s and 1980s (with a pronounced downturn occurring in the early 1980s during a timber recession). In response to changing societal values and expectations, timber harvest levels began declining precipitously in the early 1990s (fig. 60). Much of the reduction since the early 1990s is related to an emphasis on wildlife protections and other non-commodity values, many of which are reflected in amendments to the Land and Resource Management Plans for Blue Mountains national forests (specifically the PACFISH and Eastside Screens amendments).

Figure 59 – Sawmill located at Troy, Oregon, showing associated log holding pond, log decks, and a teepee-style burner for disposing of sawdust and other milling waste. From the 1950s through the 1970s, almost every community in eastern Oregon had at least one wood-processing facility (Gedney 1963).

Figure 60 – Timber harvest trend for the Umatilla National Forest, 1922-2010 (this chart is based on cut volume, not sold volume). Harvest levels were low during the custodial era (before the 1940s), and high between the late 1950s and early 1990s (except for a timber recession in the early 1980s).

Figure 61 – Young, moist-forest stand with moderate species diversity (mostly lodgepole pine in foreground, but Douglas-fir, Engelmann spruce, and other species in middleground), sinuous harvest-unit boundaries, and some aspects of variable-retention harvest including unharvested reserve islands (upper left corner) (DeLong 2011, Franklin et al. 2007; also see Box 3).

Figure 62 – Results of historical management practices (timber harvest, broadcast burning, grass seeding, and related activities) on a moist-forest area. This is Tincan Unit #1, a site in the grand fir/queencup beadlily plant association. It was clearcut and broadcast burned in 1972, planted in 1973 to grand fir (91%) and western larch (9%), gopher baited in 1974, and then replanted in 1975 to grand fir (28%), western larch (50%), and Engelmann spruce (22%). The inspection trip shown here depicts the unit in 1979, 7 years after harvest. The unit was eventually machine scalped as a site preparation treatment in 1983, and also replanted in 1983 to western larch (13%), ponderosa pine (52%), and Douglas-fir (35%). It was replanted again in 1986, and gopher baited again in 1987. Tree seedling survival on this site was inhibited by frost, small mammal populations, and intermittently high levels of elk herbivory. The heavy equipment use, broadcast burning with a high fireline intensity, and grass seeding with exotic species (the combination of intermediate and pubescent wheatgrasses comprised 35% cover) were also major contributing factors. The result shown here was not common in response to moist-forest regeneration cutting (app. 2), but it did happen occasionally (units in the Notch, Herren, Coal, Texas, Hollywood, Jarboe, and Ninemile timber sales from the 1970s to 1990s had similar results). Ecologically, much of this outcome relates to the fact that moist forests are inherently complex (Puettmann et al. 2009); certain management actions of the 1960s and 1970s were intentionally designed to reduce complexity rather than maintain it (Christensen 1997) (e.g., using high-intensity prescribed fire to make a site as 'clean' as possible by combusting the down wood, seeding exotic grass species at high rates to maximize forage production for livestock and wild ungulates, etc.). Resource managers of this earlier era, however, did not have access to ecological classifications or successional guides to help them understand ecosystem processes and predict how moist-forest sites might respond to timber harvest, site preparation, and similar management activities (section 6.5). And knowing only the plant associations of a project area may be inadequate; we also need to understand the implications of alternative successional trajectories (Green and Jensen 1991), such as those described by Clausnitzer (1993) for many moist grand fir associations of the Blue Mountains.

5.14 Non-Native, Invasive Forest Species

In a forestry context, a good example of enduring change caused by a non-native species is provided by white pine blister rust, a fungus pathogen introduced from Europe by way of western Canada about 1910. White pine blister rust affects all of the five-needled pine species, with western white pine and whitebark pine being affected most seriously in the Blue Mountains. By the 1940s, blister rust occurred throughout the range of western white pine in the interior Pacific Northwest.

Western white pine, a mid-seral tree species, is occasionally found on cool moist, cool wet, and warm moist sites in the upper montane and lower subalpine zones (Powell 2000). Table 15 shows the seral status of white pine and other tree species by plant association (Powell 2000). Western white pine was characterized as having a restricted geographical distribution in the Blue Mountains (Haig et al. 1941). In actuality, it has a relatively wide distribution as a minor species in mixed-conifer forests, seldom comprising a plurality of the basal area in any individual stand.

Due to changes wrought by fire suppression (encouraging invasion by subalpine fir), bark-beetle outbreaks, white pine blister rust, and other factors, it is believed that white pine in the Blue Mountains was more abundant historically than at present: “White pine was at one time distributed over the entire Forest but it was killed out by fires, to which it is so particularly susceptible, years ago. It is thought that white pine would be an excellent tree to plant on all the burns found on the higher altitudes of the Wenaha. It attains good size and form in such places, and its wood is superior to any of the other species with which it occurs in such places” (Bright and Powell 2008).

Land managers tried a variety of strategies to eradicate blister rust, including pulling Ribes plants (the rust’s alternate host), removing infected branches before the fungus could migrate to the main stem, and injecting antibiotics into the bark of affected trees (Fins et al. 2001). None of these measures worked well, and efforts to directly control the rust were abandoned in the late 1960s. Currently, a multi-faceted approach featuring tree breeding for rust resistance, along with careful matching of rust-resistant planting stock to potential reforestation sites, is being employed across white pine’s range (Fins et al. 2001, Neuenschwander et al. 1999).

Blister rust also affects other five-needled pines such as bristlecone pine, limber pine, and whitebark pine, contributing to wildlife habitat deterioration and other changes in ecosystems dominated by these species (Keane et al. 1995, Tomback et al. 2001). Of these pines, blister rust is particularly severe on whitebark pine. Blister rust was first discovered on whitebark pine on August 4, 1936 on the south side of Mount Hood in Oregon. According to the survey, blister rust had not yet affected whitebark populations in the Blue and Wallowa Mountains of northeastern Oregon (Childs et al. 1938), although that is certainly not true today.

Table 15: Seral status of tree species for plant associations of the moist upland forest potential vegetation group.

PLANT ASSOCIATION	PP	DF	WL	LP	WP	ES	GF	SF
ABGR/ACGL	ES	MS	ES		MS	LS	PNC	
ABGR/BRVU		MS	ES			LS	PNC	
ABGR/CLUN	ES	MS	ES	ES	MS	LS	PNC	
ABGR/GYDR							PNC	
ABGR/LIBO2	ES	MS	ES	ES	MS	LS	PNC	
ABGR/POMU-ASCA3			ES			LS	PNC	
ABGR/TABR/CLUN	ES	MS	ES			LS	PNC	
ABGR/TABR/LIBO2		MS	ES		MS	LS	PNC	
ABGR/TRCA3			ES			LS	PNC	
ABGR/VAME	ES	MS	ES	ES		LS	PNC	
ABGR/VASC-LIBO2		MS	ES	ES		LS	PNC	LS
ABLA2/CLUN			ES			LS		PNC
ABLA2/LIBO2			ES			LS		PNC
ABLA2/STAM								
ABLA2/TRCA3				ES		LS		PNC
ABLA2/VAME			ES	ES		LS		PNC
PSME/ACGL-PHMA	ES	PNC						
PSME/HODI	ES	PNC						

Sources/Notes: Plant association symbols are described in appendix 1; species codes (column headings) are described in a footnote to table 11. Seral status was derived from Clausnitzer (1993), Hall (1973), Johnson and Clausnitzer (1992), and Steele et al. (1981). Seral status codes are: PNC = dominates the potential natural community; LS = late seral; MS = mid seral; ES = early seral; A = accidental occurrence (Hall et al. 1995).

Another exotic organism affecting moist upland forests is balsam woolly adelgid, which was introduced into North America from Europe around 1920 (Livingston et al. 2000, Mitchell et al. 2001, Ragenovich and Mitchell 2006). The aphid-like insect has spread throughout the subalpine fir host type in the Blue Mountains, causing concern about whether this forest type will persist over the long-term in this portion of its range. The adelgid causes 'gouting' on branches, gradually killing tree crowns and eventually the whole tree (Mitchell and Buffam 2001). Trees of all size classes are killed, with heavy lichen growth replacing live foliage as the tree declines and then dies.

In many portions of the subalpine fir range in the Blue Mountains, more than half of the host type has been killed, with variable effects on the small regeneration (in some areas, subalpine fir seedlings and saplings seem to be surviving, whereas in others there appears to be complete mortality). Balsam woolly adelgid also affects grand fir (Livingston et al. 2000), although at this time, damage in the Blue Mountains seems to be confined exclusively to the subalpine fir host type (fig. 63).

Figure 63 – Subalpine fir mortality caused by balsam woolly adelgid (BWA) in northern Blue Mountains. Relatively large subalpine firs throughout the northern Blue Mountains look like the tree in this photograph – they were killed by BWA and now support a copious lichen growth. In the Blue Mountains, BWA does not appear to affect small trees (seedlings and saplings) to anywhere near the same extent as larger trees, although this generalization seems to vary from one geographical area to another. BWA threatens long-term viability of subalpine fir in the Blue Mountains.

5.15 Climate Change

The Intergovenmental Panel on Climate Change concluded with high confidence (8 out of 10 chance) that "disturbances such as wildfire and insect outbreaks are increasing and are likely to intensify in a warmer future with drier soils and longer growing seasons, and to interact with changing land use and development affecting the future of wildland ecosystems" *Parry et al. 2007, page 56*

Why climate change? Why include climate change in section 5 as a disturbance process like wildfire or wind? The previous paradigm was that climate functions primarily as a 'background' phenomenon, and although there was wide recognition that climate is not unchanging, the changes were believed to occur on multi-century or millennial time scales. Under this homeostatic view of climate – it was broad-scale, changed slowly, and was resistant to anthropogenic control – it was believed that physical site factors (elevation, geology, soil type, etc.) exerted more influence on variation in biophysical environments than climate. But since climate is changing more rapidly than it did historically, it is now acknowledged that climate change functions as a pervasive, overarching influence with dual roles as a stand-alone disturbance process (e.g., drought), and as a factor that 'ramps up' other disturbance processes (wildfire, defoliating insects, etc.) to substantially higher levels than they reached in the past.

Climate Change Background

A pressing matter of crucial concern is the threat of a long-term increase in the earth's surface temperature. This threat goes under several names – climate change, global change, greenhouse effect, and others. 'Greenhouse effect' might be an apt term because the principle of a greenhouse applies in this situation – the enclosing shell allows passage of incoming solar radiation but traps a portion of the reflected infrared (longwave) radiation, warming the greenhouse's interior above the outside temperature. Greenhouse gases in the earth's atmosphere play a similar role – they function to raise the temperature of the earth and make it habitable. Without greenhouse gases, the surface of the earth would be about 30 °C (86 °F) cooler than it is today, rendering human life impossible. But 'global change' could also be an appropriate moniker because climate change is obviously global in the sense that greenhouse gases created anywhere in the world have potential impacts everywhere.

Since the beginning of what is termed the 'industrial revolution' (mid 1700s), combustion of fossil fuels, together with persistent deforestation and other activities, led to an increase in the carbon dioxide content of the atmosphere by about 20 percent. In the last three decades alone, it increased by 8 percent, and scientists predict an approximate doubling by the middle of the 21st century. [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 trace gases (carbon monoxide, ozone-depleting chemicals, halocarbons, etc.).] Note that currently high and increasing levels of carbon dioxide in the atmosphere are unprecedented, at least when considering trends over the past 24 million years (Pearson and Palmer 2000).

Instrumented temperature records, the ice composition of long-lived glaciers and ice fields, and tree rings or other proxy sources show that the earth has warmed about 0.5 °C (1 °F) over the past 100 years. Some climate models predict that during this 21st century, temperatures could rise by 1.5 to 4.5 °C, or about 0.3 °C per decade. This might not seem like much change, but historical studies showed that past episodes of warming and cooling occurred at a rate of only about 0.05 °C per decade, which was sufficient to cause ice ages and major dislocations for human agrarian societies (Fagan 2002, Lomborg 2001, Mann 2006). A recent analysis concluded that the decade of the 1990s in general, and the year 1998 specifically, were likely the warmest for the northern hemisphere in at least a millennium (Mann et al. 1999). [But this source was published before the even warmer decade of the 2000s.]

Climate change effects are not expected to be uniform – in the northern hemisphere, polar regions will warm faster than equatorial zones, and the centers of continental landmasses are expected to become drier than the peripheries. In ice ages of the past, weather changed gradually enough to allow plants and animals to migrate and survive; the rapid pace of change occurring now may be too quick for many organisms to adjust to modified conditions. For this reason, some of the biggest impacts of climate change could involve agriculture and forestry and, of the two, forestry probably has fewer mitigation or adaptation options than agriculture because forestry has much longer intervals between successive generations (narrative to this point in section 5.15 is based primarily on Karl et al. 2009).

Climate Change Implications For Moist-Forest Management

Much of the concern about climate change relates to how it might affect baseline climate conditions. But would climate change effects be additive, subtractive, or neutral on baseline temperature and moisture relationships, and would their magnitude be great enough to exceed the environmental tolerances of existing plant species (table 16 and fig. 64)? If the answer to the second question is yes, then one likely effect of climate change could be extirpation of certain plant species, and their associated fauna and ecosystem services, from portions of the Blue Mountains.

Figure 64 shows the predicted future distributions for four existing tree species of the Blue Mountains (ponderosa and lodgepole pines, Engelmann spruce, and Douglas-fir), and for two species that do not currently occur in the Blues but might be expected to migrate here under future climate conditions (Gambel oak and pinyon pine). Figure 64 suggests that Engelmann spruce and lodgepole pine will experience significant contraction under future climate conditions (and other research also came to similar conclusions; see Coops and Waring 2011 as an example).

Table 16: Selected life history traits with relevance to climate-change adaptability.

	Ponderosa pine	**Western larch**	**Lodgepole pine**	**Western white pine**	**Douglas-fir**	**Engelmann spruce**	**Grand fir**	**Subalpine fir**
Tolerance to shading	L	L	L	M	M	H	H	H
Tolerance to full sunlight	H	H	H	H	M	L	L	L
Seral status	Early	Early	Early	Mid	Mid	Late	Late	Late
Tolerance to frost	L	L	H	H	L	H	M	M
Tolerance to drought	H	M	M	M	M	L	M	L
Rooting habit (depth)	D	D	M	M	D	S	S	S
Fire resistance	H	H	L	M	M	L	L	L
Evolutionary mode	Inter.	Inter.	Spec.	Gen.	Spec.	Inter.	N.R.	N.R.
Seed germination on charred or ashy soil	IN	NE	NE	IN	IN	RE	IN	N.R.
Maximum seed dispersal distance (feet)	120	150	200	400	330	120	200	100
Potential for regeneration in the open	H	H	H	H	H	M	L	L
Overall reproductive capacity	H	H	H	H	H	M	M	M
Potential initial growth rate (first 5 years)	H	H	H	M	M	L	M	L

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; NE, no effect; and RE, reduced. Overall reproductive capacity considers minimum cone-bearing age, seed crop frequency and size, seed soundness, and related factors. Evolutionary mode refers to the amount of genetic differentiation and is an indicator of how well a species could adapt to future climates (Gen. is generalist; Inter. is intermediate; Spec. is specialist; N.R. is not rated; source is Rehfeldt 1994).

Note that the tree-range distributions in figure 64 are based on climate envelope models, and it has been observed that these statistical models tend to overemphasize the role of climate in determining tree growth and mortality, while neglecting the influences of competition, dispersal barriers, carbon fertilization, soil characteristics, insects and diseases, topography, and macroclimate (Loehle and LeBlanc 1996). As is demonstrated by the maps for Gambel oak and pinyon pine (fig. 64), these models suggest that some tree species can thrive in climate zones beyond their present range limits.

Since trees have long intergenerational periods, they have relatively slow migration rates. This means that without assisted migration (e.g., intentional movement of trees to new sites where it is believed they will be better adapted), it is expected that existing species will gradually become less adapted to their current habitats. As adaptation falters, forests will probably experience more episodes of dieback or decline (often termed syndrome) characterized by interaction of several biotic and abiotic factors acting either simultaneously or successively (Schütt and Cowling 1985).

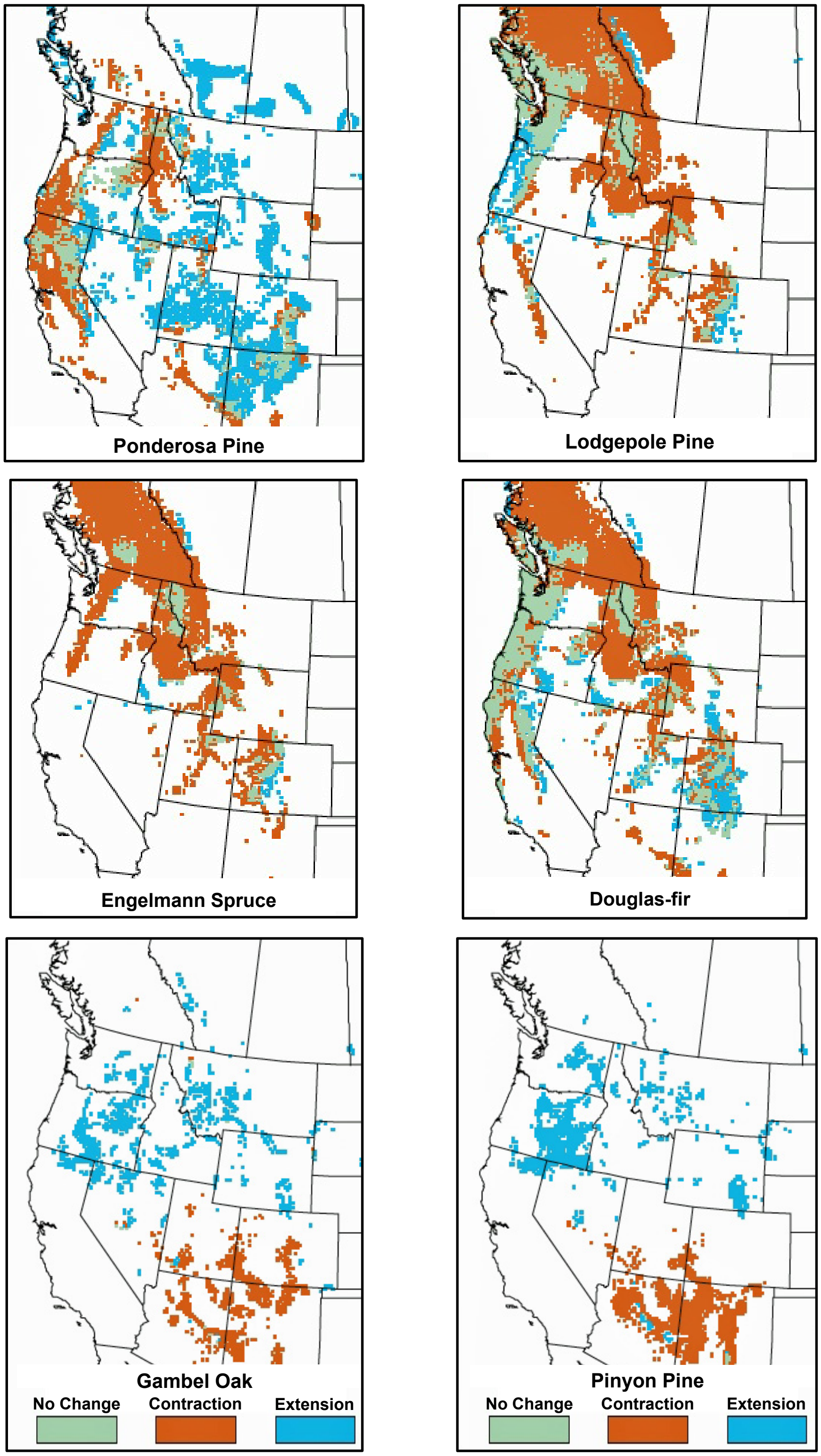


Figure 64 – Predicted future species distributions for selected tree species of the Blue Mountains (adapted from Thompson et al. 1998). The top four maps show future distributions of species often found on moist sites; the bottom maps show species that could migrate into the Blue Mountains in response to a warmer and dryer future (neither species occurs there now).

When considering possible effects of climate change on future tree species distributions for the Blue Mountains (such as Devine et al. 2012), it is important to examine a suite of life history strategies because any individual trait offers physiological or morphological trade-offs preventing a species from being optimally adapted to every possible environment (McCune 1988). A small sample of important life-history traits for eight primary conifers of the moist-forest zone is provided in table 16.

Certain life history traits in table 16, such as 'tolerance to frost,' might seem unrelated to climate change, at least in the context of global warming. But the cold hardiness of trees has apparently been influenced by climate change, with boreal forests experiencing earlier loss of cold hardiness in response to early-spring warming (late April to early May), followed by severe frost damage during subsequent cold snaps in mid spring (mid to late May) (Man et al. 2009). Before the onset of climate change, frost damage in mid May was an unusual event because boreal trees had not lost cold hardiness at that point in the spring.

Where can life-history information be found? In addition to a detailed autecological summary prepared by Minore (1979), helpful life history information is provided by the North America silvics manuals (Burns and Honkala 1990a, 1990b), the USDA Fire Effects Information System (Fischer et al. 1996), and by on-line databases available through the National Biological Information Infrastructure.

When considering precipitation patterns, it's not just the potential for more and longer droughts in the future that is problematic – it's the projected change in precipitation form, with less being received as snow and more as rain (fig. 65). This trend might improve forest growth in the short-term because as the growing season lengthens into early spring, spring rains and maximum soil moisture would coincide with a period when temperatures are reaching an optimum for tree growth. And since the Blue Mountains have a summer-dry, Mediterranean climate where soil-based snowmelt storage is crucial for sustaining plant growth across a relatively long growing season, a change in precipitation from snow to rain is likely to induce earlier summer plant dormancy, lengthen the fire season, shorten the wetland saturation period, and affect many other ecosystem services (van Mantgem et al. 2009).

A 'Dust Bowl' (circa 1930s) magnitude drought has occurred once or twice per century over the previous 300-400 years (Woodhouse and Overpeck 1998), and multi-year droughts are common in the western U.S. over the past millennium (Cook et al. 2004). Climatic drought is projected to be more common in the future because mid-summer temperatures are expected to be higher, and summer precipitation amounts lower, than at present. Sometimes, there is a perception that moist environments have a surplus of moisture, and that their relatively high precipitation amounts buffer them against the effects of drought and similar climate changes. But, new evidence suggests that most tree species maintain only a small hydraulic safety margin, and that moist or wet forest is just as vulnerable as dry forest, if not more so, to drought-induced tree mortality in the future (Choat et al. 2012).

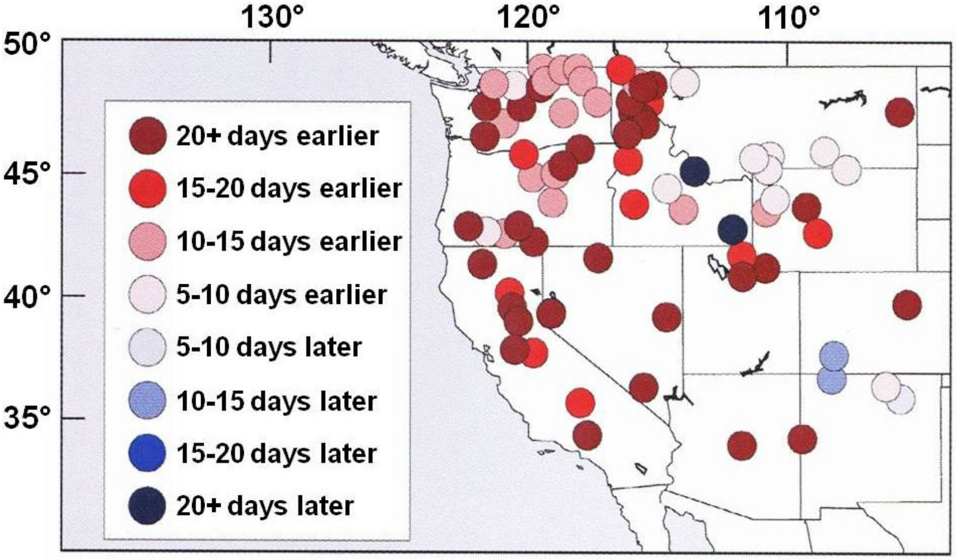


Figure 65 – Recent changes in spring snowmelt runoff timing for the western United States (Karl et al. 2009, p. 33). This chart shows trends in streamflow runoff timing for the 1948-2000 period; Blue Mountain river basins occur in a zone where runoff occurred 10-20+ days earlier for the 1948-2000 period than it did previously. Ongoing climate change is expected to exacerbate this trend (Furniss et al. 2010, Stewart et al. 2004).

Van Mantgem et al. (2009) found that widespread increases in mortality of old trees across the western United States was linked to regional warming; predicted trends in future temperature and precipitation suggest that bouts of increased tree mortality could occur more frequently, and with greater intensity or magnitude, than has been observed recently (van Mantgem et al. 2013). “If projections of a warmer and possibly drier climate are realized over the next century, massive tree mortality is a likely consequence, with major impacts on many of the values society places on forests, such as timber, recreation, wildlife habitat, watershed protection and reduction in atmospheric carbon dioxide concentration” (Birdsey and Pan 2011).

Climate Change And Moist Forest Restoration

“If denial ain’t just a river in Egypt,
despair ain’t just a tire in the trunk”
Al Gore

If moist upland forests are to have a reasonable opportunity for persistence under the future climate regime, restoring conditions more similar to the historical spatial characteristics of a heterogeneous, mosaic pattern is likely to function as a useful startpoint. Sustainable, moist-forest conditions can be achieved, even in the context of climate change, by completing the following activities: (1) reintroduce mixed-severity fire to change fire-free intervals from centuries to decades; (2) reduce accumulated surface fuels to levels ranging between 10 and 30 tons per acre (Brown et al. 2003); (3) reduce dense canopy and ladder fuels to a canopy bulk density level of 0.05 kg/m^3 or less (Powell 2010); and (4) increase the representation of fire-resistant trees: ponderosa pines, western larches, and large-diameter Douglas-firs.

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

There is no assurance that current amounts and spatial configuration of moist forest will remain constant in the future, particularly in response to increased disturbance levels under climate change. Research suggests that changes in fire and other disturbance processes due to climate feedbacks could have an important influence on moist-forest vegetation conditions. Carroll et al. (2004) showed that a shift in mountain pine beetle outbreaks occurred with warming climate, leading to increasing outbreak frequency in areas with previously unsuitable climatic conditions and a decline in previously suitable areas.

"Nitschke and Innes (2008a) found that warming by 4 °C (expected by the 2080s) would increase fire size (from a mean of 7,961 ha to 19,076 ha), increase fire severity (by 40% in spring, 95% in summer, and 30% in fall), increase fire season length (and consequently fire frequency, by 30%), increase the risk of crown-fire ignition and severe fire behaviour (by 4% to 7%), and decrease the extent of fire-free areas (-39%)" (Haughian et al. 2012). Conclusions reached in the Nitschke and Innes (2008a) study are similar to other investigations involving the interior Pacific Northwest (Flannigan et al. 2000, Rogers et al. 2011, Spracklen et al. 2009) – Littell et al. (2010) projected that the area burned in the Northwest could double or triple (or more) by the 2080s (fig. 66).

"The divergence in fire and climate since the mid 1800s has created a fire deficit in the West that is jointly attributable to human activities and climate change, and unsustainable given the current trajectory of climate change" (Marlon et al. 2012). The conclusion from Marlon et al. (2012) demonstrates that even though recent levels of fire activity (since the mid 1980s) are viewed as uncharacteristically high by contemporary fire managers, particularly when interpreted in the context of digital fire atlas data available from 1910 to the present, they are actually low when considered in the context of fire activity for the previous 3,000 years.

This means that recent levels of increased fire activity (i.e., levels experienced since the mid 1980s) are still perceived as a fire deficit when compared with a much longer temporal baseline (Marlon et al. 2012), and it is sobering to contemplate that climate change is expected to shift the fire-activity baseline to a significantly higher level, at least for certain fire regimes (fig. 66; Littell et al. 2009). And since the mid

1980s, many western states, including Arizona, New Mexico, Colorado, and Oregon, have all established records for their largest recorded forest fire. Over the past 10 years in the western U.S., there have been at least five dozen fires reaching 100,000 acres or more in size.

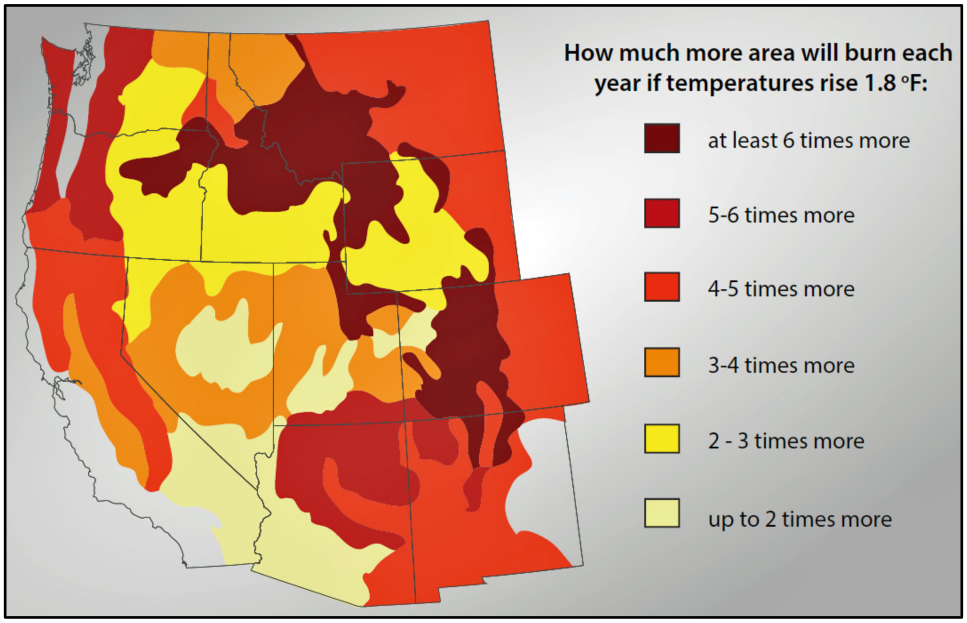


Figure 66 – Predicted increase in area burned by wildfire as associated with a mean annual temperature increase of 1 °C (1.8 °F), shown as the percentage change relative to the median annual area burned during 1950-2003 (source: Climate Central 2012). Results are aggregated to ecoprovinces (Bailey 1995) of the western United States. Climate-fire models were derived from National Climatic Data Center climate division records and observed area burned data following methods described in Littell et al. (2009). The prediction shown here is similar to several reports from the National Research Council showing at least a quadrupling of area burned in the western U.S. with each 1 °C of temperature increase (figure adapted from figure 5.8 in National Research Council 2011).

This prediction is alarming because over the period from 1970-99 to 2070-99, an increase in average annual temperature of 3.3 to 9.7 °F is projected, depending largely on whether global emissions eventually decline (the B1 greenhouse gas emissions scenario) or continue to rise (the A1B, A2 emission scenarios), and the temperature increase is projected to be largest in summer.

Mitigation And Adaptation In Response To Climate Change

Many of the policy proposals being considered to address climate change are based on mitigation – reducing greenhouse gas emissions from energy use and land-use changes to minimize the pace and magnitude of climate change. While mitigation is crucial, adaptation to climate change is increasingly viewed as a necessary and complementary strategy to mitigation (Joyce et al. 2009). Table 17 provides adaptation strategies proposed for the National Forest System and pertaining to upland forest vegetation. Table 17 also describes the predicted compatibility of active management treatments with climate change adaptation strategies.

Table 17: Estimated compatibility of climate change adaptation strategies and active management activities for moist upland forests.

Climate Change Adaptation Strategies	Compatibility With Moist Forests
Improve the capability of ecosystems to withstand uncharacteristically severe drought, wildfires, and insect outbreaks at landscape scales.	Thinning and similar active management practices might be necessary to improve the resistance and resilience of moist-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 16).
Reduce homogeneity of stand structure and synchrony of disturbance patterns across broad landscapes by promoting diverse age classes and species mixes, stand diversities, and genetic diversity.	This strategy could best be addressed by perpetuating age-class diversity, introducing additional species diversity when appropriate, and trying new genotypes offering better environmental fitness.
Reset ecological trajectories to take advantage of early successional stages that are adaptive to present rather than past climates.	Composition could be changed to favor early-seral species with high tolerance or resistance to drought, open conditions, and fire (table 16).
Use historical ecological information to identify environments buffered against climate change and which would be good candidates for conservation.	Many literature sources provide historical information with relevance for moist-forest ecosystems (Bright and Powell 2008, Gannett 1902, Munger 1917, and others).
Encourage local industries that can adapt to or cope with variable types of forest products because of the uncertainty about which tree species will prosper in the future.	Small-diameter trees could be removed periodically as restoration activities are implemented and, depending on circumstances, they could be used for biomass products.
Reforestation after disturbance may require different species than were present before the disturbance to better match site-level changes associated with climate change.	We can use species life-history data such as fire resistance and drought tolerance (table 16) to reforest with species having high resilience to future climates. But should we also consider new species (adaptive migration)?
After a disturbance event, use intensive site preparation activities to remove competing vegetation (Royo and Carson 2006) and replant with high-quality, genetically appropriate, and diverse stock.	This recommendation is similar to the one just before it, but with additional detail. It is feasible to use site preparation before planting, but any ‘intensive’ measures need to ensure protection of soil integrity.

Climate Change Adaptation Strategies	Compatibility With Moist Forests
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 moist upland forests. A rapid response to mortality would help address increased fire and insect risk as related to climate change impact.
Plan for higher-elevation insect outbreaks, species mortality events, and altered fire regimes.	It is expected that some fire regime III (mixed-severity) areas could transition to fire regime I (low severity) as the future climate warms and dries.

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

Table 17 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 and mid-term adaptation if they also reflect goals for other ecosystem services such as late-old structure and water quality (Joyce et al. 2009). And note that projected climate change is expected to influence more than just wildfire – temperature has been shown to play a major role in determining the abundance and distribution of insects and pathogenic fungi (Kliejunas 2011, Kliejunas et al. 2009), and elevated CO_2 levels are associated with increased growth and aggressiveness of tree pathogens (Pinkard et al. 2011).

Proposed restoration activities are expected to improve the adaptive capacity (Olsson et al. 2004) of moist upland forests in the Blue Mountains, particularly by alleviating the chronic stress associated with high tree density levels. Dense tree stands exist in a sort of perpetual physiological drought because there is not enough soil moisture to meet the water needs of all trees; thinning treatments are used to alleviate moisture stress, and allow residual trees to survive and continue growing. Since climate change could amplify the effects of density-caused stress by exposing forests to physiological and climatic drought simultaneously, the need for thinning is expected to be much greater in the future than at present, particularly because the improved physiological tree vigor resulting from thinning allows trees to produce more of the resins important for repelling insect and disease attacks (Kolb et al. 1998, Mitchell et al. 1983, Pitman et al. 1982, Safranyik et al. 1998).

We should also evaluate new silvicultural approaches for improving intra-stand heterogeneity in a climate change context, especially when considering whether to apply them across large spatial extents. One example is variable-density thinning (VDT) with skips and gaps (see the Young Stand Development discussion in section 6) – this approach offers many advantages in terms of improving heterogeneity for moist-forest plantations, but research found that when trees are retained in an even distribution (in contrast to a heterogeneous VDT approach with skips and gaps), the change in soil water equivalent, which is related to snow accumulation, was significantly increased when compared with untreated controls (Woods et al. 2006).

Management practices such as assisting species migration, creating porous landscapes (e.g., managed matrix providing habitat conditions through which species can move), and increasing genetic diversity in species planting mixes may be appropriate responses to changing climatic and disturbance regimes. Land managers, for example, could focus on increasing the resilience of ecosystems to rapid climate change by establishing greater plant diversity (table 17) (Moore 2005).

Managers could also use reference conditions from drier, lower, and more southerly sites to establish a 'future range of variation' (FRV), which would allow them to examine opportunities for species migration into, and across, areas currently classified as moist forest, while also accounting for how moist-forest areas might evolve (change) in the future (Hessburg et al. 2013).

This FRV concept recognizes that the last decade of the 20th century was the warmest of the past millennium (Mann et al. 1999), perhaps suggesting that the relatively xeric climatic conditions of the Holocene era (app. 6,000 years before present) may serve as a better analogue for near-term climate than the mesic conditions prevailing during the past two centuries (Mock and Brunelle-Daines 1999).

The approaches being considered to address climate change are near-term (mitigation or resistance) and far-term (adaptation or resilience). These are not 'either-or' options – both responses are needed to meet the climate change challenge. "In the long term, a sustainable forest management strategy aimed at maintaining or increasing forest carbon stocks, while producing an annual sustained yield of timber, fibre or energy from the forest, will generate the largest sustained mitigation benefit" (Intergovernmental Panel on Climate Change, Fourth Assessment Report, 2007).

Since climate change has already progressed to a point where some effects are unavoidable, or already occurring (fig. 65 is an example), the adaptation strategy functions as an insurance policy to protect ourselves and our natural resource investments (plantations, etc.) from some of the future impacts of global change.

The direct effects of climate change on temperature and precipitation, in combination with indirect effects related to wildfire, insects, and other disturbance processes (Dale et al. 2001, Logan et al. 2003), could detrimentally affect the future provision of old-forest structure, properly functioning soil and water services, wildlife habitat, animal and plant diversity, recreational opportunities, and carbon storage. Climate modeling, for example, suggests that Engelmann spruce and western larch could be extirpated from the Blue Mountains by the early 22nd century (Rehfeldt et al. 2006).

Accounting For Increased Future Fire Occurrence Due To Climate Change

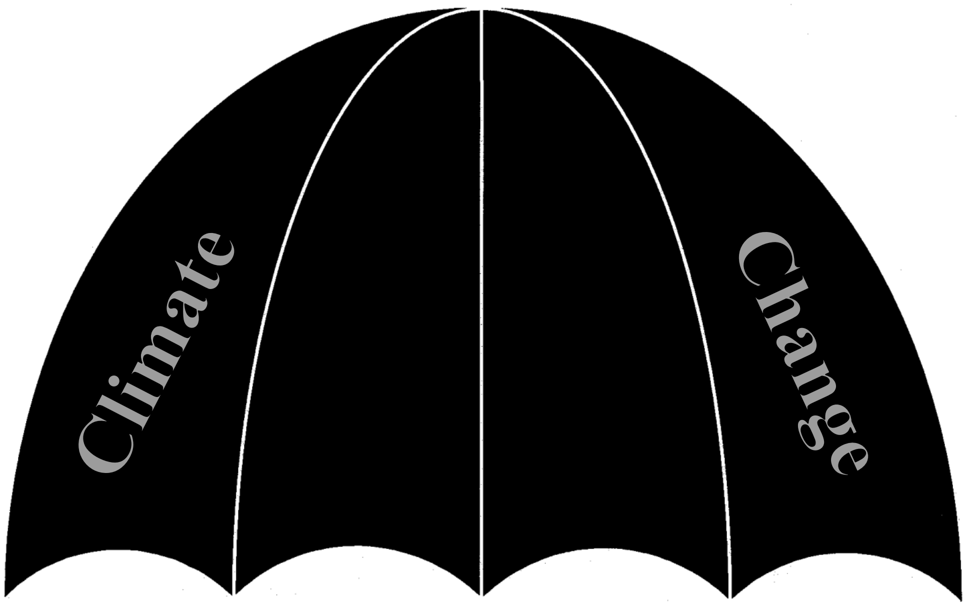
"When either the spatial extent or, more commonly, the frequency of a given severe disturbance is at or beyond the extreme end of its historical range of variability, the regenerative capacity of an ecosystem may be overwhelmed (Paine et al. 1998)" (Drever et al. 2006). The potential for greater wildfire impact could promote establishment of shaded fuelbreaks (Agee et al. 2000) to help protect threatened ecosystems, and it could foster greater application of defensive actions (such as low-density thinning) to protect forests from increasingly common bark-beetle activity, particularly during droughts.

As wildfire, bark-beetles, and other processes increase to a point where they affect a greater proportion of the future landscape, managers could try to reduce landscape synchrony by promoting diversity in tree age classes, genotypes, and species assemblages (and, another reason to consider shaded fuelbreaks is not just for protection of values-at-risk, but to help prepare landscapes for wildfire in order to capitalize on fire's ecological benefits).

A 'managed wildfire' strategy, which would undoubtedly rely on a shaded fuelbreak system in conjunction with place-based treatments protecting values-at-risk, would offer many potential benefits in a warmer and drier future: "Fire suppression can be relaxed in remote areas, letting unplanned ignitions burn under the observation of fire crews, even for weeks or months in summer, so that alternating smolder-and-run fire behavior develops in phase with a broad range of weather conditions, promoting heterogeneous burn severities over the landscape as reported in numerous pre-suppression accounts" (Goforth and Minnich 2013).

Mitigation and adaptation strategies other than managed wildfire exist for upland forest biophysical environments; 13 general recommendations are provided in figure 67 by grouping them into three primary ecosystem components influencing upland forests – composition, structure, and stand density – and the components are arrayed under an overarching umbrella of climate change. The reason that climate change is shown as an overarching influence is that it is expected to function in just this way – all three of the ecosystem components will be affected by future climate change, and they will likely be affected in a similar manner and to a similar extent.

Obviously, changes in the extent, severity, or frequency of a disturbance process to the degree portrayed for wildfire in figure 66 is likely to disrupt the regenerative capacity of a tree species, perhaps leading ultimately to species extirpations. Such changes would also cause substantial ripple effects across many biological webs (Perry et al. 2008). If climate change, along with associated changes in disturbance, precludes us from sustaining our current level of ecosystem components (composition, structure, density), then how can we expect to sustain the ecosystem goods and services relying on these components (Ford et al. 2011)?



COMPOSITION	STRUCTURE	STAND DENSITY
Promote drought resistance	**Reduce ladder fuels**	**Maintain low tree density levels**
Promote fire resistance	**Promote diverse age classes**	**Favor historical density (as buffer)**
Favor mixed species/genotypes	**Favor early-seral stages**	**Manage to LLMZ level or below**
Plant with variable species & spacings	**Favor heterogeneous structures**	**Use wide shelter-wood spacings**
Expect some sites to transition to nonforest		

Figure 67 – Mitigation and adaptation options as related to predicted climate change (adapted from Joyce et al. (2008, 2009) and West et al. (2009)). This figure provides a few ideas for addressing climate change by incorporating short-term mitigation and long-term adaptation approaches into an active management strategy. Note that these suggestions are directed primarily at addressing predicted environmental conditions (temperature and precipitation) for the interior Pacific Northwest (Littell et al. 2010). Climate change is shown as an 'umbrella' because it is expected to have overarching effects on ecosystem components (species composition, forest structure, and stand density), and on disturbance regimes influencing these components. This means the options shown here may be intuitively attractive for addressing changes in the components caused by a warmer and dryer future, but they may not necessarily represent the 'best' approaches for addressing changes to disturbance regimes. An example: promoting diverse age classes and heterogeneous structures is designed to help maintain options for an uncertain future (i.e., don't put all the eggs in one basket), but this approach is not well aligned with a future where wildfire occurrence may be two or three or perhaps even six times greater than at present (Littell et al. 2010, and see fig. 66). [Note that LLMZ refers to the lower limit of the management zone – see figure 77 later in this paper for more information about the LLMZ.]

6. ACTIVE FOREST MANAGEMENT CONSIDERATIONS

SECTION CONTENTS

Section Summary

- Nature provides two patterns for managers to follow – succession and disturbance. They are opposing yet complementary forces; succession includes normal growth and development of a forest, but disturbance can interrupt succession at any time. Active management is used to impose disturbance (silviculture, including prescribed fire) and modify forest development.
- Active management can be used to address altered disturbance regimes and improve forest health. Restoring healthy forests should not attempt to recreate previous historical conditions, but it should focus instead on protecting and nurturing an ecosystem's capacity for change (its innate processes and functions). This concept can help us evaluate whether fires, insect outbreaks, and other disturbance processes are 'healthy' or not.
- Range of variation (RV) is an analytical technique for characterizing ecosystem conditions through time. An RV section provides moist-forest RV information for vegetation cover types, forest structural stages, stand density classes, insect and disease susceptibility, and canopy fuel loading.
- Healthy, resilient, and productive forests rely on healthy, resilient, and productive soils. Effects of forest management on soil productivity are described, including implications of wildfire and prescribed fire on soil nutrients. Nitrogen, a nutrient readily volatized by a fire's heat, occurs in high amounts in organic matter of moist-forest soils, so that nitrogen source is vulnerable to loss from wildfire. This section provides 14 nutrient management considerations for moist-forest soils.
- An ecological approach to active management is adaptive, and it appraises forest conditions in a context of proper function. Group selection cutting emulates gap-phase processes such as pathogenic stem decay and root diseases; regeneration cutting mimics patch-creating processes like mixed-severity wildfire, mountain pine beetle in lodgepole pine, and windstorms in spruce-fir forest.
- Blue Mountains moist forest is resilient, and much of this resilience stems from one fact – moist forests occupy moderate or intermediate ecological settings (as described in figure 9, section 4). Their mesic environmental conditions result in high levels of species diversity. High tree species diversity provides valuable ecosystem redundancy because any particular disturbance process is unlikely to affect all species with similar levels of severity (tree mortality).
- As is true for most natural resource issues, scale is an important consideration for active management of moist forests. A functionally important scale for moist forest is the stand or patch level because moist-forest processes or functions typically affect an entire patch or stand. However, results of process and function (composition, structure, and density) should be interpreted in a landscape context to understand their ecological significance.
- Section 6 concludes with perspectives about disturbance emulation; a future role for timber management in moist forest; active management for wildfire resilience, post-fire recovery, young stand development, and restoration of limited vegetation components; adaptive management in support of moist-forest project planning; and consideration of best available science when planning moist-forest projects.

6.1 Disturbance, Succession, And Silviculture

Goal: apply the right practices, in the right places, at the right times, and for the right reasons.

Broadly speaking, nature provides two patterns for silviculturists to follow. The first is called succession – the normal growth and development of an existing forest. The second is known as disturbance – the partial or complete destruction of an existing forest through natural events. Ecologically, succession and disturbance determine the development of a forest stand. Silviculture, however, does not precisely mimic nature because nature's ways are far more random and sometimes more disruptive than society finds acceptable.[4]

Succession and disturbance are opposing yet complementary forces. When undergoing succession, a community progresses from early rapid changes to later stages featuring slow, almost imperceptible changes. As described in sections 4 and 5 of this white paper, some or all of the vegetation is killed during disturbance, setting succession back to an earlier stage (see figures 10, 11, and 16).

Secondary forest succession is what we think of as the normal growth and development of a forest stand. It begins after disturbance when new trees start to grow, and it continues through four distinct stages that if unaffected later by another severe disturbance process, extends ultimately to an old forest (see table 5 on page 31).

Succession can be explained in terms of an idealized time continuum. Assume that at a starting point called year zero, fire, windstorm, or another major disturbance event kills most of the vegetation on a site. Secondary forest succession begins with this deforested condition at year zero, immediately following the disturbance. From this point on, and assuming no more stand-killing disturbances, the stand will pass through four distinct stages of development (table 5).

Disturbance is the ecological counterpoint to succession. Plant communities develop during succession by progressing from immature to mature stages, but disturbance can interrupt succession at any time. Some disturbances are severe enough to set a plant community back to the beginning of the stand initiation stage. Others are so minor that only one or a few trees are affected, and these small changes can actually help advance forest succession during the stem exclusion, understory reinitiation, and old-forest stages (table 5).

When judged using ecological time scales, the amount of time that disturbances directly affect a stand is infinitesimal. But these changes are extremely important ecologically because they allow new generations of vegetation to get established and develop. Disturbances are of keen interest to ecologists and foresters because they establish the ecological conditions under which new plant communities are created.

---

[4] The introductory portion of section 6 (consisting of about 2½ pages) was adapted from Guldin (1996).

Disturbances vary in at least three dimensions. Frequency is the rate at which disturbances recur over time. Frequent disturbances occur every few years, whereas infrequent disturbances occur once every few centuries. Predictability describes the regularity of a disturbance, such as timing of flood events on a river or stream (cottonwood and willow regeneration is tightly synchronized with flood timing, not with flood frequency). Magnitude is the duration of a disturbance event, and it varies from minutes (such as wildfire) to years (such as drought). Magnitude is often expressed using two related concepts called intensity and severity (see table 3, p. 20).

In the Pacific Northwest, a severe disturbance that sets succession back to the stand initiation stage is not uncommon. These large events, such as the volcanic eruption of Mt. St. Helens in 1980 or the Tower wildfire in 1996, are spectacular but only tend to occur every half century or more. Partial disturbance in which some of a stand is killed, but much of the overstory and mid-story survives, is also common. Stands resulting from partial disturbance have more diversity in structure and plant species composition than stands created by a complete disturbance (see fig. 9, p. 27). And wildlife species also show a strong response to disturbance: "species show a range of responses to disturbance. Some require open, early-successional conditions, some require mature communities, and some can survive in both early and late-successional habitats. Increases or decreases in disturbance levels relative to historical conditions may threaten biological diversity" (White et al. 1999).

Using Silviculture To Impose Disturbance

Foresters use silviculture to impose disturbance and modify successional development in a stand. Silvicultural treatments are used to remove some of the trees so those remaining can develop better. The degree to which these prescribed actions imitate nature depends on how they are implemented. Regeneration cutting imitates disturbance; thinnings and other intermediate treatments imitate succession.

A forester's first alternative is that of no treatment. But other alternatives involve removing increasing proportions of the vegetation. The choice of alternatives must be consistent with the ecology of species comprising a stand, existing stand conditions, and future conditions desired by the Forest Plan. Even-aged regeneration cutting imitates disturbance affecting an entire stand; uneven-aged cutting mimics disturbance affecting portions of a stand. Figure 14 (page 41) shows four silvicultural cutting methods and how they retain varying numbers of residual trees.

The stages of stand development, along with gradients of disturbance magnitude, provide an ecological basis for silviculture. The early stages of stand development set the stage for even-aged silviculture. By imposing disturbances severe enough to promote regeneration, the forester can encourage the development of intolerant and mid-tolerant species distributed uniformly across a stand. In figure 13 (page 40), these treatments are called stand initiating, as compared to stand-maintaining treatments designed to mimic the late stages of stand development.

The later stages of succession, primarily the understory reinitiation and old forest stages, provide the ecological basis for uneven-aged silviculture. A silvicultural prescription that imitates scattered natural mortality in the upper crown classes can promote development of reproduction continuously over time. The goal of uneven-aged silviculture is to stabilize stand structure and biomass over long periods, thus emulating the old forest stage. But other ecological attributes of the old forest stage (such as snags and down wood) are also provided by uneven-aged silviculture.

The most intense, small-scale disturbances do not affect an entire stand, but can create openings within a stand. Natural examples include a localized insect infestation such as western pine beetle, a small area of windthrow, or an area of torching within a larger surface fire. Such a disturbance creates a gap in the tree canopy; reproduction becomes established and develops within this opening. Ecological conditions within the gap are affected by bordering trees, depending on opening size and shape. Foresters often use group selection cutting to mimic these conditions.

The least intense, small-scale disturbance in a stand is a single tree falling, or dying while standing, in the woods. Causes of such individual tree mortality include diseases, insects, lightning, windthrow, or some combination of these factors. If the dying tree had a large crown, shade-tolerant reproduction will become established in the gap created in the canopy. In the smallest gaps, the opening may close before reproduction can grow into the main canopy, and the reproduction may then persist without further growth (stagnate), or it might even become suppressed and die. Foresters often use individual-tree selection cutting to mimic these conditions.

Ecologically, what these succession and silvicultural approaches have in common is that they all involve some sort of renewal cycle, generally initiated by a disturbance event. The disturbance event can be a wildfire, a 'pest' infestation of insects or disease, blowdown associated with a windstorm, or it could be a human-caused fire or a patch of forest harvested to meet socioeconomic needs (Gunderson et al. 2010). If a disturbance is small (e.g., it has limited spatial extent and intensity or severity), then it can function as a 'release' agent contributing to system renewal by facilitating a reorganization phase in which the 'memory' of an ecosystem (legacies) is transferred from one generation to the next (perhaps a good example of ecosystem memory is serotinous cones in lodgepole pine forest).

But an interesting and informative aspect often overlooked is the 'social memory' represented by traditional ecological knowledge, particularly as reflected in long cultural traditions of using prescribed fire and other resource management practices to maintain and enhance huckleberries, root plants (camas, bitterroot, cous, etc.), and other first foods (Turner 1999) (fig. 51, and section 5.11, describe first foods considerations in more detail). And while the Native American aspects of cultural tradition are often emphasized (for good reason), we shouldn't forget the cultural customs, values, and traditions introduced by early Euro-American emigrants to the Blue Mountains area (as described in sources such as Tucker 1940).

6.2 Forest Health

The mission of the USDA Forest Service is to sustain the health, diversity, and productivity of the Nation's forests and grasslands to meet the needs of present and future generations.

Historical Context For Forest Health Discussion

Over the last 30 years, Blue Mountain forests experienced increasing impacts from wildfire, insects and diseases. Scientific assessments documented the high damage levels and speculated about their underlying causes (Caraher et al. 1992, Gast et al. 1991, Hessburg et al. 1999, Lehmkuhl et al. 1994, Quigley and Arbelbide 1997, Quigley et al. 1996, Shlisky 1994). Partly in response to the scientific assessments, the Blue Mountains were portrayed in numerous newspaper and magazine articles as having perhaps the worst forest health in the western United States (Durbin 1992; East Oregonian 1992; Gray and Clark 1992; Kenworthy 1992; Lucas 1992, 1993; McLean 1992; Petersen 1992; Phillips 1995; Richards 1992).

As the Blue Mountains gained notoriety as a prime example of poor forest health, much of this perception reflected the previous 'balance-of-nature' paradigm where large forest fires and landscape-scale insect outbreaks were viewed as indicators of impaired ecosystem function. While some aspects of this perspective have merit (compositional and structural changes have most certainly moved contemporary ecosystems further away from their historical reference conditions), the current paradigm acknowledges that insects, fires, and other disturbance processes also function as ecosystem 'tools' for responding to altered conditions.

Altered disturbance regimes often result in forest health issues such as stand-initiating wildfires or insect outbreaks, but the conditions contributing to these changes take decades or centuries to develop. Plant succession in combination with human influence and extremes in weather are the primary initiators of forest health issues; insect outbreaks and disease epidemics may be little more than symptoms of an underlying problem (Sloan 1998, Steele 1994). Forest ecosystems will adjust to altered disturbance regimes with the only tools available – insects, diseases, wildfire, and to a limited extent, microbial decomposition (Harvey 1994; also see fig. 7).

The Blue Mountains forest health crisis of the early 1990s raised several important questions. Do insect outbreaks and disease epidemics indicate that ecosystems are unhealthy? And what do large, landscape-scale fires indicate in an ecological context? Since ecosystems are constantly changing, we need to evaluate their health in a similar manner. Resilient forests not only tolerate periodic disturbance, they may depend on it for rejuvenation and renewal (Johnson et al. 1994). Significant changes in the magnitude (extent), intensity, or pattern of disturbance, however, may be indicators of impaired ecosystem integrity (Sampson and Adams 1994).

Many disturbance processes have been characterized as catastrophic (Kauffman 2004) – a recent example was an outbreak of western spruce budworm in the Blue Mountains between 1980 and 1992. Such characterizations are often inaccurate, since true catastrophes are rare and seldom repeated with regularity (Rogers 1996). Catastrophe, like beauty, may be in the eye of the beholder. Although the 1980-92 budworm outbreak might qualify as an uncharacteristic disturbance event for areas such as the North Fork John Day River basin (Shlisky 1994), where host-type forests experienced atypical impacts, it was not a catastrophe because species extirpations and other enduring ecological changes did not result (Powell 1994).

Defining And Interpreting Forest Health

Forest health has been defined in many different ways (see one definition in glossary from Helms 1998). A popular definition is: "forest health is a condition of forest ecosystems that sustains their complexity while providing for human needs" (O'Laughlin et al. 1994). Some folks are uncomfortable with this definition because it contains a human component – the "providing for human needs" aspect. But this portion of the definition is valuable because it provides context for evaluating forest health, i.e., are changes in forest ecosystems affecting society's capability to achieve the objectives established for a landscape? Inherent in this concept is the realization that societal objectives vary from one landscape to another, so what qualifies as a healthy landscape could also vary from one area to another.

Another potentially bothersome aspect of forest health is the term 'health.' Although forest health helps with communication because people can easily draw an analogy to human health, it is widely recognized that human health is not a valid metaphor for ecosystem health (Wicklum and Davies 1995). In humans, determining health is relatively simple because vital signs (blood pressure, body temperature, heart rate, etc.) are well known, and they vary only slightly around a predictable value determined by readings taken from many individuals. However, defining one optimal condition for an ecosystem is not possible and, although structural components of an ecosystem are interconnected, the ecosystem itself does not rely on individual components to function as a whole. For example, if all the trees in an ecosystem were eliminated, the original ecosystem type would no longer exist but an ecosystem would still continue to function – it would just be dominated by something other than trees (Wicklum and Davies 1995).

It is clear from the forest health definition given above that healthy forests can contain insects, pathogens, parasites, and other tree-killing agents, but the dead trees they create should occur at levels approximating historical ranges, and not inhibit perpetuation of sustainable ecological conditions. This means that historical information in general, and the range of variation in particular (section 6.3), can be useful tools when deciding if current conditions are 'healthy' or not.

Restoring Health And Resilience

In some portions of the Blue Mountains, landscape-scale changes have occurred so extensively that simply reintroducing native disturbance processes (wide-ranging wildfire, for example) would produce effects well outside of any historical precedent. These effects are undesirable if they move an ecosystem further away from, rather than closer to, the desired future condition (Landres et al. 1999). A common perception, for example, is that today's large fires are an ecological catastrophe because they burn vast areas (Keane et al. 2008). But just because a fire is large doesn't mean it is unnatural or outside of any historical precedent (Malamud et al. 2005, Pyne 1997). And, depending on fire behavior and associated fire effects, it may be appropriate to view large fires as fuel management treatments providing ecosystem restoration benefits (Donovan and Brown 2008, Keane et al. 2008).

In situations where current conditions deviate significantly from reference (historical) conditions, some type of active management (using commercial thinning to reduce tree biomass levels, or big-game hunting to relieve herbivory pressure) may be needed before a disturbance process could be successfully reintroduced (Aplet and Keeton 1999, Case and Kauffman 1997, Pickett and Parker 1994). And, this strategy recognizes that the long-term health of wildland ecosystems and rural economies are complementary, rather than opposing, goals (Berkes 2004).

Restoration in the context of a museum curator's work with paintings, sculpture, original manuscripts, and other priceless artifacts attempts to completely halt, or even reverse, the effects of change (Lundholm and Larson 2004). This differs from biological or ecological restoration, which is not about recreating the past except as a reference point. Biological restoration seeks not to preserve an historical artifact like a painting or sculpture, but to protect and nurture an ecosystem's inherent capacity for change (Falk 1990). Thus, ecosystem restoration focuses on function, rather than attempting to recreate an exact species assemblage or landscape patch distribution.

The concept of ecological restoration being oriented toward function (process) rather than composition or structure might be analogous to physical rehabilitation – when a person loses a limb, a physician would replace it with a prosthetic device. Because a prosthetic device is designed to restore the function of a lost limb, not to replace original flesh and bones, this medical process is typically referred to as rehabilitation rather than restoration (Choi 2007).

All of these restoration concepts are aligned with the Chief's vision: "Our goal is to sustain and restore ecosystems that can deliver all the benefits that Americans want and need. Due to changing climate, we may not be able to restore them to their original condition, but we can move them toward ecological integrity and health. The Forest Service recognizes that increasing the pace and scale of restoration and active management of the National Forests is critically needed to address these threats to the resiliency of our forests and watersheds and the health and safety of America's forest-dependent communities" (Tidwell 2012).

6.3 Range Of Variation[5]

What is 300 or 400 years in the sweep of ecological history?
Nothing but a slender thread in the tapestry of time.

The range of variation (RV) recognizes that ecosystems are complex and they experience a range of conditions across which processes are resilient and self-sustaining (fig. 68). When allowed to move beyond the limits of the range of variation, ecosystems inevitably move into a state of disequilibrium or disorganization (Egan and Howell 2001, Holling and Meffe 1996, Kaufmann et al. 1994).

The range of variation (RV) is an analytical technique to characterize inherent variation in ecosystem composition, structure, and function, reflecting recent evolutionary history and the dynamic interplay of biotic and abiotic factors (fig. 68). "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). "Because one of the goals of current forest management is to model management actions on the temporal and spatial dynamics of natural processes, knowledge about the historic variability of natural disturbances is an important prerequisite" (Wong et al. 2003). But also because of natural variation, we should acknowledge that natural resource systems are extraordinarily difficult to control with management actions (Holling and Meffe 1996).

RV is meant to reflect ecosystem properties free of major influence by Euro-Americans, thereby providing an insight into ecosystem resilience (Kaufmann et al. 1994). It helps us to understand what an ecosystem is capable of, how historical disturbance regimes functioned, and the underlying variation in ecosystem processes and functions – the patterns, connectivity, seral stages and cover types produced by ecological processes operating at a landscape scale (USDA Forest Service 1997). RV uses a range of reference conditions pertaining to the presettlement era – a time-frame defined as the mid 1800s for the Blue Mountains.

Perhaps the best yardstick for addressing ecosystem health is the range of variation – are changes caused by insects, diseases and wildfire consistent with what would be expected (the RV) for similar ecosystems and vegetative conditions? But since the composition and structure of a forest ecosystem changes as development progresses, it is important that land managers understand how plant succession influences forest development to ensure that management activities are placed on a sound ecological foundation: "manipulation of a forest ecosystem should work within the limits established by natural disturbance patterns prior to extensive human alteration of the landscape" (Hunter 1999, p. 29).

---

[5] Detailed information about the range of variation is provided by white paper F14-SO-WP-Silv-3: "Range of variation recommendations for dry, moist, and cold forests" (Powell 2019).

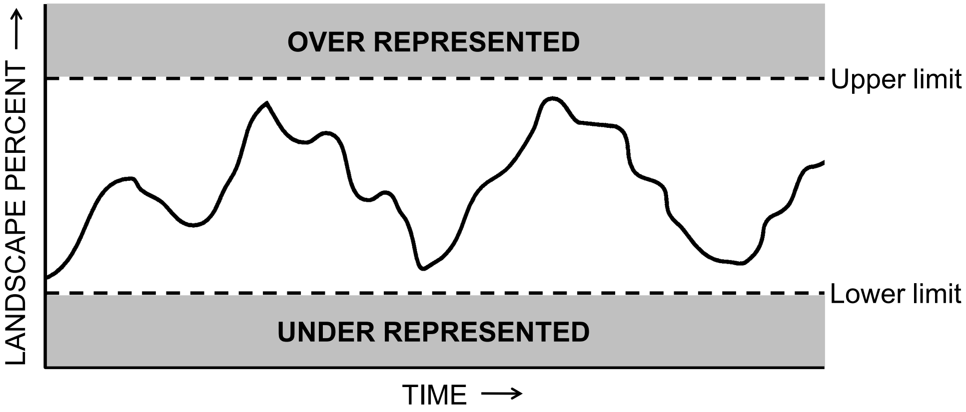


Figure 68 – The range of variation (RV) places existing amounts of vegetation composition, structure, and density in an ecological context (Aplet and Keeton 1999, Morgan et al. 1994, Swanson et al. 1994). This diagram shows the ecological trajectory of an ecosystem component (the solid line) varying through time because the phrase 'range of variation' is meant to encompass more than just the extreme values (the upper and lower limits, shown as dashed lines) (diagram modified from Morgan et al. 1994).

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

Studies examining plant succession suggest that instead of seeking an elusive or nonexistent equilibrium, we should be looking for the trajectory over which an ecosystem is most likely to proceed in the future. Rather than trying to maintain a forest in one homeostatic state, for example, we should be asking whether or not an action (or no-action) will allow it to stay within a set of bounds (the RV; fig. 68) that seem either normal or preferred (Botkin 1994, Steele 1994). Although the RV approach has recently been questioned, it is still believed to function as a useful tool to inform management practices rather than to set firm targets (Thompson et al. 2009).

RV And Active Management

As mentioned on the previous page, RV has been proposed as an appropriate way to assess forest health. Since a key premise of RV is that native species have evolved with, and are adapted to, the disturbance regime of an area, ecosystem components occurring within their historical range are believed to represent healthy situations (Aplet and Keeton 1999, Swanson et al. 1994). At a landscape scale, a moist forest might be considered healthy if the spatial and temporal patterns of compositional and structural patches are within RV for any particular biophysical environment.

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

RV concepts were included in a recent definition of forest ecosystem integrity: "a forest ecosystem has integrity if its structure and species composition, the rate of its ecological processes, and its ability to resist change in the face of disturbance or stress are within the characteristic range exhibited historically by that ecosystem" (Kimmins 1996). For example, the range of variation acknowledges that susceptibility to spruce beetle (a disturbance agent and a stressor) increases as spruce-fir forests mature, even though not all mature spruce-fir forest is predestined to experience a spruce beetle outbreak.

Much of our current standard of living depends on converting landscapes to conditions outside their range of variation to satisfy human purposes and needs (rivers dammed to provide hydroelectric power and irrigation water, fish stocking to support recreational and commercial fishing, fire suppression to protect property and natural resources, bluebunch wheatgrass steppes converted to wheat fields for food production, etc.) (DeFries et al. 2004).

When human activities alter landscapes, regardless of the impact, the alterations tend to continue as long as the majority of the population finds them acceptable. This situation suggests that a 'social range of variation' is needed – the range of ecological conditions that society finds acceptable (Reeves and Duncan 2009). But our experience also suggests that because of their inherent variation, natural systems can be difficult to 'control' with human activities, if for no other reason than ecosystem cause-and-effect relationships may be unclear or difficult to recognize.

For moist upland forests that have been substantially disturbed or are far outside their range of variation, restoration treatments are often prescribed (Morgan 2004). Realigning current conditions with predicted future conditions, rather than restoration to historical conditions, may be a preferred choice when considering expected climate change. In effect, such an approach seeks to align current moist-forest conditions with a range of expected future environments (Alfaro et al. 2010).

Although the historical range of variation concept is viewed as incompatible with future conditions affected by climate change (Millar and Woolfenden 1999), we must still acknowledge that historical information is invaluable, as current ecosystems have adapted to and survived abrupt climate change in the far past (including a very warm period 4,000-8,000 years ago). Also, it may be important to recognize that restoring forests to their presettlement condition would generally result in composition, structure, and density that is more resilient and resistant to expected future changes in climate and disturbance regimes than is represented by the vast majority of existing forest conditions for the western U.S. (Stephens et al. 2010, 2012).

Range of variation information for species composition, structural stage, tree density, and insect and disease susceptibility is provided in tables 18-21, and all of it pertains specifically to moist, upland-forest biophysical environments of the Blue Mountains ecoregion.

Table 18: RV information for vegetation cover type on the moist upland forest potential vegetation group.

Vegetation Cover Type	Range of Variation (Percent)
Grass-forb	0-5
Shrub	0-5
Western juniper	NA
Ponderosa pine	5-15
Douglas-fir	15-30
Western larch	10-30
Broadleaved trees	1-10
Lodgepole pine	25-45
Western white pine	0-5
Grand fir	15-30
Whitebark pine	NA
Subalpine fir and spruce	1-10

Sources/Notes: Derived from disturbance process modeling using the Vegetation Dynamics Development Tool (VDDT) (Powell 2012). NA is Not Applicable. Cover types, which reflect the existing vegetation composition of a polygon (Eyre 1980, Shiflet 1994), 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.

Explanatory note for species composition RV information: The information in table 18 expresses the percentage of a moist-forest landscape (preferably at least 15,000-35,000 acres in size) occupied by various vegetation cover types (e.g., ponderosa pine, grand fir, etc.). A cover type may have more than 50% of one species (e.g., ABGR); if less than 50% of a species is predominant, then the cover type is named for a species comprising the plurality of stocking (e.g., mix-ABGR). I must emphasize that the species composition information provided in table 18 does NOT pertain to the composition one might expect for any individual mixed-species, moist-forest stand. In other words, a mixed-species, moist-forest stand would not be expected to contain 5-15% ponderosa pine, 15-30% Douglas-fir, and so forth.

Table 19: RV information for forest structural stage on the moist upland forest potential vegetation group.

Forest Structural Stage	Range of Variation (Percent)
Stand initiation	20-30
Stem exclusion	20-30
Understory reinitiation	15-25
Old forest single stratum	10-20
Old forest multi strata	15-20

Sources/Notes: Based on disturbance process modeling from the Vegetation Dynamics Development Tool (VDDT) (Powell 2012). Forest structural stages are described in table 5.

Table 20: RV information for tree density on the moist upland forest potential vegetation group.

Tree Density Class (mixed composition at a quadratic mean diameter of 10")	Range of Variation (Percent)
Low (<78% CC; <103 ft^2/acre BAA; <189 tpa or sdi)	20-40
Moderate (78-86% CC; 103-154 ft^2/acre BAA; 189-283 tpa or sdi)	25-60
High (>86% CC; >154 ft^2/acre BAA; >283 tpa or sdi)	15-30

Sources/Notes: Derived from Powell (2019). Note that CC refers to canopy cover, BA to basal area per acre, tpa to trees per acre, and sdi to stand density index. All Tree Density Class values pertain to an even-aged stand structure and a mixed species composition (30% Douglas-fir, 20% western larch, 20% lodgepole pine, and 30% grand fir). The tpa and sdi values are the same because the sdi system uses a 10" quadratic mean diameter (QMD) as its reference tree size; sdi and tpa values are the same when QMD is 10 inches.

Table 21: RV information for insect and disease susceptibility on the moist upland forest potential vegetation group.

Insect and Disease Agents[1]	Range of Variation (Percent)
Defoliating insects	
Low susceptibility	5-10
Moderate susceptibility	20-30
High susceptibility	35-80
Douglas-fir beetle	
Low susceptibility	30-60
Moderate susceptibility	20-40
High susceptibility	10-30
Fir engraver	
Low susceptibility	30-70
Moderate susceptibility	10-20
High susceptibility	20-40
Spruce beetle	
Low susceptibility	50-95
Moderate susceptibility	10-25
High susceptibility	0-10

Insect and Disease Agents[1]	Range of Variation (Percent)
Bark beetles in ponderosa pine	
Low susceptibility	30-65
Moderate susceptibility	15-30
High susceptibility	15-35
Mountain pine beetle in lodgepole pine	
Low susceptibility	30-60
Moderate susceptibility	25-40
High susceptibility	5-30
Douglas-fir dwarf mistletoe	
Low susceptibility	30-65
Moderate susceptibility	20-45
High susceptibility	10-20
Western larch dwarf mistletoe	
Low susceptibility	5-20
Moderate susceptibility	15-40
High susceptibility	40-70
Root diseases	
Low susceptibility	5-25
Moderate susceptibility	20-40
High susceptibility	35-65

Sources/Notes: Derived from Schmitt and Powell (2012). Queries for calculating susceptibility ratings for forest polygons are available from Schmitt and Powell (2005). Conceptually, forest components (composition, structure, density; tables 18-20) occurring within their ranges of variation would produce insect and disease susceptibility occurring within the ranges shown in this table.

[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.

6.4 Forest Management And Soil Quality

In the timber production era of the 1950s to 1980s, “rapid industrialization and inexpensive energy created the illusion that forests could be indefinitely modified to boost timber production. Consequently, silviculture adopted methods more akin to agricultural crop production, such as preference for monocultures, genetic breeding, intensive crop manipulation, and the use of fertilizers and pesticides” (Kuuluvainen and Grenfell 2012, p. 1186; also see Puettmann et al. 2009). Over the last 30 years, significant advances have been made in terms of reducing the negative impacts of timber production activities on site productivity, nutrient availability and cycling, and other ecosystem goods and services.

Major forest nutrients include nitrogen, potassium, phosphorus, sulfur, calcium, and boron. Many forest sites are nitrogen deficient (Mandzak and Moore 1994). For Blue Mountain forests, sulfur is often limiting too – adding nitrogen without addressing a sulfur deficiency will commonly result in little or no growth response and may actually end up killing some trees. Boron can become deficient on sites exposed to intense mechanical site preparation treatments, particularly regarding how the treatment affects residual foliage (leaf-on slash) because foliage is high in boron.

Foliage and branches have a small proportion of a tree's biomass (app. 15%), but they contain the highest concentration of nutrients. Stemwood has a high amount of a tree's biomass (app. 60%), but relatively little of its nutrients. Woody roots provide a tree's anchorage, but have low nutrient concentrations. The non-woody portion of a root system has high nutrient levels. Roots, in total, have about 25% of a tree's biomass. Table 22 shows that timber harvest leaves at least 77 percent of the total nitrogen on site, and usually much more depending on how well the forest floor is conserved. Does this mean we don't have to worry about the possible effects of mechanized harvest or site preparation treatments on site nutrients? Not really, because only a portion of a site's soil-pool nutrients are available at any particular time.

Table 22: Nutrient concentrations, by component, for true fir and pine stands.

	MOIST FIR STAND		**DRY PINE STAND**	
Component	**Nitrogen**	**Phosphorus**	**Nitrogen**	**Phosphorus**
Trees: Aboveground	3%	1%	8%	1%
Trees: Roots	< 1%	< 1%	1%	< 1%
Understory	< 1%	< 1%	< 1%	< 1%
Forest Floor	13%	3%	14%	2%
Soil to 1-meter depth	83%	96%	77%	96%

Source: Powers (1989, page 9).

Nutrients that are unavailable for tree growth obviously count toward the total nutrient capital of a site, but they are basically unaffected by management practices (unless something happens to cause serious erosion and off-site soil loss). A significant portion of the nutrients found in soil to a 1-meter depth is unavailable for tree growth in the short term. In the long term, weathering (decomposition) processes will eventually make these nutrients available for plant growth.

When conifer foliage and small branches are shed, they fall to the forest floor and accumulate with other organic material (detritus). This material has been classified into an O1 layer (litter) where the original plant or animal structures are still recognizable to the human eye, and a deeper O2 layer in which the organic material has been reduced to an amorphous form (duff). Invertebrates, small mammals, and other organisms help speed the decomposition process by converting detritus into smaller pieces, which increases its surface area. Nitrogen and other nutrients are released in small amounts by leaching processes during this phase of decomposition. After reaching the lower O2 and humus layers, organic material is worked by springtails, arthropods, and a wide variety of soil organisms. Nitrogen suitable for plant uptake is released in various forms after being turned over multiple times by decomposers functioning in an intricate detrital food web (Perry et al. 2008).

Fire Effects On Soil Nutrients

Fire-based slash disposal and site preparation activities can have important influences on these forest-floor nutrient cycling processes. Severe or extreme burns have been shown to cause nitrogen losses of 92 percent or more; light burns have shown no effect or slight increases in available (mineralizable) nitrogen, ranging up to an increase of 28% (Harvey et al. 1989). Sulfur is another important nutrient affected by prescribed fire and wildfire – it is highly vulnerable to volatilization losses when exposed to combustion temperatures of 375-575 °C for a period of only five minutes (Tiedemann and Anderson 1980) (according to Busse et al. (2005), maximum surface temperatures commonly reach 600 °C for dry soils affected by late-summer wildfire or autumn prescribed fire).

Potential long-term effects of fire on nutrients depend on volatilization of certain elements (nitrogen, potassium, sulfur, phosphorus), and on off-site movement of fine particulate ash by wind and water. The amounts of nutrient loss depend on: (1) temperature of the fire; (2) type of fire; (3) tree species on the site (see fig. 69); (4) composition of the undergrowth flora; and (5) climatic conditions following the fire (Prescott et al. 2000b).

High-intensity prescribed fire, particularly if used when soils are moist, could also be expected to cause detrimental impacts on soil fauna. The density of soil fauna in an undisturbed forest soil is usually quite large – in a study involving the organic and upper soil layers under a ponderosa pine forest near Grass Valley, California, it was found there was a population density of about 200,000 arthropods per square meter of forest floor volume. About 150 species were represented, dominated by mites and springtails (Mitchell and Martin 1980). Post-fire changes in pH seem to have an important influence on the abundance and vigor of microbial communities (Reaves et al. 1984, 1990; Switzer et al. 2012). Neary et al. (1999) provide a useful synthesis of fire effects on below-ground sustainability, including fire's influence on a soil's physical, chemical, and biological functions and processes.

Species Composition Effects On Soil Nutrients

Characteristics of forest floor organic material and nutrient storage vary with overstory and understory species composition (Harvey et al. 1999, Miesel et al. 2008) (fig. 69), along with temperature and moisture conditions contributing to forest floor decomposition rates. The extent to which the tree overstory acts as a nutrient sink depends on stand conditions and how they evolve during forest succession (species composition, structural stage, stand density, etc.). One example is nutrient decomposition rates associated with the quaking aspen seral-stage when compared with lodgepole pine and other conifers: aspen litter decayed significantly faster than conifer litter, at least partly in response to a much higher diversity and biomass of earthworms for the aspen-dominated areas (fig. 70) (González et al. 2003).

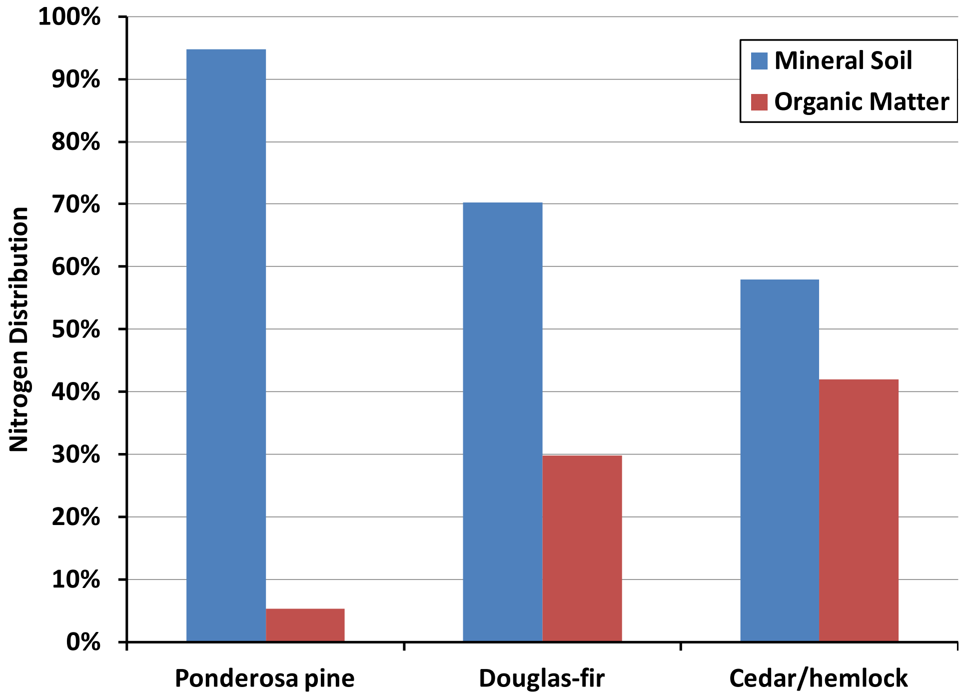


Figure 69 – Nitrogen distribution in mineral soil versus organic matter layers (the O horizon) of ponderosa pine, Douglas-fir, and cedar-hemlock forests in Montana (data derived from Jurgensen et al. 1997, p. 237). Note that the cedar/hemlock forest type is equivalent ecologically to moist grand fir forest in the northern Blue Mountains. The organic matter component consists of two stocks: woody residue greater than 3 inches in diameter, and the forest floor (litter and duff). Mineral soil analyses included nitrogen content to a depth of 1 foot. This chart shows a progressive change in nitrogen storage as forest type changes – there is a gradual shift in the location of nitrogen reserves from primarily deep in the mineral soil for ponderosa pine, to much greater amounts in the surface organic horizons for Douglas-fir and cedar/hemlock forests. The nutrient progression shown here demonstrates that dry pine forests tend to have critical nutrients like nitrogen better protected from fire and other surface disturbance processes than is true for the Douglas-fir or cedar/hemlock forest types (Harvey et al. 1999).

Early-successional or shade-intolerant species have a significantly greater proportion of roots occurring deep when compared with late-successional or shade-tolerant species. Differences in vertical root distribution are presumed to be related to inherent genetic variation, and how it affects the capability of species to exploit nutrients and water following disturbance. Early-successional species adapt better to sites with limited water and nutrients because of their ability to exploit larger volumes of soil. Late- successional or shallow-rooted species are better adapted to sites where resources are concentrated near the soil surface (Gale and Grigal 1987).

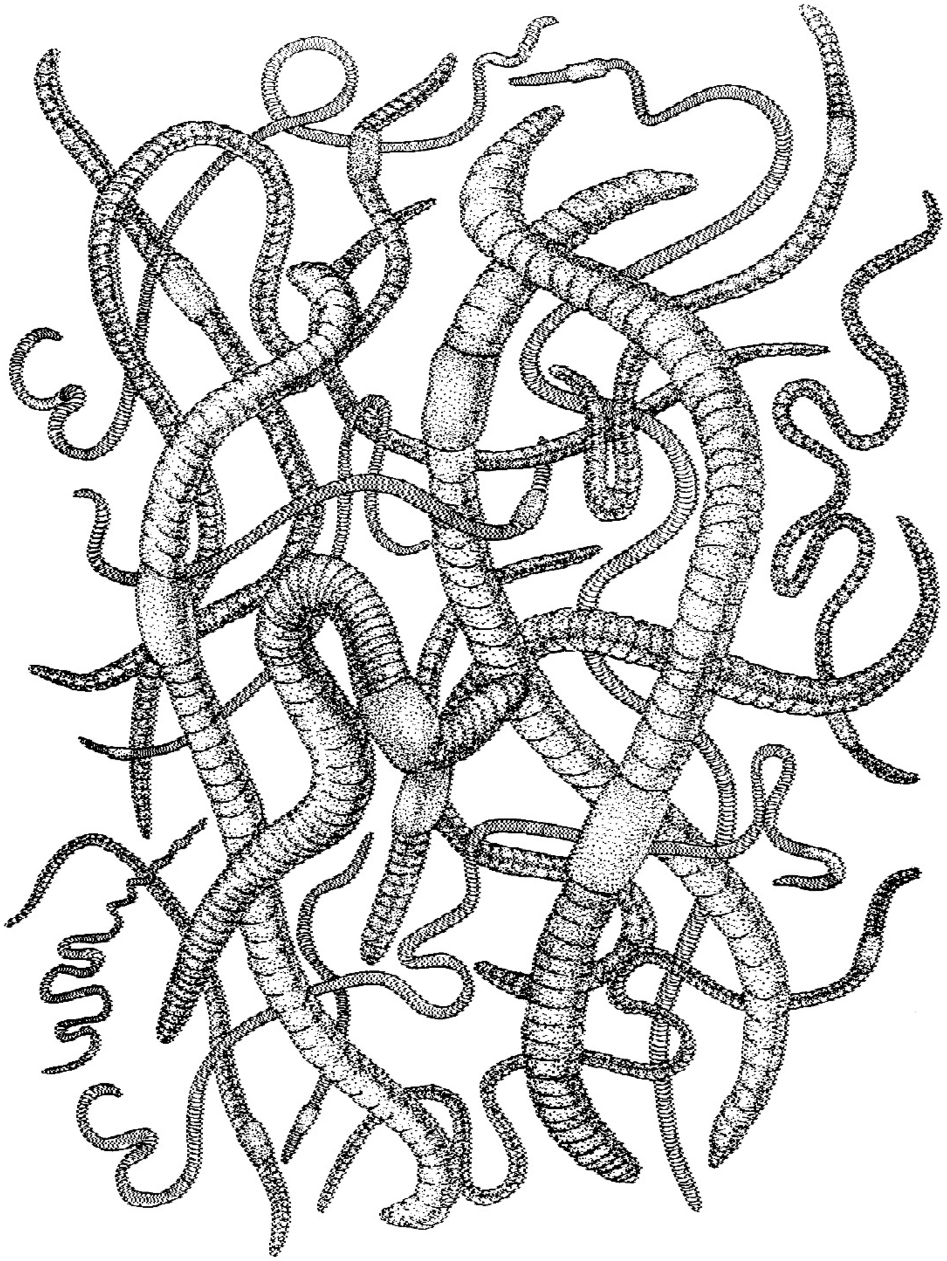


Figure 70 – Earthworms are an important component of the soil fauna, being especially influential on soil structure and organic matter decomposition (Paul 2007). Among other ecosystem services provided by earthworms, they function as an important decomposer organism (along with a tremendous variety of soil-based micro- and macro-arthropods). In a subalpine forest study in Colorado, for example, a much higher density and biomass of earthworms was associated with decomposing aspen litter than with decomposing lodgepole pine litter, and the amount of litter remaining at the end of the study, regardless of tree species, was related significantly to earthworm biomass (González et al. 2003).

Silvicultural treatments, wildfire, and other disturbance processes affecting species composition indirectly influence the nutrient status of moist forests (fig. 69), as do any of the active management practices designed to direct forest growth and development (thinning and harvest, slash disposal, prescribed fire, forest fertilization, etc.). Some of these nutrient influences involve the potential effects of disturbance processes, primarily wild and prescribed fire, on charcoal production.

The porous nature of charcoal provides physical benefits to the soil (e.g., improved water-holding capacity and reduced soil bulk density), while also serving as a substrate for microbial and mycorrhizal activity (DeLuca and Aplet 2008, Kolb et al. 2009). After being produced, most charcoal remains at shallow depths in the soil (70% or more remains at a depth of 4 inches or less; DeLuca and Aplet 2008).

Although the mean residence time for soil-based charcoal can exceed 8,000 years (DeLuca and Aplet 2008), proferring potential carbon sequestration advantages, research in boreal forests suggests it is possible for charcoal to become deactivated with time and no longer support large amounts of microbial biomass, influencing nutrient cycling and energy flows. Also, reduced microbial activity can allow phenolic compounds to accumulate because they are immobilized or degraded by microbial processes, and high levels of phenolics have been shown to suppress tree regeneration and seedling growth by being phytotoxic substances (Zackrisson et al. 1996).

Note, however, that these microbial relationships are complex – results similar to those of Zackrisson et al. (1996) for boreal ecosystems with high levels of ericaceous shrubs were not observed for temperate forest ecosystems (Kolb et al. 2009), although the temperate-forest study examined short-term effects of charcoal additions rather than examining a long-term successional sequence as was done for the boreal-forest study.

Soil Compaction Considerations

Soil compaction has been associated with timber harvest, but it can also occur in conjunction with mechanical site preparation treatments. Machine piling, which was historically used with moist-forest sites (see fig. 61), is one management activity with high potential for compaction. Compaction affects the aeration and infiltration capacity of a soil, which in turn affects ectomycorrhizal abundance and activity (Amaranthus and Perry 1994; Harvey et al. 1981, 1986, 1987). (Ectomycorrhizae are fungi establishing on a tree's root system and facilitating its uptake of both water and nutrients, and the root system supplies carbohydrates to the fungi.) Without ectomycorrhizae, tree growth usually suffers (Harvey et al. 1986).

Since compaction can be persistent without some sort of remediation, compaction could influence soil productivity for decades (Geist et al. 1989). When compaction was examined for a silt loam soil (such as an ash- or loess-dominated soil) and a cut-to-length harvest operation for the interior Pacific Northwest, it was found that "the soil that was driest during machine traffic (low moisture treatment) had the lowest penetration resistance. Slash was important for protecting the soil against compaction in the medium and high soil moisture treatments. Penetration resistance did not significantly increase after the second pass of a fully-loaded forwarder (31,752 kg) at any moisture content or slash level. Managing felling operations to take advantage of dry soil conditions or using slash when soils are moist may help reduce ruts and avoid long-term compaction impacts on this soil type" (Han et al. 2006).

Note that concerns about soil compaction for fine-textured, loamy soils are particularly germane to moist mixed-conifer, lodgepole pine, and subalpine fir forests because they are strongly associated with ash- or loess-dominated soils in the Blue Mountains (table 23), and both of these soil groups feature fine, loamy textures.

The forest floor's ecological importance does not mean it cannot be manipulated, in a careful and prudent fashion, to achieve resource objectives for natural regeneration (Box 6), insect or disease resistance, and nutrient cycling (Anderson 1988, Burger 2009). "A major objective of thinnings for host-dominated high-risk stands [for western spruce budworm] on cool sites is to increase the temperatures of the forest floor and topsoil and thus improve nutrient cycling rates and nutrient availability" (Stoszek and Mika 1984, p. 147).

Nutrient Conservation

Man—despite his artistic pretensions, his sophistication, and his many accomplishments—owes his existence to a six-inch layer of topsoil and the fact that it rains *(Author unknown)*

As climate warms and dries in the future, and if thinning is used to improve crown-fire and drought resistance for moist-forest sites, then the resulting open forest structure would be expected to increase microbial decomposition rates (this outcome would occur until drying reaches a level where soil moisture becomes limiting for microbial decomposition). As described above in the quote by Karl Stoszek and Peter Mika, higher forest-floor temperatures could be beneficial, up to a point, by increasing nutrient cycling and availability, thus improving stand resistance to spruce budworm defoliation (Stoszek and Mika 1984).

Research found that thinning can affect nutrient distribution within a forest ecosystem: essentially, more nutrients are held in the trees, forest floor, and woody debris components of unthinned stands than in the soil pool, whereas the opposite situation is true for thinned stands – more nutrients are stored in the soil. This means that thinning decreases the amount of nitrogen and other nutrients held aboveground, and it increases nutrient availability belowground (Garrison and Moore 2002).

Basically, thinning reallocates a stand's aboveground biomass to fewer stems with less total canopy, which results in slightly less nutrient utilization, but thinning is also used to preferentially modify a stand's species composition and, as shown in figure 69, the tree species comprising an ecosystem have a strong influence on the type of litter being produced and how it is processed (cycled and stored).

Table 23. Relationship between forest type and soil parent material.

	Tephra (Ash) or Loess As a Surficial Layer	Residual Parent Material (Basalt Primarily)
↓ INCREASING ELEVATION		**PONDEROSA PINE**
		DRY MIXED CONIFER
	MOIST MIXED CONIFER	
	LODGEPOLE PINE	
	SUBALPINE FIR	

Sources/Notes: Derived primarily from Geist and Strickler (1978). This table demonstrates that forest types in the Blue Mountains are related strongly to soil parent material – ponderosa pine and dry mixed-conifer forest types are most commonly associated with residual parent materials typically derived from the Columbia River basalt geology group featuring flood-type basalts. The moist mixed-conifer, lodgepole pine, and subalpine fir forest types, however, are almost always associated with soils containing a relatively deep surficial deposit consisting of volcanic tephra (Mount Mazama or Glacier Peak ash, primarily) or loess derived mostly from glacial materials transported initially by glacial-outburst (Lake Missoula) floods (Allen et al. 2002) and then subsequently by wind.

The moist forests forming the basis for this white paper are analogous to the moist mixed-conifer type shown in this table, and they almost always occur on ash- or loess-cap soils. This species distribution pattern shown above makes sense when considering the life-history traits of the trees involved – ash or loess surface deposits have high water storage capacity, and they are “capable of yielding comparatively large proportions of this water to plants within low soil moisture stress limits (0.1 to 1.0 bar)” (Geist and Strickler 1978). Grand fir, lodgepole pine, Engelmann spruce, and subalpine fir have high soil moisture requirements, relatively low drought tolerance, and comparatively high amounts of transpiring foliage (either on a per-tree basis (true firs, spruce), or on a stand basis such as extremely dense lodgepole pine forest). As described in figure 69, tree species exert a strong influence on the nutrient status of soils on which they grow, and ash or loess caps are no different – “volcanic ash soils supporting a spruce-fir [Engelmann spruce-subalpine fir] overstory had higher amounts of organic matter, higher total nitrogen, and lower cations than ash soils supporting other species” (Geist and Strickler 1978, p. iii).

And as described throughout this soil quality section, it is important to remember that ash- or loess-cap soils can be quite susceptible to soil compaction, and that trees have higher susceptibility to Armillaria root disease when growing on compacted soils (Curran et al. 2007).

Box 6. Organic Matter and Regeneration on Moist-Forest Sites

Deeply shaded forest floors virtually devoid of plants are common in mesic coniferous forests of the Pacific Northwest. Ecologists refer to such sites as depauperate. It is often assumed that depauperate undergrowth results from excessive shading (a lack of sunlight). A lack of regeneration in many forest stands, particularly by tolerant species such as Engelmann spruce and true firs, might not reflect the effects of shade. An interesting study came to this conclusion by examining lethal and nonlethal effects of forest soil organic horizons on seed germination for several associated conifer species (Daniel and Schmidt 1972).

In this study, organic horizon refers to an 'O horizon' in soil classification terminology, generally viewed as consisting of two parts – an O1 layer in which original animal or plant parts (leaves, twigs, etc.) are still recognizable, and an O2 layer where organic litter has been reduced to an amorphous, unrecognizable state (Daniel et al. 1979).

Investigators found that Engelmann spruce O-horizon (e.g., O horizon formed beneath a pure canopy of Engelmann spruce trees) was not only lethal to Engelmann spruce seeds, but also to seeds of subalpine fir, Douglas-fir, and lodgepole pine. Subalpine fir O-horizon was lethal to its own seeds, but only marginally harmful to other species. Douglas-fir O-horizon had a significant adverse impact on its own seeds, and was moderately harmful to seeds of other species. Lodgepole pine O-horizon was basically neutral for all seeds (Daniel and Schmidt 1972).

Which factors contributed to a lack of tree regeneration? Failure of conifer seed to germinate was attributed primarily to action of pathogenic fungi. Non-sterilized O-horizons supported huge masses of fungal hyphae after 90 days, whereas sterilized O-horizons had no fungal activity (Daniel and Schmidt 1972).

Results from this study provide one possible explanation for why a mineral-soil surface commonly has beneficial effects on seed germination and tree regeneration (Feller 1998, Lowdermilk 1925, Oswald et al. 1999, Zhong and van der Kamp 1999). It also suggests an important reason for why Engelmann spruce and subalpine fir seedlings are often found on decomposing wood, stumps or root wads, and the exposed soil of uprooted (windthrown) trees, rather than on a forest floor itself.

Ted Daniel and Josef Schmidt (1972) describe how organic matter (litter and duff in this instance) exerted an inhibitory effect on tree regeneration by serving as habitat for pathogenic fungi. If this study is put into a broader ecological context, I wonder if some of what has been reported in local ecological literature as allelopathy might actually have involved pathogenic fungi instead?

Allelopathy refers to a competitive strategy in which certain plant species produce chemical compounds interfering with germination, growth, or development of competing plants (Dunster and Dunster 1996). Fred Hall, an R-6 Forest Service ecologist, speculates 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 was able to accumulate in soil (or in O 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 Daniel and Schmidt (1972) study, I wonder if Fred's 'selective inhibitory substance' might have involved pathogenic fungi, allelopathic phytotoxins, or perhaps some combination of both?

Why might these species-based nutrient relationships be important for moist forests and their active management? If wildfire is expected to influence moist forests more often in the future in response to a warming and drying climate (see fig. 66), then nutrients held aboveground are more likely to be volatized and lost from the site than nutrients held in the soil. And because nutrient conservation has important implications for provision of ecosystem goods and services, thinning and similar activities ultimately protecting nutrients could provide climate-change mitigation benefits.

Why are we concerned about conserving nutrients? Well, an important reason is that stands growing on sites with good nutrition status contain trees with improved physiological vigor, and trees with improved vigor produce more of the resins used to repel insect and disease attacks (Langenheim 1990, Kolb et al. 1998, Mitchell et al. 1983, Nebeker et al. 1995, Phillips and Croteau 1999, Pitman et al. 1982, Safranyik et al. 1998). High-vigor trees also tend to have better foliage chemistry, at least in the context of insect and disease resistance, improving their resilience to defoliating insects because foliage chemistry affects the nutritional quality of foliage as a substrate for insect larval development (Clancy et al. 1993).

High soil quality is a fundamental prerequisite for creating and maintaining healthy forests (Aber et al. 2000, Page-Dumroese et al. 2010, Stoszek 1988a). Forest growth and productivity affect soil organic matter (its quantity and quality), which in turn affects the physical, chemical, and biological properties of forest soil (including soil food webs), and these properties interact to influence forest growth and productivity (and the 'cycle of high soil quality' is then complete) (Grigal and Vance 2000).

Many nutrient management strategies are compatible with a goal of sustaining high levels of soil quality and site productivity for moist forests:

1. Prevention is the best option; leave as many of the rich nutrients on site and undisturbed as possible (particularly foliage and the litter/duff layers).
2. Manage with a light hand (minimize equipment effects on soils, nutrients, and the residual tree stand).
3. Try to keep heavy equipment off the soil (operate heavy equipment on deep slash mats if possible). Careful equipment use is particularly important for ash caps and other fine-textured soils because they are most likely to experience productivity declines caused by soil compaction, rutting, or erosion.
4. Limit scarification treatments to keep nutrients and organic matter on site.
5. Keep broadcast burns cool (fireline intensity of 200 to 400 °F or less; 2 to 4 foot flame lengths) to conserve and sustain nutrients.
6. Be especially careful to conserve nutrients on harsh sites because nutrient cycling is slow there (this recommendation pertains to hot dry sites where ponderosa pine is climax, and to cold dry sites dominated by subalpine fir).
7. Limb trees where they fall, thereby leaving most of the tops and branches on site to replenish the forest floor and encourage nutrient cycling.

8. Manage forest floor resources (litter and duff) as carefully as possible because this is an area of rapid cycling and high nutrient availability.
9. When leaving coarse down wood, try to retain 'brown-rot' species (pines, Douglas-fir, and western larch, which support brown-rot fungi during decomposition) more often than 'white-rot' species (true firs and Engelmann spruce, which generally support white-rot fungi during decomposition) because brown-rot species last up to 450 years, creating what is known as 'brown cubical crap' or BCC (Jurgensen et al. 1979, 1989; Larsen et al. 1978). BCC provides non-symbiotic nitrogen fixation, mycorrhizal habitat, and soil moisture storage (fig. 71).
10. On sites where nutrients deteriorated following uncharacteristic fires or inappropriate management activities, consider nutrient amendments as mitigation. Fertilization provides opportunities to modify foliar chemistry and improve resistance to budworm defoliation (Clancy et al. 1993). It may also help reduce stem decay for grand firs wounded during timber harvest (Filip et al. 1992). And by changing root chemistry, nitrogen and potassium amendments apparently have beneficial effects on resistance to Armillaria root disease (Entry et al. 1992).
11. Consider forest-floor condition when prescribing silvicultural treatments. When a forest floor is present, then other management objectives will probably dictate the choice of silvicultural cutting methods. But when the forest floor is absent or very thin, then an uneven-aged cutting method (e.g., group selection or individual-tree selection) might be appropriate (Prescott et al. 2000b).
12. The challenge when practicing silviculture is to fine-tune activities so mineralization of the humus layer is promoted when nutrient uptake is important (e.g., during the regeneration phase, or when a young forest is growing rapidly), and yet discourage nutrient losses. For sites where humus has accumulated at the soil surface, silvicultural practices to reduce or activate (but not eliminate) humus are preferred, including situations where nutrient release from decomposing humus could improve budworm resistance for young stands (Stoszek 1988b).
13. Frost heaving kills or damages freshly planted tree seedlings, especially at high elevations or on moist soils just receiving a mechanical site preparation treatment. Site preparation is important when promoting early-seral tree species, but removing the organic layer to expose a broad expanse of mineral soil can also promote frost heaving. [Removing organic material immediately adjacent to planted seedlings, such as hand scalping, seldom contributes to frost heaving.]
14. Conifer seedling growth is enhanced when roots have established a symbiotic relationship with ectomycorrhizal fungi (Amaranthus and Perry 1994, Amaranthus et al. 1996). To enhance ectomycorrhizae and seedling growth, we should:
    (a) harvest in late autumn or winter;
    (b) avoid burning;
    (c) keep harvest units small, relatively circular, and well spaced; and
    (d) replant early in the following spring (Harvey et al. 1980).
    [But, ectomycorrhizal and decomposer fungi are apparently affected more by severe wildfire than by regeneration cutting (Visser and Parkinson 1999).]

Figure 71 – Actively decomposing log on a moist-forest site. Brown-rot fungi create brown cubical 'crap' by decomposing resinous heartwood tissue. When 'white-rot' tree species (true firs, etc.) decompose, their spongy, stringy wood is decayed quickly and it tends to be ephemeral in the soil system. When 'brown-rot' species (most pines, larch, Douglas-fir, etc.) decompose, their wood is decayed slowly (especially the resinous heartwood), and it tends to remain in the soil system for long periods, providing important substrates for moisture and nutrient retention, and for mycorrhizal activity (Harvey et al. 1999).

6.5 Adopting An Ecological Approach To Active Management

For every complex problem there is an answer
that is clear, simple, and wrong. *H.L. Mencken*

Management intervention should use an adaptive approach that considers the forest as a fully-functioning ecosystem. Ecological principles form the basis of this approach, which assumes that if the effects of forest management activities closely resemble those of indigenous disturbances, the risk of losing native species and altering ecosystem processes is greatly reduced (Delong and Tanner 1996, Rowe 1992). A manager's first responsibility is to ensure the integrity of the forest ecosystem by preventing damage to soil, streams, and other site components (Minckler 1974).

It is important that management action focuses on the effects of disturbance processes and the function of biological legacies, rather than attempting to directly replicate any particular disturbance agent (Carey 2007) (fig. 72). When trying to emulate the effects of a mixed-severity fire regime, for example, evidence for some amount of low-to-moderate severity fire provides ecological precedent for application of silvicultural activities like low thinning, prescribed fire, and prescribed natural fire, where appropriate (Amoroso et al. 2011). But as might be expected, some stakeholders do not agree with the premise that timber harvest can serve as a proxy for one or more components of a mixed-severity fire regime (Lindenmayer et al. 2009).

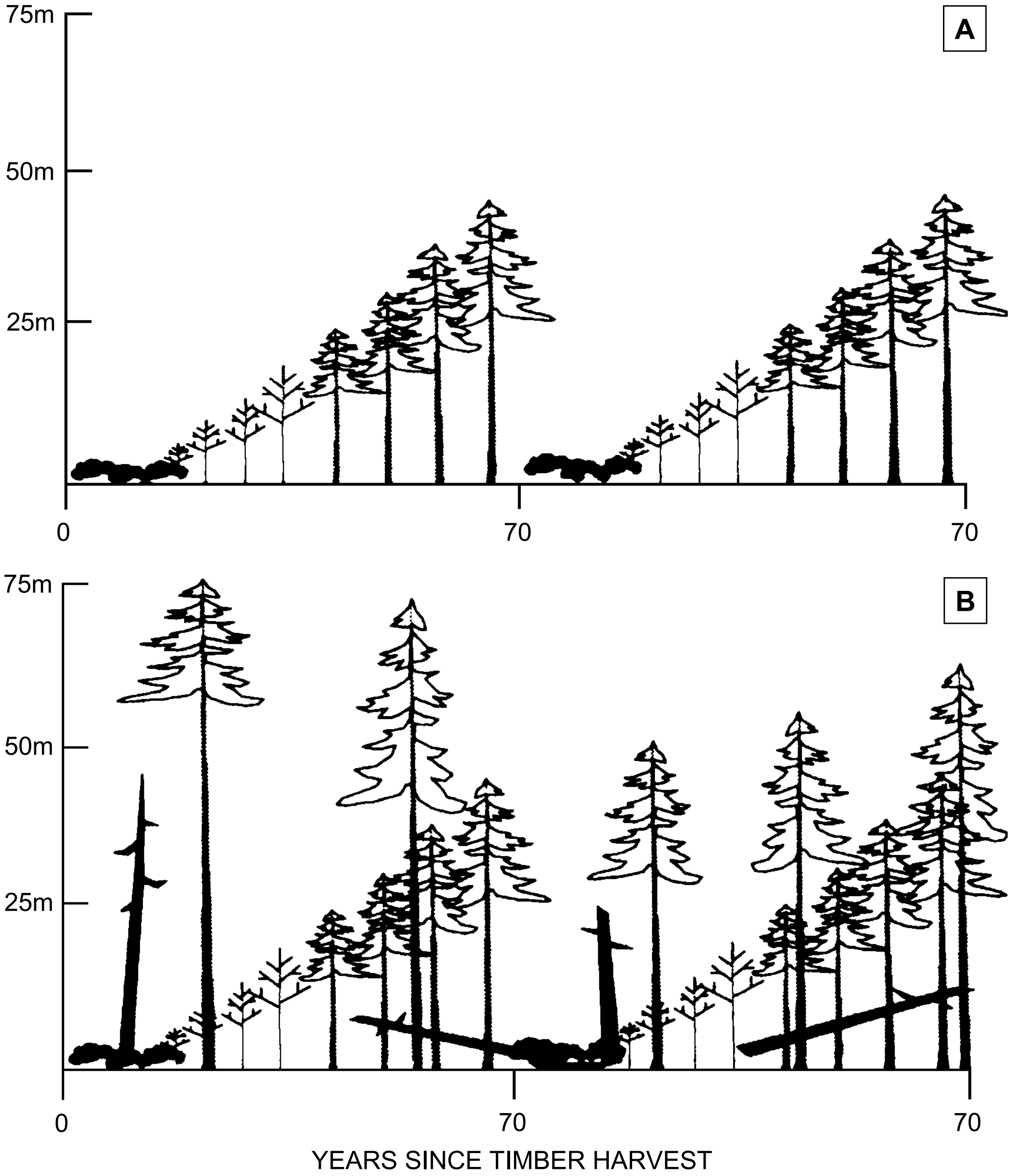


Figure 72 – Comparison of idealized development in stands designed for (A) maximum wood production and (B) both structural diversity and wood production (adapted from Hansen et al. 1991, and taken from Powell 2000). Traditional management practices for maximizing wood production tended to create tree plantations that were relatively simple with respect to structure, habitat diversity, and retention of biological legacies. Contemporary practices that better mimic natural forest patterns, processes, and interactions can maintain biological diversity while simultaneously providing an opportunity for wood production. In addition to a young (regenerating) tree cohort, the contemporary stand (B) features these legacies derived from one or more of the previous forest generations: remnant old, live trees; downed dead wood; and standing dead wood (snags). Providing legacies recognizes that the effects of stand-replacing disturbance are heterogeneous, creating a variety of patch sizes, shapes, ages, and intra-patch structural elements (shrubs, snags, decaying wood, live conifers and broadleaf trees, etc.) (Aplet and Keeton 1999, Rochelle et al. 1999).

Over the last 30 years, political and legal pressures related to society's concerns about forest management practices caused a major reassessment of forestry objectives across North America. In large part, I believe this reassessment was designed to address short-term concerns like aesthetics (e.g., limit clearcutting because of its unattractive appearance) (Bliss 2000), rather than considering the long-term implications of existing ecological conditions. During this reassessment era, some stakeholders applied increasing pressure for a 'no active management' approach, a decision that also has enduring ecological consequences (Hansen and Goheen 2000).

Clearcutting and other even-aged, regeneration cutting methods (shelterwood and seed-tree cutting) are widely perceived as being incompatible with ecological integrity, so recent trends have featured more use of selective harvest and uneven-aged management (individual-tree and group selection cutting). In dry-forest ecosystems, uneven-aged management is appropriate for many circumstances and, if skillfully applied by well-trained and experienced practitioners, it would be expected to help restore a more sustainable mix of species composition, forest structure, and stand density (Graham and Jain 2005, Graham et al. 2007). But for some portions of the dry mixed-conifer zone, uneven-aged management could also promote uncharacteristic representation of Douglas-fir, grand fir, and western dwarf mistletoe, which are some of the late-seral species on these sites (see Powell 2014 for more details).

For moist mixed-conifer forest forming the basis of this white paper, individual-tree selection and similar variants of partial cutting would not be expected to create outcomes congruent with natural disturbance processes, particularly if applied across large landscapes or as a dominant management practice. For gap-phase processes such as pathogenic stem decay or root disease, group selection cutting could certainly be used to mimic their fine spatial scale, but this strategy would not necessarily reduce future impact from these agents because the early- or mid-seral species (western larch, and western white, lodgepole, and ponderosa pines) that are immune or relatively disease-resistant are generally not favored by the small opening sizes and high amounts of side shading associated with group-selection cutting.

Is Timber Harvest Appropriate For Moist Forests?

Some special-interest groups would like to return forest ecosystems to their presettlement condition, particularly if this result would reduce or eliminate tree harvest and other anthropogenic activities. Their agenda is about preservation rather than conservation, focusing on naturalism and aesthetics, and promoting little if any human interaction with the environment. According to this steady-state perspective, if the presettlement condition was one of continuous trees, then little logging or active management would be appropriate in the contemporary forest (Botkin 1995, Budiansky 1995, Chase 2001, Cronon 1996, Lomborg 2001).

But what if the presettlement landscape was a mosaic, featuring forests interspersed with meadows and wetlands, and within the forest occurred a patchwork of different ages and structural stages? And what if disturbance processes operated freely in these historical landscapes? If these questions could be answered in the affirmative, then more opening of the forest would be consistent with presettlement conditions, and tree harvest could occur at a higher level (Botkin 1995). And, consistent with the Natural Disturbance Model (Box 3) and the Intermediate Disturbance Hypothesis (fig. 9), the goal of timber harvest is to design harvesting patterns that emulate the ecological outcomes of natural disturbance.

Furthermore, what if Native Americans had a pervasive influence on presettlement conditions? If this is true, and historical evidence indicates it is (Anderson 2006, Barrett 1980, Boyd 1999, Denevan 1992, Kay 1994, Knudson 1980, Robbins 1997, Stewart 2009), then we might need to carry out a similar level of human activity to approximate the presettlement conditions (Botkin 1995). [See "Native Americans and Fire" in section 5.11 for more background information about this subject.]

A broader recognition of Native American influence logically leads to the following result: "a holistic landscape approach to conservation, driven by a vision of humans and other species co-mingling across reserves and developed lands, has gradually gained prominence over the last 20 years" (Heller and Zavaleta 2009).

More wood grows per acre per year in the interior Northwest than can be decomposed by natural processes in its summer-dry Mediterranean climate (see fig. 7) (Oliver et al. 1994, Olson 1963). "The right combinations of moisture and temperature do not exist long enough each year for wood to decompose as fast as it grows. Fires, insects, and diseases are the disturbance agents that historically recycled 'excess' biomass in the interior West" (Ferguson et al. 1998).

The situation where wildfire, spruce budworm, and other disturbance agents function as initiators and catalysts for fine-scale decomposition processes (such as fungal decay) also exists for the Umatilla National Forest, where four of the seven primary tree species (grand fir, lodgepole pine, subalpine fir, and western larch, specifically) have average annual mortality rates that currently exceed average annual growth[6] (Christensen et al. 2007, p. 31).

Forest managers can emulate some aspects of the decomposition functions of natural disturbance processes by how they implement timber harvest, slash treatment, and site preparation activities for moist-forest ecosystems (fig. 73). And, note that thinnings such as the one shown in figure 73 may have high value as a future climate change mitigation measure: research found that when trees were retained in an even distribution (in contrast to a groupy or clumpy pattern), change in soil water equivalent, which is related to snow accumulation and soil moisture retention, was increased significantly as compared to untreated controls (Woods et al. 2006).

---

[6] During a previous era emphasizing timber production, the Umatilla National Forest's current situation, where four of seven primary forest types are dying faster than they are growing (see fig. 31 in Christensen et al. 2007), would certainly be viewed as an indicator of poor forest health!

Figure 73 – Contemporary timber harvest in moist grand fir forest. This is an improvement cutting on the Loon timber sale (unit 17), Walla Walla Ranger District of the Umatilla National Forest. Moist-forest timber sales emphasize activities such as creating openings to encourage regeneration, leaving a few high-density areas as elk thermal or hiding cover, and reducing tree density to create large structure more quickly (Franklin et al. 2007). Historically, many of the entries into mature grand fir stands implemented clearcutting or even-aged regeneration cutting methods (figure 14 provides examples of regeneration cutting methods; the Young Stand Development narrative later in this section provides more background history about the clearcutting era in moist forests).

6.6 Active Management Examples And Recommendations

> **Hands off management shows good taste but poor insight. The hope of the future lies not in curbing the influence of human occupancy – it is already too late for that – but in creating a better understanding of the extent of that influence and a new ethic for its governance.**
>
> *Aldo Leopold, Game Management (1933)*

Moist forests are extremely resilient. If defoliating insects kill many of the Douglas-fir and grand fir trees, then surviving western larches and lodgepole pines ensure continuity of the forest. A mixed-severity or stand-replacement fire promotes rapid development of postfire lodgepole pine or larch-lodgepole stands. Depending on the circumstances, a mountain pine beetle outbreak in lodgepole pine can release advance regeneration of grand fir or Engelmann spruce, or result in another generation of lodgepole pine – but either eventuality provides forest continuity. And when wind blows over a tall cohort of Engelmann spruce, advance regeneration of subalpine fir or spruce is ready and willing to claim the liberated growing space.

The high level of moist-forest resilience is anchored in the ecological redundancy provided by a diverse tree species composition, but it does not imply that any and all changes to moist forest are equally desirable. A wide range of active management options are feasible for moist forest, and each of them has varying influences on future forest conditions. But regardless of their influence, active management treatments that most closely approximate the natural disturbance regime are most likely to be successful over the long term.

Unlike the dry-forest biophysical environment of the Blue Mountains, where few acres remain within their range of variation for composition, structure, or density, many areas within the moist-forest zone still exist within their range of variation. This major difference between the two environments relates to numerous factors:

1) Moist forest has inherently broader ranges of variation than other biophysical environments (see the 'Goldilocks zone' in figure 9).
2) Moist forest has experienced less anthropogenic modification than dry forest (from timber harvest, livestock grazing, etc.).
3) The longer fire return intervals for moist forest (especially fire regime IIIb and IIIc), in contrast to short intervals associated with dry forest (fire regime I; fig. 44), results in fewer missed fire cycles (for moist forest) during the past century.

An important caveat: although moist forest may be largely within the range of variation at the scale of any individual stand, it is becoming increasingly obvious that moist-forest changes have occurred at a landscape scale, and these changes are reflected in recent trends for wildfire, defoliating insects, and other disturbance processes (see "Effects of Fire Suppression on Mixed-Severity Fire Regimes" in section 5.11 for an example). Broad-scale assessments disclosed many of these changes.

One broad-scale assessment found that three watersheds in the northern part of the Umatilla National Forest experienced a 90% decline in ponderosa pine cover, and corresponding 35% to 230% increases in Douglas-fir and grand fir cover, between 1938 and 1987. Western larch cover also declined by 80% to 100% in the same watersheds (Lehmkuhl et al. 1994). Two of the three watersheds used by Lehmkuhl et al. (1994) occur in the Wenaha-Tucannon Wilderness (both are located in the Grande Ronde River basin). Even in these relatively 'undisturbed' Wilderness watersheds, it was found that substantial declines in ponderosa pine, grass-forb, and other early-seral patch types had occurred.

These results (Lehmkuhl et al. 1994) reflect an important impact of long-term fire suppression – the landscape has become more homogeneous, with fewer vegetation types (particularly early-seral stages), larger patches at lower patch densities, and less total edge than would have been produced by the historical disturbance regime. These changes make it much more difficult to interpret existing landscape conditions as a way to infer historical disturbance regimes because increased homogeneity often masks historical spatial patterns. Similar results were obtained from broad-scale landscape assessments completed during the Interior Columbia Basin Ecosystem Management Project (as reported in Hessburg et al. 1999a, 2000).

An All-Important Consideration Of Scale

The fundamental importance of scale permeates all aspects of active management (Cumming et al. 2013) (fig. 74). For example, many recent journal papers in the landscape ecology and forest management literature have emphasized the importance of spatial heterogeneity (Hessburg et al. 1999b, 1999c, 2000, 2004, 2007; Turner et al. 1989, 1994, 2001; and others in the References section).

When evaluating spatial heterogeneity for dry forests, it is important to consider the sub-stand level because scale is fine-grained and intricate for dry forests – ponderosa pine stands historically featured a groupy or clumpy structure at the sub-stand scale (Harrod et al. 1999; also see Powell 2014). The 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 the stand functions as an aggregating unit (e.g., stands are a mid-scale unit representing aggregations of fine-scale clumps).
Note: I define a 'base-level' unit as the scale at which ecosystem processes result in tree regeneration sufficient to perpetuate the forest type.

For moist forests, the landscape functions as an aggregating unit – in this ecosystem type, discrete stands (patches) with relatively homogeneous composition or structure are created by mid-scale disturbance processes such as mixed-severity fire. In contrast to the dry-forest situation, where tree clumps are functionally important, stands function as the base-level unit influencing moist-forest regeneration processes. So, spatial heterogeneity for moist forests consists of varying compositional or structural patches distributed across a landscape, creating a mosaic where stands (rather than groups or clumps) function as primary building blocks of inherent pattern. These functional differences, most of which are strongly related to scale, form the ecological crux between dry and moist forest.

Note about clumps: Tree clusters are a common feature of many forest ecosystem types. For dry forests, tree clusters can be created by the disturbance regime (i.e., variable thinning provided by surface fire, or pockets of tree mortality caused by western pine beetle), in which case clusters function as an effective indicator of ecosystem process. But dry-forest clusters are also caused by the seed caching activity of deer mice, chipmunks, and other small mammals (Keyes et al. 2007).

In moist or cold forests, tree clusters also occur (Larson and Churchill 2008), but in my experience, a groupy or clumpy structure in these ecosystem types may not necessarily be produced by the disturbance regime (although windthrow often leads to clumps). Moist-forest clumps often reflect the vagaries of seed dispersal, the pattern and juxtaposition of seedling establishment 'safe sites,' seed caching patterns by birds and small mammals, and other random or stochastic factors.

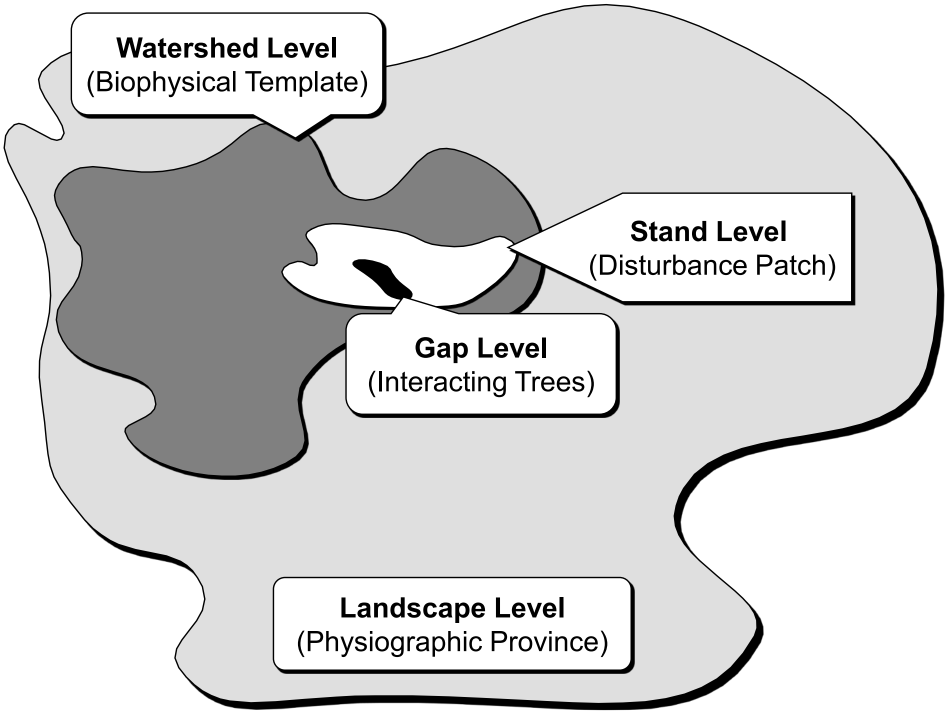


Figure 74 – Generalized representation of a scales hierarchy (based on Urban et al. 1987). It is now widely recognized that ecosystems occur at a variety of spatial scales, with smaller systems nested within larger ones in a hierarchical fashion. This figure depicts a forested landscape as a hierarchy of gaps, stands, and watersheds. Trees within a gap interact more among themselves than with trees beyond the gap. By extension, a larger forest area can be considered as a mosaic of gap-sized patches, with each gap developing somewhat independently. Stands are comprised of contiguous gaps having similar characteristics. Stands in one drainage share a similar biophysical template and interact more with each other than with stands in other watersheds. At a higher level, landscapes can be defined as aggregations of similar, interacting watersheds (Urban et al. 1987).

Note that a landscape perspective has been emphasized over the past few decades, but a landscape perspective is more than just a geographical concept – it involves accounting for long temporal periods; acknowledging a wide diversity of issues, concerns, and resources; and incorporating similar broad-scale considerations. Also, note that 'landscape analysis,' as a process, differs from using a landscape as an analysis area – landscape analysis involves explicit characterization of landscape components (e.g., matrix, patches, corridors, and processes; see table 7 for an example), whereas using a landscape for broad-scale analysis purposes involves delineating a large geographical area for which a planning or assessment process will be conducted.

The main difference is this: in some circumstances, a clumpy tree structure reflects ecosystem function (e.g., pattern follows process, and process influences future pattern), whereas in others, it may reflect the inherently random nature of seed dispersal and seedling establishment. Why is this distinction important?

Practitioners must be able to interpret spatial pattern in order to understand if it should be emulated by their proposed treatments. Does pattern reflect inherent ecosystem process, in which case it would be repeatable across a landscape or planning area? Or, is it simply a product of random historical circumstances that may not be repeated again? The answer to these questions is important because contemporary forest science literature emphasizes provision of spatial heterogeneity, but:
it is most important to ensure heterogeneity for biophysical environments where it was produced by the inherent disturbance regime.

Moist-Forest Scale: Seen Versus The Unseen

The fundamental importance of scale 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 the sub-stand scale for dry forest (e.g., at the scale of a tree clump) – Is a characteristic clumpy structure still evident for a dry forest (such as figs. 44-46 in Powell 2014)? If so, does the composition of any particular dry-forest stand feature a majority of ponderosa pine rather than late-seral species such as grand fir?
Dry-forest composition could be evaluated this way: up to 70% of the clumps in a dry-forest stand should have a majority of ponderosa pine (not necessarily pure pine), rather than a majority of Douglas-fir or grand fir.

Dry-forest conditions have changed remarkably as a result of fire exclusion, livestock grazing, and selective timber harvest (Powell 2014) – and these changes are overtly expressed in existing conditions, including at the clump scale, so when entering most dry-forest stands, it quickly becomes apparent when they are substantially departed from reference conditions.

Contrast a dry-forest evaluation with moist forest, where monitoring of composition, structure, and density is best accomplished at a landscape scale. Does a moist-forest landscape, for example, contain the proper proportion of larch-dominated patches, and if larch is not the matrix component, then which matrix type does the larch patches exist in?

This moist-forest example demonstrates that most of the moist-forest components are not group- or clump-driven but patch-driven – the native (unsuppressed) disturbance regime creates patches, and patches exist at the stand scale. Contrast this situation with dry forest, where fine-grain or gap-phase processes create clumps or groups at a sub-stand scale.[7]

---

[7] This scale discussion makes a specific distinction between 'patches,' and 'groups' or 'clumps.' As used in this section, groups or clumps are considered to be synonymous; they represent the primary organizational unit for dry forests, consisting of tree clusters occupying 0.1 to 0.5 acres (Powell 2014). In a forestry context, groups or clumps are much too small to be managed and monitored as individual stands. Generally, some combination of groups or clumps in a repeating pattern is aggregated into a 'stand' for management purposes. Patches, however, are much larger than groups or clumps. An individual patch will typically feature a relatively consistent species composition, structural stage, or stand density. Patches are managed as individual stands. Moist-forest landscapes are dominated by discrete patches, not by groups or clumps.

And when entering most mid- to late-seral moist-forest patches, it may not seem as if they have departed from an expected reference condition – but what can be important when evaluating moist forest is not what is present now (the seen), but what might have been present had disturbance processes not been influenced in the past (the unseen). Would an existing late-seral patch of grand fir have been an early-seral patch of western larch if wildfire, or a fire ignition, had not been suppressed in 1939?

This difference in perspective occurs because dry-forest conditions (changes in species composition, density, multi-layer structure, etc.) tend to be obvious at a fine scale (foreground view) for the great majority of dry-forest stands. For moist forest, changes are much more subtle – late-seral stand conditions can be entirely appropriate if a late-seral patch is what would have been expected.

But if a moist-forest landscape has a predominance of late-seral patches, then the 'unseen' is important because we would expect a properly-functioning disturbance regime to create an appropriate representation of early- and mid-seral patches. For example, 10-30% of the patches in a moist-forest landscape would be dominated by early-seral western larch (table 18) if the disturbance regime is functioning properly.

And at the risk of over-simplifying these relationships, differences between dry and moist forest illustrate that ecosystem resilience is best addressed at the stand level for dry forest (does any individual stand have proper amounts and distribution of clumps with respect to composition, structure, and density?), and at the landscape level for moist forest (does any individual landscape have proper amounts and distribution of patches or stands with respect to composition, structure, and density?).

Disturbance Emulation As A Basis For Moist-Forest Management

The management strategy most likely to sustain high levels of ecosystem integrity and resilience is emulation of natural disturbance processes (see box 3 and table 6, and discussion in section 4.5). A disturbance emulation strategy outlines objectives and practices resulting in the least possible difference between active management practices and natural disturbance. A basic premise of disturbance emulation is that silvicultural practices, including prescribed fire and other management activities, should mimic the natural disturbance regime (Cui and Perera 2008; Drever et al. 2006; Perera and Cui 2010; Perera et al. 2004, 2008) – and not just the fire characteristics, but all aspects including consideration of wind and other processes.

A fundamental assumption of the disturbance emulation strategy is that plants and animals of a forest are adapted to conditions created by natural disturbance, so they should cope most easily with the ecological changes caused by forest management if the pattern and structure created by these activities resemble those of natural disturbance (DeLong 2011). And we should also acknowledge that "no single near-to-nature land management or silvicultural system is universally suitable. Instead, many methods can be adopted in different combinations and contexts, depending on forest ecology, local land use history, and the management goal for the landscape" (Angelstam 2003, p. 212).

While a safe presumption is that active management will never function as a perfect surrogate for disturbance, presumably because nature has tremendous inherent variability and no two successive disturbance events will ever be identical anyway, it is also true that silvicultural interventions can effectively mimic certain aspects of disturbance processes. Disturbance emulation is similar to the range of variation in this respect – disturbance functioned within an envelope of variability characteristic to both the process and to site potential, but the effect of repeated events fluctuated between the upper and lower limits of a range, corroborating that nature did not 'manage' with perfect replication from one event to the next. If active management can mimic nature well enough to ensure its effects are within the range of variation for a process, then it is reasonable to assume that management is not causing impact beyond what would have been produced naturally.

Silvicultural practices can be used to modify species composition and forest structure, reduce intertree competition, maintain biological diversity (particularly regarding undergrowth plant composition, vigor, and density), address the susceptibility or vulnerability of forest stands to insect and disease attack, and increase a forest's resistance and resilience characteristics by improving tree vigor. One example is that small timber-harvest patches (fig. 75) can facilitate development of a diverse age-class structure for a landscape that has been intensively managed in the past (e.g., landscapes simplified by using timber harvest in homogeneous, even-aged patches, and by fire exclusion) (Lyons-Tinsley and Peterson 2012).

Using silvicultural practices to emulate inherent disturbance regimes (Everett et al. 2000) is also congruent with our Forest Service Course to the Future (Thomas 1995, p. 158): "the following examples illustrate possible management actions to protect ecosystems: (i) understanding the roles of fire, insects and diseases, and drought cycles in shaping ecosystems, and bringing that understanding to bear in management; (ii) developing and using measures of sustainability; (iii) establishing and managing wetland and riparian conservation areas; (iv) evaluating the effects of human use and habitation on the sustainability of ecosystems (in urban, suburban, and rural settings) while supporting the quality of life in those ecosystems."

Disturbance processes can be categorized as either stand-initiating or stand-maintaining, and the stand-maintaining category is further subdivided into two functional groups: top-down (releasing) and bottom-up (maintenance) (see fig. 13). Thinnings mimic the bottom-up effects of insect defoliators (and their feeding-ladder effect; fig. 22), dwarf mistletoe parasitism, and density-dependent mortality, whereas salvage harvest of dead trees, or regeneration cutting in live stands (fig. 75), can mimic certain aspects of the windstorm, root disease, bark beetle, and senescence processes within the top-down functional group. But harvest only mimics certain aspects of these processes because it removes material that would otherwise remain, and it does not remove material that would otherwise be consumed or lost.

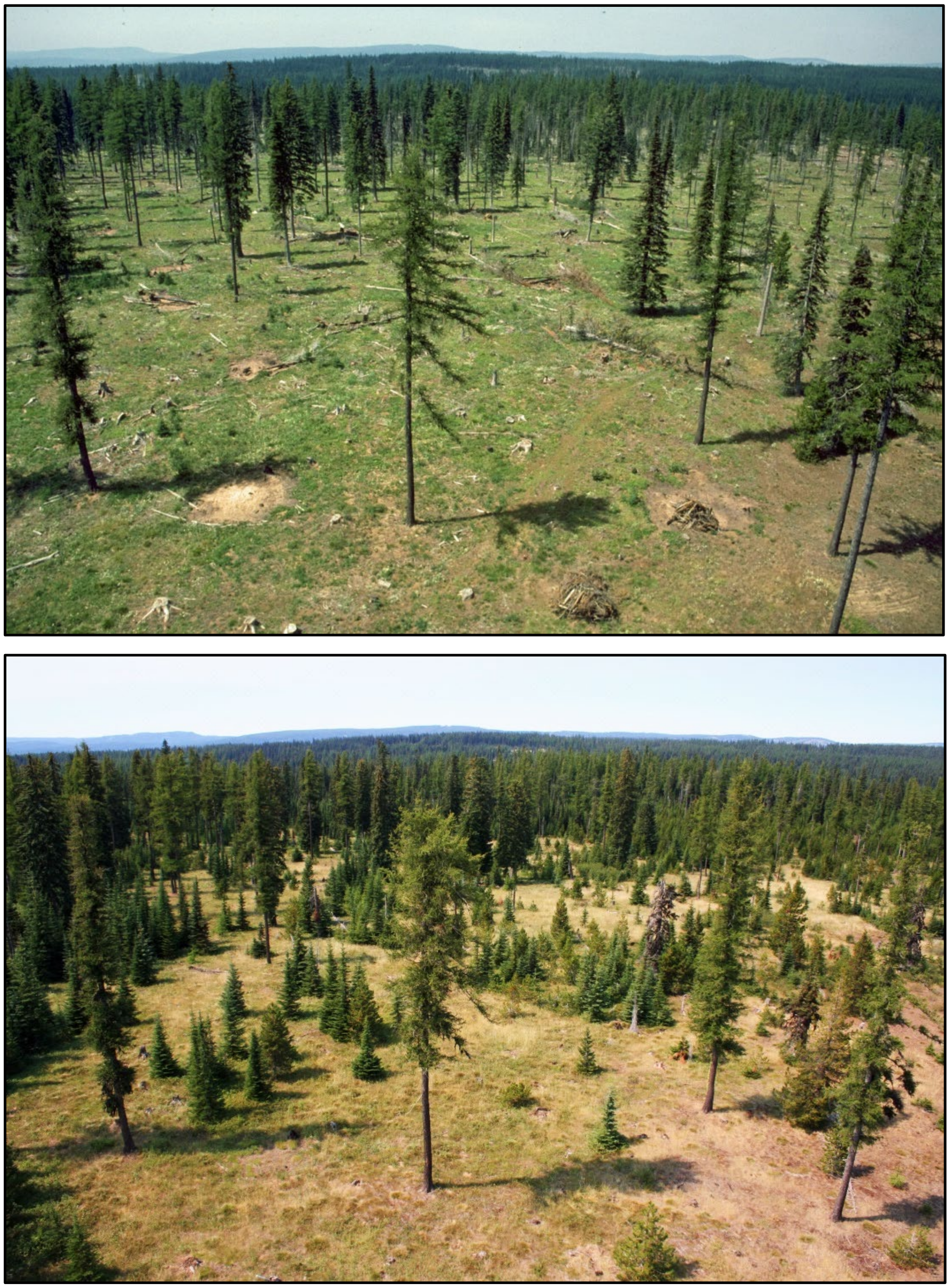


Figure 75 – Seed-tree seed cut in the Lookout Mountain area of the northern Blue Mountains (late 1990s view immediately after implementation, above, and summer 2013 view, below). This activity was designed to increase the representation of western larch on a moist-forest biophysical environment (see fig. 80) by creating post-treatment conditions suitable for regeneration of early-seral tree species such as western larch and lodgepole pine. Although post-treatment conditions were appropriate in terms of meeting the regeneration ecology requirements of early-seral species, note that a relative paucity of dead wood was retained, both in the standing (snags) and down form. Although western larch seed trees were retained exclusively in the foreground view (top), much of the established regeneration (bottom) appears to be late-seral species such as subalpine fir, Engelmann spruce, and grand fir.

An ecological forestry approach typically emphasizes increasing stand complexity as a way to address biodiversity concerns (Franklin et al. 2007), and a recent study suggests that "management prescriptions should foster a well-thinned, yet diverse sub-canopy with a canopy of larger than average trees" if an objective is to improve bat habitat (Dodd et al. 2012). This bat-habitat study reinforces the concept that an important feature of stand complexity is layering, but layers don't necessarily have to be dense to provide valuable ecosystem services such as bat habitat.

"Maintaining forest biodiversity means that both the range of natural disturbance regimes and the resulting forest and woodland environments to which species have adapted (the ecological dimension) must be understood, and that a sufficiently wide range of different land management regimes must be applied (the management dimension). It also means that the management regime chosen for a given forest environment must harmonize with its ecological past" (Angelstam 2003, p. 217).

If society, as reflected in the perspectives of stakeholders, partners, and collaborators concerned about public lands management, is truly concerned about sustaining biodiversity for moist forests of the Blue Mountains, then disturbance emulation strategies incorporating certain aspects of both even-aged and uneven-aged silvicultural systems will be developed. And if this eventuality occurs, it should reflect a local context by using a collaborative approach involving managers and stakeholders, and it should strive to develop a consensus-based framework to create, sustain, and monitor a full range of successional (developmental) stages through time and space.

Historical Context For Moist-Forest Timber Production

As national forests in the Blue Mountains began to be managed shortly after the turn of the 20th century, there quickly arose a concern about how to deal with what was termed 'inferior' species, e.g., anything that wasn't western yellow (ponderosa) pine. T.J. Starker (1915) described options for dealing with inferior species on the Whitman National Forest; they occurred primarily in two ecological settings: as a North Slope type comprised mostly of western larch and firs, and in the western yellow pine areas where inferior species occurred singly or in scattered groups. On north-slope sites, the timber sales of that era resulted in all of the western yellow pine being removed, along with most of the Douglas-fir and western larch. This approach resulted in lodgepole pine and grand fir being retained (and therefore promoted because they functioned as a seed source for future natural regeneration).

Starker (1915) recounts a site visit from Raphael Zon, an early ecologist stationed in the Forest Service's Washington Office. Mr. Zon recommended that the Forest not touch the north slopes since they were a losing proposition commercially (the loggers claimed they lost $3 to $4 for every thousand board feet of inferior species removed), the post-harvest stand had doubtful character and value (silviculturally), and because north slopes had an important influence on stream flow.

For areas where inferior species were intermixed with yellow pine, Starker (1915) felt they should be cut very heavily. By removing the inferior species, it would be possible for these areas to regenerate to commercially valuable yellow pine. It was also felt that larch and fir were 'out of their place' on dryer sites supporting a high percentage of yellow pine, so their removal was desirable from a silvicultural perspective as well.

It was estimated, as based on empirical information from the W.H. Eccles and Baker White Pine Lumber Company timber sale areas, that inferior species comprised 15% of the yellow pine areas. Starker felt that no less than one third of the inferior species should be removed in a timber sale, not just for the reasons mentioned above but because they were more prone to windthrow than yellow pine.

Starker also recommended that creek bottom sites be deferred from logging until the inferior species problem was solved, since they contained a high proportion of inferior species. Starker reported on seven Whitman National Forest timber sales operating between 1910 and 1913; the volume of inferior species for these sales ranged from 11.4% to 31.9%, showing that inferior species were obviously being removed. Starker summarized his thoughts about the 'inferior species' issue in this quote: "Under the present system of conducting our timber sales we are cutting all the yellow pine and most of the Douglas fir and larch on the north slopes. This leaves a majority of lodgepole pine and white fir, which soon becomes so dense that no other species can get a foothold and the resulting stand will be a very inferior jungle" (Starker 1915).

What Role Might Timber Management Play In The Future?

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; Lehmkuhl et al. 1994; Hessburg et al. 1994, 1999a; 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 moist forests.

Using timber management practices to maintain stands within their 'management zone' (fig. 76) could help sustain scenic beauty by minimizing bark-beetle outbreaks. Research found that "psychological utility or visual preference drops rapidly as damage increases to approximately 10 percent of the forest area. Declines in preference are slight thereafter. It appears that it is more important to prevent new insect outbreaks than it is to prevent additional spread from an esthetic impact standpoint" (Buhyoff and Leuschner 1978, p. 424).

Commercial thinnings and other density management treatments are often targeted toward stands with a predominance of early-seral tree species, primarily because early-seral species have lower resistance to chronic, competition-induced stress than late-seral species (fig. 79). Under these circumstances, thinning can relieve density-dependent stress and thereby improve the survivability of early-seral species, particularly for mixed stands where early- and late-seral species coexist (Powell 1999b).

"The human agents of disturbance [Native Americans] are no longer operating, so the effects of American Indian burning must be simulated using either prescribed fire or mechanical methods. As Dr. Leopold said in his 1983 letter to the U.S. Park Service, 'A chain-saw would do wonders.' Science-based timber harvesting is the safest and most effective way to mimic Indian and lightning fires within an occupied landscape where prescribed fire is too dangerous. It can also be used to meet contemporary resource needs while still restoring an approximation of the historical forest ecosystem. That is, timber harvesting has the added advantage of creating jobs, producing wood, and generating revenue to pay for ecosystem management" (Bonnicksen et al. 1999).

"Even-aged timber harvesting can mimic natural fires by leaving snags and fallen logs behind, and by creating openings shaped to look as if they were formed by fire. Patch cuts and group selection timber harvesting can mimic surface fires by producing small scale mosaics. Single tree selection can mimic single tree falls in uneven-aged forests composed of shade-tolerant trees. Snags and fallen logs should also be left behind when using these techniques. Thinning and prescribed burning also can keep the forest clear of debris and small trees. Low intensity prescribed fire may also be required following timber harvesting to more closely approximate the ecological effects of natural fires" (Bonnicksen et al. 1999).

When considering how, and whether, we might choose to mimic natural fires by using timber harvest, we should accept that harvest will be perceived as having both positive and negative impacts. "If we believe the impacts of harvesting and [wood] consumption are primarily positive impacts, we should embrace them locally. If we believe the impacts are negative, we should take responsibility for them locally and mitigate them. If we believe the impacts are a mix of positive and negative, we should welcome the positive aspects and mitigate the negative aspects as we endeavor to do a better job of forest ecosystem management" (Shifley 2006).

Explanatory Notes for Figures 77-78, and Table 24. Figure 78 provides stocking-level tools for active management of moist upland forest. Figure 78 has three parts – conventional stocking levels expressed by using four stand density thresholds (fig. 78a), 'special-purpose' stocking levels expressed by using two levels of crown-fire susceptibility (fig. 78b), and an example of how the stocking-level charts can be used to assess treatment effectiveness (fig. 78c). Notes about the threshold levels on the stocking charts (figs. 78a and 78b) are as follows:

Maximum density: Although rarely observed in nature, maximum density represents a useful upper limit, so it is often used when establishing stocking levels. Maximum density is included in figure 77, and in figures 78a and 78c.

Full stocking: Full stocking is also referred to as normal density. Full stocking pertains to single-cohort (even-aged) stands where intertree competition results in development of crown classes (e.g., differentiation) – dominant, codominant, intermediate, and subcanopy trees are present in differentiated stands. As shown in figure 77, normal density/full-stocking occurs in the self-thinning zone where stand density (and intertree competition) is high enough to cause tree mortality.

Upper limit of the management zone (Upper Limit in fig. 78a): This stocking level corresponds with the 'lower limit of the self-thinning zone' threshold shown in figure 77. 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 the management zone (Lower Limit in fig. 78a): This stocking level corresponds with the 'lower limit of full site occupancy' threshold shown in figure 77. This threshold functions well as a lower management limit because a site is fully occupied at stocking levels above it – growing space is not being 'wasted' (under-utilized) at these stocking levels.

High susceptibility to crown fire (High Susceptibility in fig. 78b): 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. 78b): 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).

The CBD values (0.10 and 0.05 kg/m^3) for crown-fire susceptibility were translated into their corresponding forestry metrics (Powell 2010a) in order to prepare a stocking chart (fig. 78b).

As shown in the titles for the stocking charts, the stocking threshold information (color lines in figures 78a and 78c) pertains to an even-aged stand structure (the lines would have the same shape but would be lower for irregular or uneven-aged stand structures), and for a mixed species composition.

The mixed-composition stocking levels represent weighted averages (30% Douglas-fir, 20% western larch, 20% lodgepole pine, and 30% grand fir for the Moist Upland Forest potential vegetation group) of the stand density index values for pure stands of the species, by stocking-level threshold (maximum density, full stocking, etc.), for the Moist Upland Forest PVG.

Here is an example of how weighted averaging was completed for the mixed-composition stocking levels: refer to data for the 25% of maximum density stocking level (specifically, the first number in the TPA column, in the High Tree Growth section, of table 24):
30% DF + 20% WL + 20% LP + 30% GF =
(105 × .3) + (116 × .2) + (84 × .2) + (163 × .3) = 120.4 (120 appears in the TPA column of table 24 for Mixed Composition at the bottom of the High Tree Growth section).

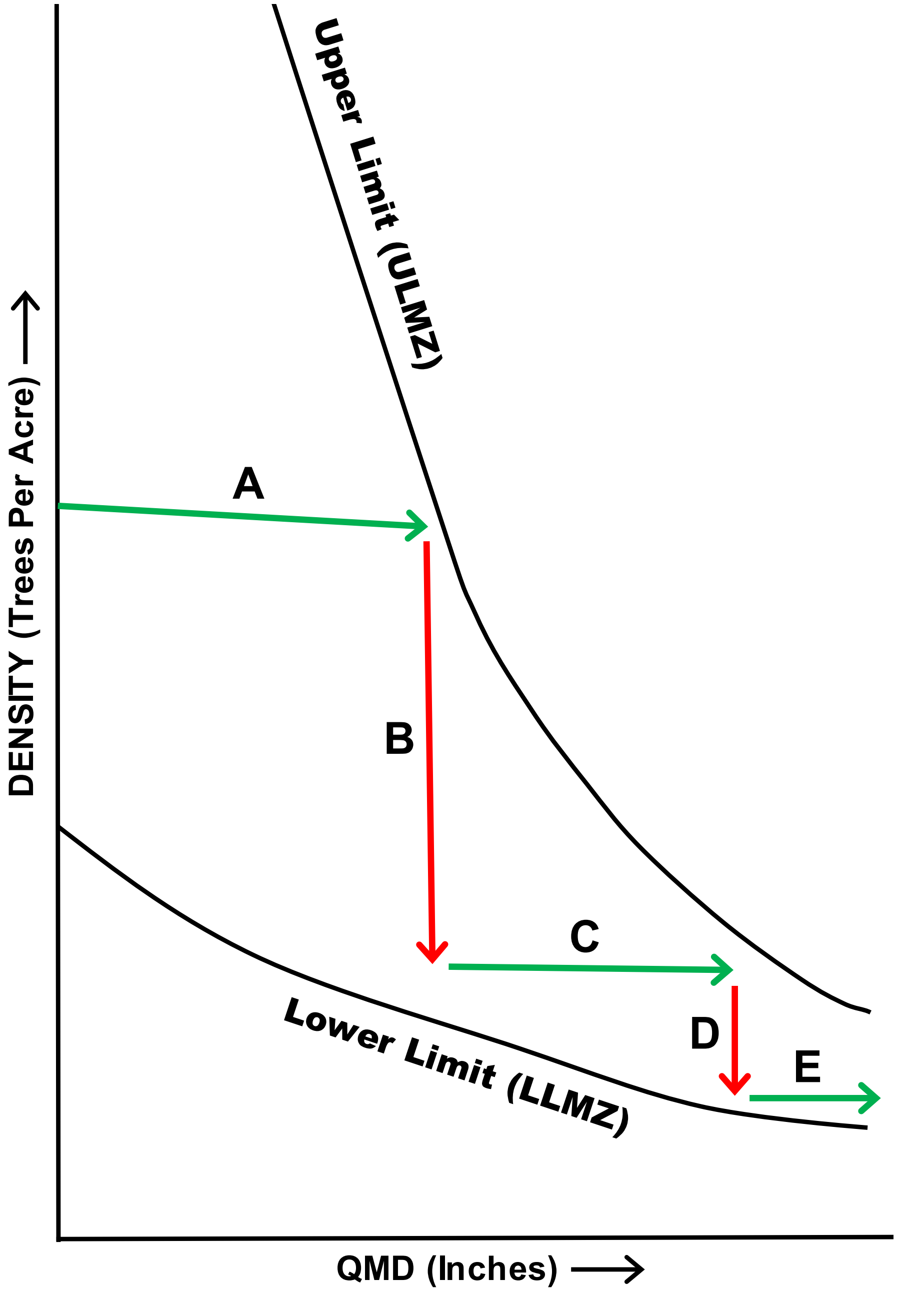


Figure 76 – Hypothetical thinning regime utilizing the upper and lower limits of a management zone as maximum and minimum stocking curves (the solid curving lines). This figure shows a stocking-level chart and how it could be used to help prepare a thinning regime. In this figure, initial stocking begins in the management zone and stand growth causes the QMD to increase toward the upper limit (this is segment A of the dashed line). When the trajectory approaches or reaches the upper limit, a thinning is completed and stocking is reduced until it reaches, or is close to, the lower limit (segment B). Further growth causes the stand to once again approach the upper limit (segment C), at which point another thinning is scheduled and stand density is reduced toward the lower limit once again (segment D). For this thinning regime, stand density would ostensibly be low enough to stay within the management zone after completing the second thinning (segment E).

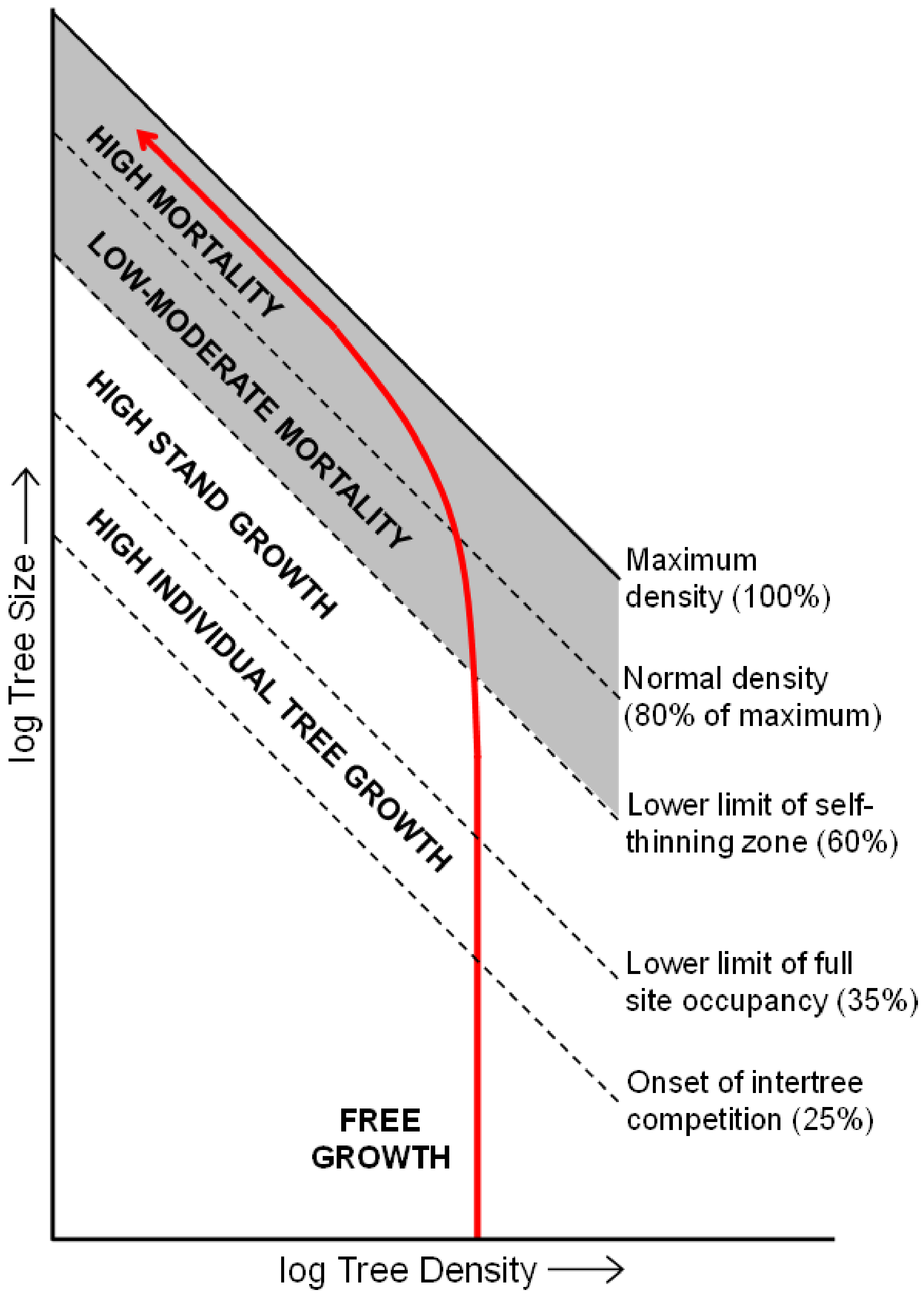


Figure 77 – Stand development indexed to maximum density (also see fig. 41). Initially, trees are too small to use all of a site's resources, and they experience a period of free growth (no intertree competition is occurring). Eventually, roots and crowns begin to interact and the 'onset of intertree competition' threshold is reached. As the stand continues growing through a zone of high individual tree growth, trees capture growing space and the 'lower limit of full site occupancy' threshold is breached. This next zone features high stand growth. As competition intensifies, stands enter a self-thinning zone by crossing the 'lower limit of self-thinning zone' threshold. In the self-thinning zone (gray area), a tree can only increase in size after neighboring trees relinquish their growing space by dying. The pace of tree mortality quickens as the stand passes the 'normal density' threshold and approaches maximum density. Maximum density, shown as a solid line because it is an absolute threshold, is the reference level (100%) for the stocking system shown here.

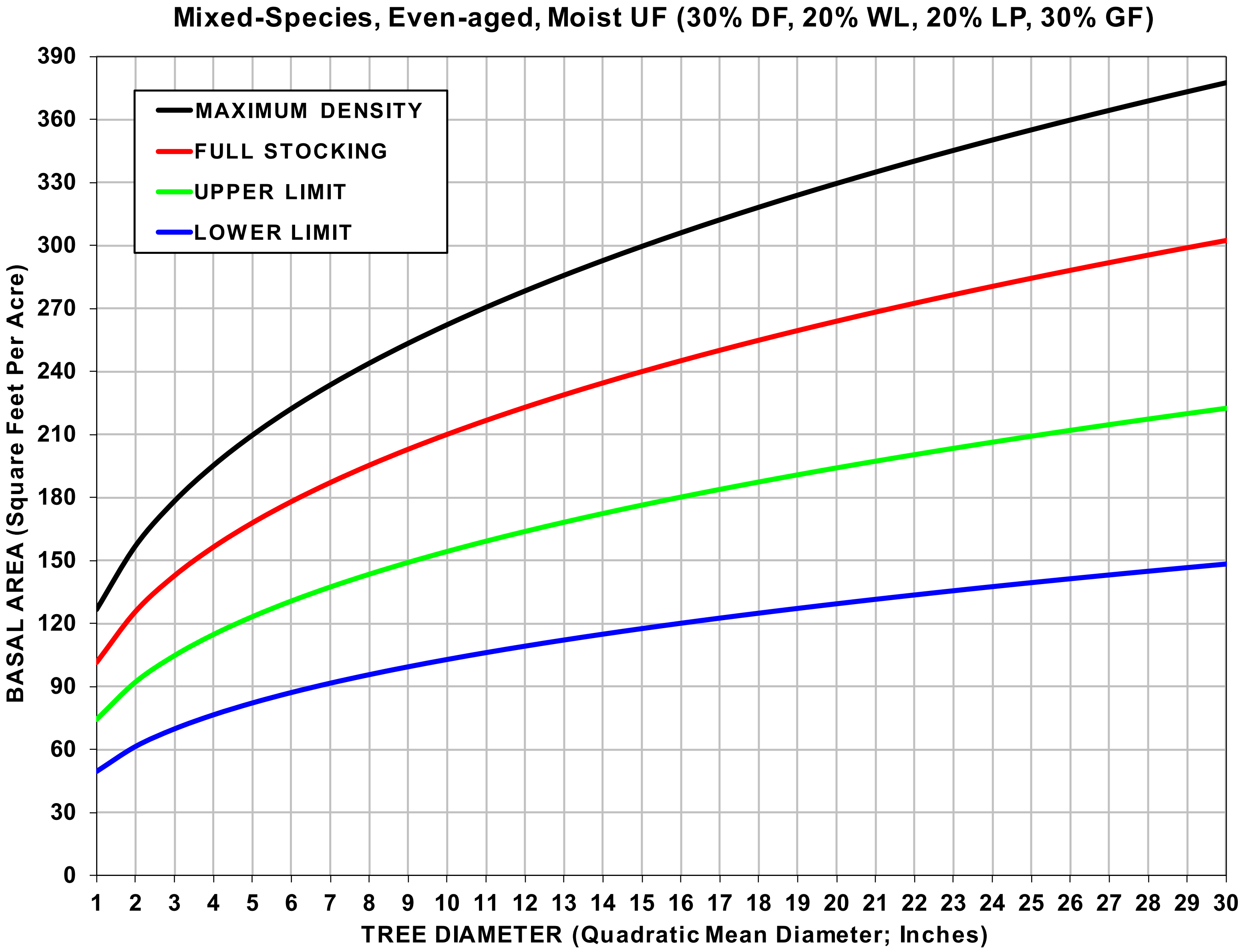


Figure 78a – Stocking chart for moist forest, expressing four stand-density thresholds (color lines) by using basal area and QMD.

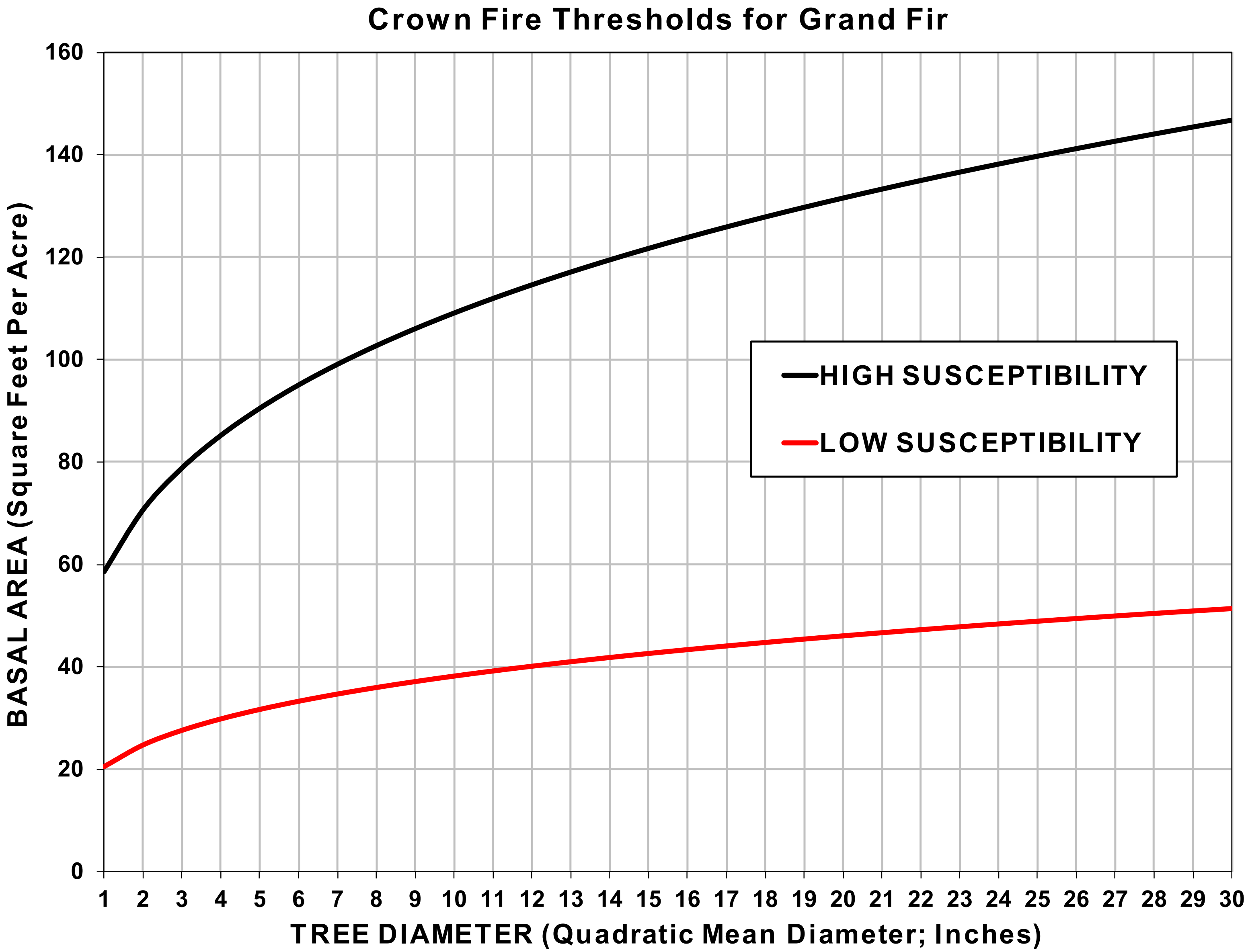


Figure 78b – Stocking chart for moist forest, showing two crown-fire susceptibility thresholds (color lines) by basal area and QMD.

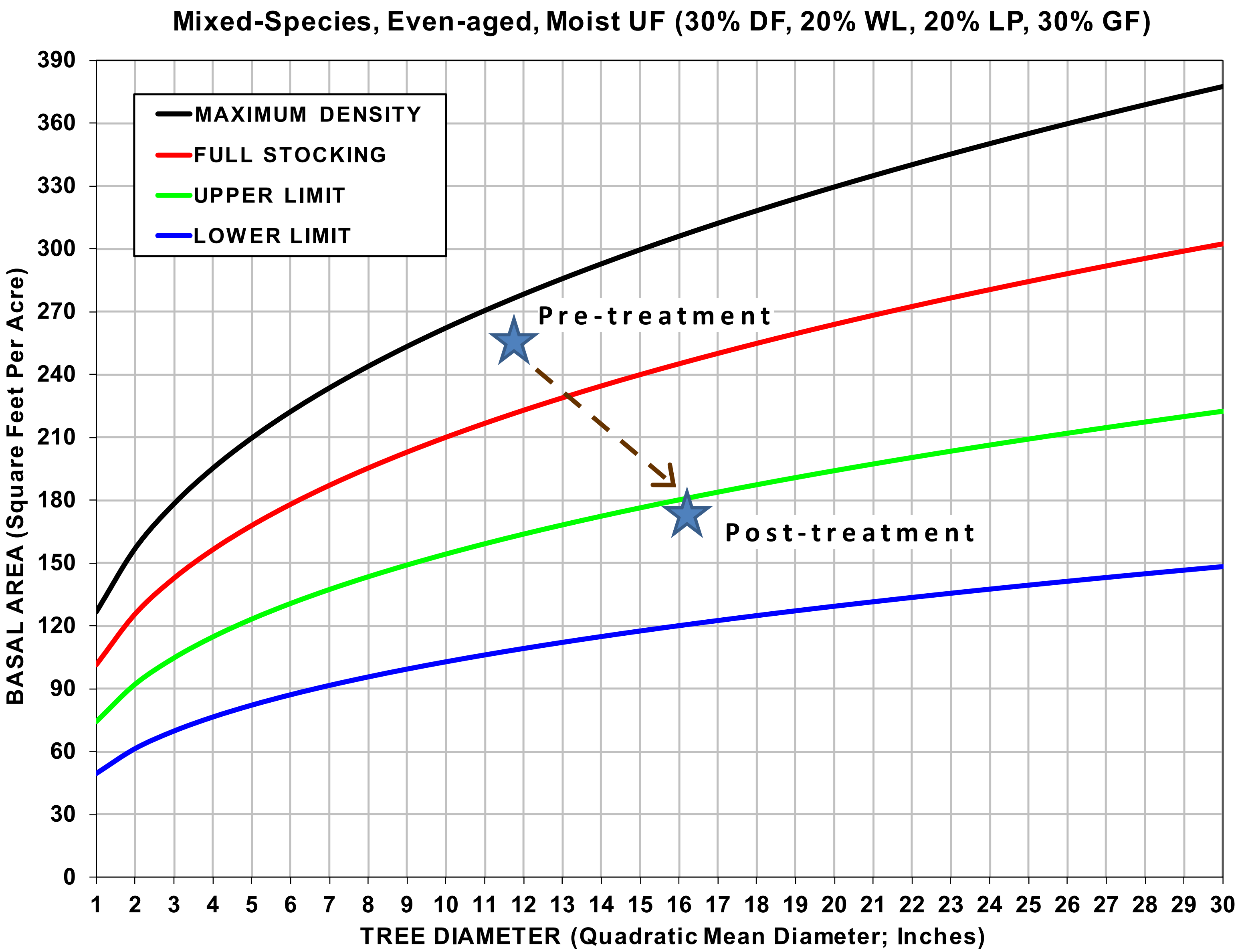


Figure 78c – Stocking chart showing four stand-density thresholds (color lines), and comparing pre- and post-treatment conditions.

Table 24: Recommended stocking levels for moist upland forest, expressed by using the stand development zones depicted in figure 77.

	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]	0-100	0-54	100-124	54-68	124-186	68-101	186-320	68-175	320-399	175-218
Douglas-fir	0-105	0-57	105-168	57-92	168-252	92-137	252-336	137-183	336-419	183-229
Western larch	0-116	0-63	116-185	63-101	185-278	101-152	278-370	152-202	370-462	202-252
Lodgepole pine[1]	0-84	0-46	84-114	46-62	114-170	62-93	170-268	93-146	268-335	146-183
Engelmann spruce	0-125	0-68	125-200	68-109	200-299	109-163	299-398	163-217	398-498	217-272
Grand fir	0-163	0-89	163-261	89-142	261-392	142-214	392-522	214-285	522-652	285-356
Subalpine fir	0-106	0-58	106-170	58-93	170-255	93-139	255-340	139-185	340-425	185-232
Mixed composition[1]	0-120	0-66	120-189	66-103	189-283	103-154	283-385	154-210	385-481	210-262

Notes: Stocking levels are means for 19 plant associations assigned to the moist upland forest potential vegetation group (PVG), and for which stocking-level information is available (e.g., Cochran et al. 1994, Powell 1999). 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 with a 10-inch quadratic mean diameter (QMD). Stocking levels would differ from those shown here when QMD is something other than 10 inches.

[1] For ponderosa and lodgepole pines, upper limits of the '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 values for ponderosa pine (399) and lodgepole pine (335). For mixed composition, stocking levels represent weighted averages (30% Douglas-fir, 20% western larch, 20% lodgepole pine, and 30% grand fir).

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

- The 'climate change' category is equivalent to 'free growth' as depicted in figure 77. Much climate change research suggests the western United States will be substantially warmer and dryer in the future as climate changes, so climate-change stocking levels are lower than conventional levels shown to the right of them.
- The 'high tree growth' zone spans 25 to 35 percent of maximum density. The 25% value corresponds to the 'onset of intertree competition' stand development threshold; the 35% value is the 'lower limit of full site occupancy' development threshold (fig. 77). For the Blue Mountains, the 'lower limit of full site occupancy' threshold is used as the 'lower limit of a management zone' stocking level (see fig. 76, and figs. 78a and 78c) (Cochran et al. 1994, Powell 1999).

- The ‘high stand growth’ zone spans 35 to 60 percent of maximum density. The 60% value corresponds to the ‘lower limit of self-thinning zone’ stand development threshold (fig. 77). Note that figure 41 in section 5.10 describes the self-thinning zone, and its lower limit, in more detail than is provided here, or in the explanatory notes section preceding figure 76. The lower limit of self-thinning zone stand development threshold is used as the ‘upper limit of a management zone’ stocking level (see fig. 76, and figs. 78a and 78c) (Cochran et al. 1994, Powell 1999).
- The ‘low-moderate mortality’ zone spans 60 to 80 percent of maximum density. The 80% value corresponds to the ‘normal density’ stand development threshold (fig. 77), which is used as the ‘full stocking’ stocking level (figs. 78a and 78c).
- The ‘high mortality’ zone spans 80 to 100 percent of maximum density. The 100% value corresponds with maximum density (figs. 77, 78a, and 78c).

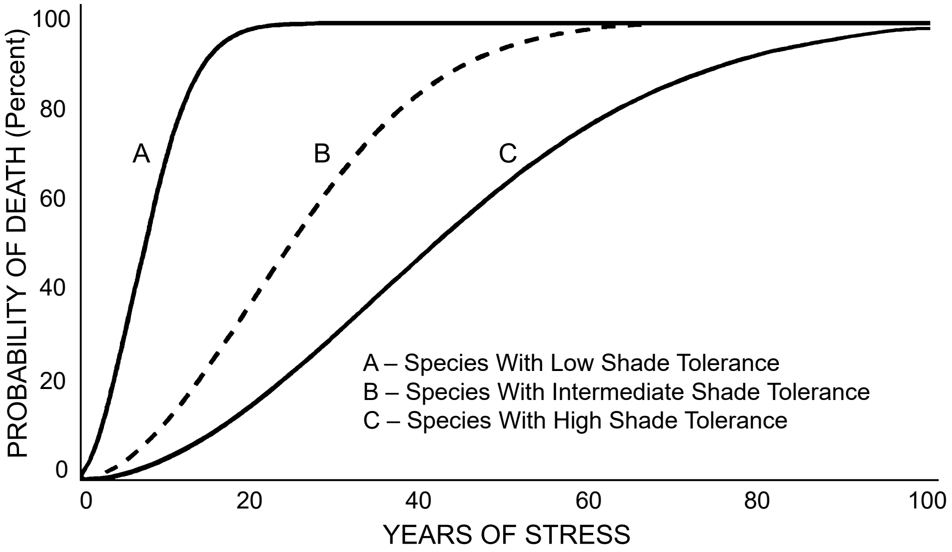


Figure 79 – Tree resistance to stress varies with shade tolerance (adapted from Keane et al. 1996). Tolerance refers to a plant's ability to withstand the effects of one or more limiting factors such as unusually high or low temperatures, a deficit of soil moisture during the growing season, or environments with a deficiency of sunlight or nutrients. In forestry, perhaps the tolerance that has received the most emphasis is shade tolerance – the capacity of a species to survive and grow in the shade of other trees (Harlow et al. 1996). Intolerant tree species (lodgepole pine, ponderosa pine, western larch) die relatively quickly when exposed to chronic stress associated with overcrowding (high stand density), drought, root disease, and other factors. 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 stress periods before experiencing mortality. These tolerance relationships are often considered when prescribing thinning treatments – some proportion of the species with intermediate and high shade tolerance are targeted for removal in order to improve the vigor and resistance of species with low tolerance.

Active Management For Fire Resilience

"Montane forests (see fig. 3, section 2) may also be candidates for active management to create more fire resilient stands. The current stand condition is primed for high severity fire. As with the dry forest, projected climate change will elevate the risks of both drought stress and insect attack, which in turn elevate fire risks. Fire risks in the montane forest may be even more closely tied to climate change than in the dry forest. At montane elevations, snowpack is a critical source of moisture for the growing season. Consequently, expected shifts in the form of winter precipitation from snow to rain [see fig. 65] may dramatically affect levels of summer drought stress and fuel moisture in montane forests. However, the mixed severity disturbance regime of these forests, and the lack of clarity as to the main drivers of change in species composition, mean that there is still uncertainty as to what management actions would increase ecosystem resilience" (Haugo et al. 2010).

For the dryer end of the mixed-severity fire regime (e.g., fire regime IIIa – see fig. 44), which is analogous to the montane forest zone referred to above in the Haugo et al. (2010) quote, "prescribed fire is an attractive alternative to large, high-intensity wildfires, because it is thought to best emulate the natural process that it is designed to replace (Schwilk et al. 2009)" (Stephens et al. 2012, p. 549). This approach also supports resilience to climate change by maintaining natural disturbance regimes and protecting primary forests, especially since research suggests that moist-forest vegetation will likely shift and contract in response to warming temperatures and increased wildfires, and that current fire suppression policies may be less effective in the future at limiting burned-area extent and fire frequency (Boisvenue and Running 2010, Dale et al. 2001, Flannigan et al. 2000, Littell et al. 2009, Loehman et al. 2011, McKenzie et al. 2004, Perry et al. 2011, Westerling et al. 2006).

Current ecological conditions for forests of the interior Pacific Northwest suggest that active management may be warranted. 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). Using a variety of cutting patterns is important to avoid uniform landscapes; grouping harvest areas reduces the total amount of edge, minimizes fragmentation, and maintains larger patches of older forest.

Negative policy implications of not being able to complete active management across wide areas are increasingly being recognized: "with less than 20% of the Sierra Nevada's forested landscape receiving needed fuels treatments, and the need to frequently re-treat many areas, the current pattern and scale of fuels reduction is unlikely to ever significantly advance restoration efforts. One means of changing current practices is to concentrate large-scale fuels reduction efforts and then move treated areas out of fire suppression [and] into fire maintenance. A fundamental change in the scale and objectives of fuels treatments is needed to emphasize treating entire firesheds and restoring ecosystem processes. As fuel loads increase, rural home construction expands, and budgets decline, delays in implementation will only make it more difficult to expand the use of managed fire. Without proactively addressing some of these conditions, the status quo will relegate many ecologically important areas (including sensitive species habitat) to continued degradation from either no fire or wildfire burning at high severity" (North et al. 2012, p. 393).

Concerns about sensitive-species habitat, particularly in the context of whether it experiences too much fire or not enough (as mentioned above in the North et al. 2012 quote), is shared by many other investigators. Wallin et al. (1996), for example, suggest that a management system replicating historical fire return intervals and spatial fire extent may be best for "producing conditions suitable for most, if not all, species…present at the time of European settlement."

Effective active management must consider the landscape context in which it occurs – Finney et al. (2007) compared the effectiveness of different rates of fuels treatment over several decades in the western U.S., and they found that treatment rates beyond 2% of the landscape per year, based on optimized treatment placement (such as strategic placement of landscape area treatments), yielded little additional benefit (Stephens et al. 2012). And fuel treatment rates as low as 2% annually are unlikely to have undesirable impacts because "most available evidence suggests that fuel-reduction objectives are typically accomplished with few unintended consequences, because most ecosystem components (vegetation, soils, wildlife, bark beetles, carbon sequestration) exhibit very subtle effects or no measurable effects at all" (Stephens et al. 2012, p. 558).

The evidence presented in this white paper suggests that although we can't predict when wildfire will occur, we can proactively use our tools, experience, and knowledge to evaluate how fuels treatment in a forested environment would influence both fire behavior and fire effects, while also incorporating predictions about climate change into simulation modeling scenarios examining future fire risk.

Active Management For Postfire Recovery

High fuel loading after a fire, and high potential for a reburn, are characteristic features of the disturbance regime for moist-forest sites. A common proposed action for postfire timber recovery is salvage harvest, an activity designed to remove some of the fire-killed trees, particularly for situations where maintenance of high postfire fuel loading may be incompatible with future fire risk and other land management objectives (Everett et al. 1996). Removing fire-killed trees from a burn could conceivably contribute to socioeconomic and other objectives, but recent experience on the Umatilla National Forest suggests that salvage harvest after wildfire is most assuredly a highly controversial activity.

A salvage effort following wildfire on moist-forest sites, if considered, should address the following vegetation concerns (Schmiegelow et al. 2006):

1. Emphasize salvage for ecological settings in the moist-forest zone with the shortest fire-return intervals (these are fire regime IIIa areas; see fig. 44). Sites meeting this criterion would address changes in species composition and stand density on warmer, moist-forest environments. In some circumstances, it may also be appropriate to consider limited amounts of salvage for riparian zones traversing fire regime IIIa sites (Reeves et al. 2006).
2. Evaluate appropriate amounts of salvage harvest, after providing for proper spatial and temporal distribution of snags and dead-tree habitat across a planning area (DeLong and Kessler 2000), for scenarios where timber volume, tree size, and species characteristics could generate sufficient revenue to finance tree planting and associated restoration activities. This recommendation accounts for the following facts and budgetary realities:
    (a) Maintaining forest ecosystems and their valuable ecosystem goods and services, as forest, is most important (Breshears et al. 2011, Ciccarese et al. 2012, Daily et al. 1997, Ford et al. 2011). The National Forest Management Act of 1976 (16 U.S.C. 1604) also requires that forest be maintained as forest.

  - (b) Treatments such as tree planting to maintain or reestablish a forest condition are expensive.
  - (c) The United States Congress, or reforestation partnership groups such as American Forests or Arbor Day Foundation, may not be willing to, or capable of, financing all of the active restoration activities that may be needed, in which case K-V funds could be withheld from the timber sale receipts and then used to help pay for the work.
  - (d) Salvage harvest can provide socioeconomic benefits by producing wood products and associated employment opportunities.

3. Consider salvage for sites where the existing density of dead trees is great enough that a future reburn would probably reduce the stocking level of postfire tree regeneration below Forest Plan minimum stocking standards.
   [When completing this evaluation, it is important to put the first fire in proper context: did it exhibit characteristic spatial extents, fire effects, and fire behavior? If not, then it might be important to identify portions of the original fire that would have been expected to experience stand-replacing severity. Although pattern follows process, process also follows pattern (i.e., vegetation conditions created by one disturbance event can exert a strong influence on the next disturbance event), so a stand-replacing fire patch is more likely to experience a stand-replacing reburn than if it had burned originally with non-replacement severity. After identifying the portions of a fire where replacement fire severity is characteristic, then it is reasonable to allocate other portions to a planting prescription since it is less likely they would be expected to experience a severe reburn.]
4. Formulate an appropriate response for the following groups of fire-damaged trees, all of which are unlikely to survive more than a few years after fire:
   - a. Ponderosa pines and western larches that have less than 20 percent green, healthy-appearing crown (by crown volume), regardless of bole scorch, scorch height, or duff consumption.
   - b. Douglas-firs having less than 40 percent green, healthy-appearing crown (by volume) AND scorch height greater than 16 feet AND more than 50% of the preburn duff around the base of the tree was consumed by the fire.
   - c. Subalpine firs, grand firs, lodgepole pines, and Engelmann spruces with less than 60 percent green, healthy-appearing crowns (by volume) AND bole scorch on greater than 50% of the tree's circumference AND scorch height greater than 4 feet AND more than 25% of the preburn duff around the base of the tree was consumed by the fire.

Reducing fuels in the high-severity portion of a mixed-severity fire could facilitate reforestation success if another wildfire (reburn) occurs during the next 10-30 years (US Congress 2004). Reforestation involves outplanting trees in areas that were previously forested, but on which the trees were killed or removed by disturbance processes such as wildfire and timber harvest. In some situations, salvage harvest could be considered where timber volume, tree size, and species characteristics would generate sufficient revenue to finance tree planting and associated restoration treatments.

And there is nothing inherently 'wrong' with harvesting some fire-killed timber – doing so provides socioeconomic benefits related to wood products utilization, while also generating trust funds for post-sale reforestation (trust funds are retained from timber sale receipts to finance post-sale work). Trust funds are valuable because Congress may not fund all of the reforestation work on a national forest, especially considering recent concerns about federal deficits and the national debt.

For burned areas where fire-killed trees are not salvaged, the National Forest Management Act (16 U.S.C. 1604) does not require that reforestation occur, either within a 5-year timeframe or at all. Even so, the United States Congress and the U.S. Forest Service are interested in reforesting many of the burned areas promptly (Watrud et al. 2012), particularly when tree planting could attain a Forest Plan desired future condition more quickly than by waiting for natural succession to reestablish appropriate forest cover. In some situations, salvage timber harvest could be used to increase the survivability of planted seedlings in the eventuality of a future fire.

Tree planting is a powerful tool for influencing the future species composition of a forest (fig. 80). In order to address future susceptibility to defoliating insects and other forest health considerations, and to select a composition ecologically appropriate for the open environments typically produced by high-severity fire effects, tree planting would typically attempt to establish a future composition with at least 60 percent of the trees being early- or mid-seral species (Carlson and Wulf 1989).

Establishment of early-seral species is also emphasized in the Umatilla NF Forest Plan, as demonstrated by item C in the Forest-wide standards and guidelines section on page 4-73: "strong consideration should be given to maintenance of stands dominated by early successional species" (USDA Forest Service 1990). Tree planting recommendations for moist forests are provided in table 25.

Active Management To Reduce Crown Fire Susceptibility

Following establishment of the National Fire Plan in 2000, there has been increasing emphasis on hazardous fuels reduction treatments. Much of the fuels reduction work in forested ecosystems has emphasized addressing crown fire potential. "If the negative impacts that result from crown fires were fully reflected in the market, there would be high motivation to avoid them, providing necessary incentive to remove excessive fuel loads in spite of the cost" (Mason et al. 2006).

Stands with high levels of canopy fuel loading (also referred to as crown or canopy bulk density, or canopy biomass) are most susceptible to sustaining an active (independent) crown fire. "Crown fire susceptibility refers to the potential for crown fire based on inherent stand characteristics such as species composition, forest structure, and tree density" (Powell 2010).

Figure 80 – Active management of a budworm-affected area. These photographs show Battle Mountain state park, located about 10 miles north of Ukiah, Oregon. This area experienced high amounts of tree damage (mortality and topkilling) during a 1980-1992 western spruce budworm outbreak. Top photograph (acquired June 2005) shows the area soon after state of Oregon salvaged budworm-killed trees, and then planted ponderosa pine seedlings (if you look closely, you can see white vexar tubes installed to protect seedlings from ungulate browsing damage). Bottom picture (acquired October 2012) shows the area after regeneration had time to develop. Tree planting is a powerful tool for directing future development of a forest ecosystem, and it deserves more consideration than it is often afforded.

Table 25: Planting recommendations for the moist upland forest potential vegetation group.

	Seedling Density[2]		Species Composition of Planting Mix (Percent)[3]							
Plant Association Group[1]	**TPA**	**Spacing**	**PP**	**WL**	**LP**	**DF**	**WP**	**GF**	**ES**	**SF**
Cool moist upland forest	222	14 feet	25%	40%	NR	20%	10%	NR	5%	NR
Cool very moist upland forest	222	14 feet		20%	NR	30%	10%	NR	40%	
Cool wet upland forest	222	14 feet	NR	20%		30%	10%	NR	40%	
Warm moist upland forest	194	15 feet	60%	NR		40%				
Warm very moist upland forest	194	15 feet	20%	30%		40%	10%	NR	NR	

[1] Moist upland forest potential vegetation group includes five plant association groups (PAGs), as shown here. When considering the number of potential vegetation types (PVTs) assigned to each PAG, cool moist upland forest is by far the largest and most common of moist-forest PAGs. Powell et al. (2007) provides information about PVT composition for each PAG.

[2] Seedling density recommendations are expressed as both a trees per acre (TPA) figure and its corresponding square-spacing value, in feet. They are based on the authors' judgment, and on minimum seedling stocking standards from the Umatilla NF Forest Plan. Seedling density values also account for the fact that monitoring results show natural regeneration to be both copious and diverse for moist forests (see appendix 2 of this white paper).

[3] Species composition of planting mix recommendations are based on Cole (1993), Powell (1997), tables 10, 15, 16, and 18 in this white paper, and professional judgment. Column heading codes are: PP: ponderosa pine; WL: western larch; LP: lodgepole pine; DF: Douglas-fir; WP: western white pine; GF: grand fir; ES: Engelmann spruce; SF: subalpine fir. NR = Natural Regeneration, showing tree species expected to establish without planting; these species were not included in the planting mix, but they could be used if seed sources for recommended species are in short supply. The potential vegetation type composition for each plant association group is provided in appendix 1.

Research following large wildfires in the interior Pacific Northwest shows that stands where canopy fuel loading has been reduced below 0.10 kg/m^3 are unlikely to sustain active crown fire; stands where canopy fuel loading has been reduced to 0.05 kg/m^3 or less are basically immune to crown fire, even under high wind conditions (Agee 1996c, Keyes and O'Hara 2002).

Since canopy fuel loading is virtually impossible to measure directly, because this would involve physically measuring all of the foliage and small twigs or branches in a tree crown, and then summing results for all of the individual trees to obtain a stand loading, a process was developed to relate canopy fuel loading (as kg of plant material per m^3 of canopy volume) to conventional stand density metrics such as stand density index, trees per acre, basal area per acre, equilateral tree spacing, and canopy cover (crown closure). These stand density measures were calculated for three levels of crown fire susceptibility – high, moderate, and low, as defined using three ranges of canopy fuel loading in kg/m^3 (Powell 2010).

But we would not expect all wildlands to have low canopy fuel loading all of the time, so range of variation information was developed to characterize how much canopy fuel loading existed historically by fire regime (table 26). Although the ranges are somewhat wide, table 26 suggests that no more than 50% of a moist-forest landscape had high canopy fuel loading at any given time.

Table 26: Range of variation information for canopy fuel loading classes (kg/m³), expressed as percentages.

Potential Vegetation Group	Fire Regime[2]	CANOPY FUEL LOADING CLASS[1] Low (≤.05 kg/m³ CBD)	 Moderate (.06-.09 kg/m³ CBD)	 High (≥.10 kg/m³ CBD)
		Range of Variation (Percentage)		
Dry Upland Forest	I	60-90	20-60	10-20
Moist Upland Forest	III	20-50	50-70	20-50
Cold Upland Forest	IV	10-20	20-60	60-90

Source/Notes: Based on Agee (1998). Potential vegetation group is described in Powell et al. (2007).

[1] Canopy fuel loading class is a derived data element; it can be calculated using queries contained in Powell (2010). CBD is crown bulk density, expressed as kilograms per cubic meter of crown volume. Class breakpoints are as follows: .05 kg/m³ = CBD threshold below which crown fire is unlikely; .10 kg/m³ = CBD threshold above which crown fire is easily sustained (Powell 2010).

[2] Fire regime describes the fire environment by characterizing fire frequency, fire intensity, fire severity, fire extent, fire timing, and historical burned area (Schmidt et al. 2002). For forest environments in the Blue Mountains, three fire regimes are most important: Fire regime I: surface; Fire regime III: mixed; Fire regime IV: replacement.

Landscape Wildfire Strategy

For some parts of the landscape, including wilderness and roadless areas, current policy does not allow fire risk to be addressed by implementing mechanical treatments (see fig. 88 later in this white paper). For these situations, it may be appropriate to formulate a landscape wildfire strategy that would:

1. Identify managed or developed portions of the landscape that need to be ‘protected’ from high-intensity fire moving into them from adjacent areas.
2. Identify roadless or wilderness portions of a landscape that need to be ‘protected’ from management-ignited prescribed fire moving into them (but these areas might still be allowed to experience wildfire originating from natural ignitions).

For either of the situations described above, a shaded fuelbreak approach (Agee et al. 2000) might be appropriate for setting up a landscape to foster differing levels of fire response and containment. But what size of a fuelbreak might be most effective or appropriate? “Considering fire spread rates under extreme conditions (up to 3 km/h; app. 1.9 mi/h), fire fighter response times (>10 min, even in urban forests), and other complicating factors, 400-500 m (app. ¼ mile) is probably a justifiable minimum width for fuel treatments expected to significantly slow or stop wildfire. Of course, this calculation ignores ember production” (Safford et al. 2012, p. 27).

Note that fuelbreaks have traditionally been deployed in support of fire suppression objectives (Omi 1996). But in a contemporary context, fuelbreaks can contribute to many of our landscape wildfire objectives (Ingalsbee 2005) by supporting a ‘containerized’ approach for managed wildfire (Ager et al. 2012). In this current and expanding era of ‘mega-fires’ (and fig. 66 suggests mega-fires will become more common in the future), we eventually must acknowledge that many conventional wildfire suppression strategies are running out of road and will not be sustainable from either an ecological or financial perspective (Donovan and Brown 2005, 2008). This eventuality suggests that a managed wildfire paradigm will get established, and that we need to incorporate the reality of mega-fires into land management, policy, and planning frameworks (North et al. 2012, Stephens et al. 2014, Williams 2013).

One key to successful stewardship of moist forest ecosystems is to apply site-specific management. Site-specific management requires detailed knowledge of soils, potential natural vegetation, and other biophysical components, along with an understanding of how physical, chemical, and biological characteristics affect sustainability of ecological settings in which moist forests occur. Inherent in a concept of site-specific management is recognition that a similar or 'standard' level of management intensity is not appropriate for all areas or situations (Fox 2000).

Effective restoration and mitigation tactics need to be site-specific because a single prescription for moist forests, and their complex mixed-severity disturbance regimes and mixed-species compositions, would not be appropriate (Hessburg et al. 2007, Schoennagel et al. 2004). And any effective moist-forest management strategy should adopt an ecological perspective viewing humans as nested within larger cultural and ecological systems, and who through their collective actions have the potential to help restore processes integral to proper ecosystem function.

Site-specific management of moist forest can be greatly assisted by treatment prioritization. Although many different prioritization strategies have merit, one worth considering is a triad approach, as described by Aplet and Wilmer (2010) and directed toward human values at risk. Their approach identifies three levels of decreasing risk to human life and property:

1. First treatment priority are areas where fire and other disturbance processes have potential for causing great damage to people and associated infrastructure (homes, outbuildings, electrical transmission facilities, etc.) (North et al. 2012). This category is similar to what is referred to as Wildland-Urban Interface (WUI), a mixture of homes and undeveloped wildlands.
2. Second priority includes areas in relatively close proximity to human developments, but are far enough removed from infrastructure to allow some flexibility for fire and other processes to play a more natural role, at least under certain (prescribed) conditions.
3. Third and final zone contains wildlands distant enough from communities and infrastructure to allow wildfire and other processes to play a natural role under a relatively wide range of circumstances; direct suppression of fire or defoliator outbreaks, for example, would occur infrequently in this zone. In this zone outside of WUI, fuel treatments "should focus on creating conditions in which fire can occur without devastating consequences" (Reinhardt et al. 2008, p. 1998).

This three-pronged approach recognizes that fire suppression will continue as a strategy for protecting urban interface, air quality, certain types of wildlife habitat, and other values at risk, but continued suppression will certainly not prevent increases in area burned because it is the few large fires (ones escaping suppression) that actually affect the most acreage (Strauss et al. 1989).

A triad approach explicitly acknowledges that fire suppression occurs along a continuum – full suppression actions should be initiated quickly when fire threatens homes or other values-at-risk, whereas remote fires may receive no management intervention beyond frequent monitoring. But the middle of this continuum is particularly interesting – a fire in this zone would be managed by using predetermined boundaries (both geographic and weather-related), with suppression occurring as fire approaches a boundary, but a fire would not be attacked aggressively when burning well away from boundaries. And with this approach, a single large fire may be managed adaptively, with active suppression occurring on one flank while another portion is allowed to burn freely for resource benefit (Dale 2006).

"Fire suppression can be relaxed in remote areas, letting unplanned ignitions burn under the observation of fire crews, even for weeks or months in summer, so that alternating smolder-and-run fire behavior develops in phase with a broad range of weather conditions, promoting heterogeneous burn severities over the landscape as reported in numerous pre-suppression accounts" (Goforth and Minnich 2008).

Letting fires burn, a central component of the managed wildfire paradigm, introduces many complexities and concerns, particularly for national forests in the Blue Mountains. Letting fires roam through large Wilderness areas in remote portions of the United States is an obvious and attractive alternative. But managed wildfire for areas like the Blue Mountains, where Wilderness and Roadless areas are relatively small and often adjacent to human developments, is more problematic.

A dominant wildfire policy for more than 100 years can be summarized by using one word – suppression (Pyne 1997). But in a contemporary context, "the traditional suppression focus is seen as more likely to put firefighters at risk and misuse resources while losing opportunities to reintroduce fire. The use of a wider range of strategies beyond full suppression is seen as leading to more cost-effective fire management over time and longer-term land-management benefits" (Steelman and McCaffrey 2011). "Although we understand that the issue is not a simple dichotomy between fire suppression and treatment, we argue that waiting for a complete understanding of the social and ecological complexities before taking action is folly" (Snider et al. 2006).

And with climate change, the acreage burned by wildfire in the Pacific Northwest is expected to double or triple (or even more when compared with a 1950-2003 baseline) by the 2080s (Littell et al. 2010, and see fig. 66) – this projected increase in acreage burned leads to even more uncertainty about the wisdom of a 'continued fire suppression' strategy for western wildlands (except, as described above, for specific values-at-risk, which will continue to merit a suppression response).

But, the flip side of this coin is that some managers or policy makers could view fuels treatment as even more important in the future (rather than pointless in an environment where fires are two, three, or even six times more common than now) because it is then more likely that fires will actually encounter a treated area, and "there is a strong positive relationship between the extent of fire in the landscape and the effectiveness of treatment" (Price 2012).

Young Stand Development

From the 1960s to late 1980s, much regeneration cutting occurred by implementing clearcutting, seed-tree, or shelterwood methods on moist upland forest sites (see section 5.13 and fig. 14). Prompt and generally effective reforestation treatments (appendix 2) contributed to establishment of dense, mixed-conifer forest on many harvested areas (Rollins 1982) (fig. 81).

But not all reforestation treatments were successful, however, and some moist-forest cuttings between Jubilee Lake and Lookout Mountain (such as Little Big Hole timber sale area on Walla Walla Ranger District, see: Ferguson et al. 2005) transitioned to a post-harvest plant community dominated by bracken fern or western coneflower instead of conifers (Box 7, fig. 82, Ferguson et al. 2005).

Many moist-forest plantations were established with a high proportion of what is termed 'off-site' planting stock (Dahl and Nicholson 1970):

1. **Ecologically off-site species**: this category refers to situations where an early-seral species adapted to dry sites (such as ponderosa pine) is planted on moist sites. Even if ponderosa pine seedlings were produced from local seed sources, they were planted on sites with biophysical conditions beyond the ecological amplitude for this species (such as subalpine fir plant associations), or they were established at a much higher density than is ecologically appropriate for a particular plant association (such as out-planting 600 ponderosa pine seedlings per acre on a moist grand fir/queencup beadlily plant association).
   [This example is counter-intuitive to readers who expect that a dry-site species would grow even better on moist sites, but many studies have shown that dry-site species do not perform well when planted on moist or cold sites – an example is Daubenmire's (1950) study about geographic races of ponderosa pine.]
2. **Genetically off-site sources**: this category refers to seedlings produced from non-local seed sources. According to Gary Rollins (Rollins 1982), long-time District Silviculturist for Walla Walla Ranger District, many ponderosa pine seedlings used in moist-forest clearcuts on Umatilla National Forest were derived from non-local sources (primarily from Malheur NF and Wallowa-Whitman NF sources; at least they weren't from Black Hills NF sources used across the West).

As a consequence of off-site plantings, many moist-forest plantations have not developed as would be expected if they supported an ecologically appropriate species composition derived from local seed sources. Tree-level indicators of impaired development include: stem deformities (sweep, crook, fork); high levels of snow breakage; increased frost damage; increased infestation by insects or diseases, particularly by unusual or atypical agents; slow growth when compared with native sources on the same site; dead terminal leaders; 'cropped' or stunted or off-color foliage appearance; and abnormal or 'stress' cone crops.

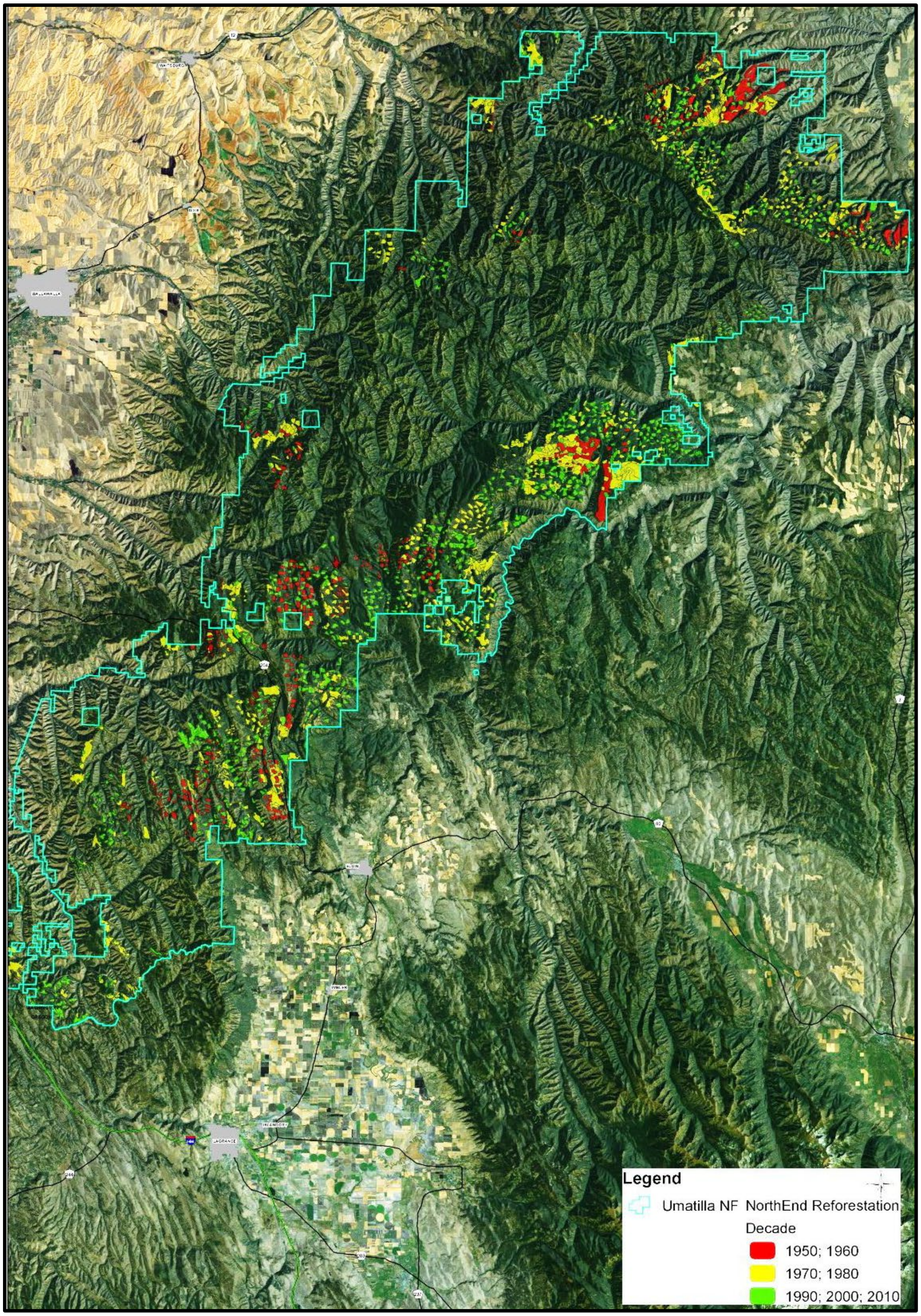


Figure 81 – Moist-forest plantations established between the 1950s and 2010 in the northern portion of the Umatilla National Forest (Pomeroy and Walla Walla Ranger Districts). This map shows that more than 96,000 acres of young stands were created on the northern half of the Umatilla NF since the 1950s. Many of these stands now need to be thinned, or modified in other ways, to ensure they continue to develop normally and provide the socioeconomic goods and services desired by human society.

Box 7. The Grand Fir Mosaic Ecosystem

Imagine a moist environment in northern Blue Mountains where supposedly early-seral plant communities dominated by bracken fern (*Pteridium aquilinum*) and western coneflower (*Rudbeckia occidentalis*) are actually climax plant communities persisting in a matrix of overmature grand fir forests. Wildfire is rare here, and long-term succession results in poor representation of early- and mid-seral conifers such as lodgepole pine, Douglas-fir, western white pine, and western larch. Late-seral, shade-tolerant species like grand fir, Pacific yew, and occasionally subalpine fir, are common, along with mid-seral Engelmann spruce. Conifer regeneration in forest canopy openings is slow and unreliable in these low pH, volcanic ash-cap soils with abundant populations of pocket gophers (*Thomomys talpoides*) (Ferguson et al. 2005). Disjunct and rare plant species often occur in and near these forests. These areas, collectively called the Grand Fir Mosaic (GFM) ecosystem, are typically associated with grand fir/ginger plant associations, not just for the Blue Mountains but in central and northern Idaho (Green and Jensen 1991).

GFM forests occur on productive volcanic ash-cap soils in and near Clearwater, Nez Perce, and southern St. Joe National Forests in northern and central Idaho, and Umatilla National Forest in northeastern Oregon (GFM research plots were installed in Little Big Hole timber sale area on Walla Walla RD). GFM name commemorates a dominant conifer (grand fir), and a variety of sizes and shapes of natural openings in forest canopy. GFM encompasses approximately 500,000 acres at elevations between 4,500 and 5,500 feet, but it is found as low as 4,200 feet and as high as 6,000 feet (Ferguson and Johnson 1996). The most common habitat type is *Abies grandis/Asarum caudatum* (grand fir/wild ginger), a cool, moist plant association defined in ecological site classifications (Cooper et al. 1991, Johnson and Clausnitzer 1992).

Successional plant communities in GFM are dominated by bracken fern and western coneflower (fig. 82). Bracken fern is usually present in low densities under forest canopies, but rapidly expands following disturbance, and can reach heights of 6 feet and densities of 116,000 fronds per acre (Znerold 1979). Below-ground bracken biomass, primarily rhizomes and fine roots, may reach 27,280 lbs per acre (Jimenez 2005).

Bracken fern glades, plant communities dominated by bracken fern and western coneflower, appear to persist for millennia. Charcoal samples found at or near a lower boundary of GFM ash-caps were found to be 1,335±75 and 7,755±75 cal. yrs before present (BP) using radiocarbon dating (Jimenez 2005). These samples suggest that woody vegetation has been absent for thousands of years, perhaps since time of ash deposition during eruption of Mt. Mazama (Crater Lake, OR) ~7,600 yrs BP (Zdanowicz et al. 1999).

Pocket gophers also alter the course of secondary succession in GFM, particularly for planted conifers. Entire plantations of seedlings can be killed by pocket gophers (Crouch 1982, Crouch and Frank 1979, Ferguson 1999). Small seedlings are usually pulled from below ground into tunnels where the whole tree is eaten. For larger seedlings and saplings, gophers may eat all or most of the root system.

Research investigations involving GFM were initiated because of difficulty in regenerating harvested areas. An initial task was to define key ecological processes potentially accounting for lack of tree regeneration and other woody species. Research was implemented to study competition and allelopathy from bracken fern and western coneflower, effects of pocket gophers, environmental characteristics of GFM relative to adjacent forests, and soil development (Ferguson et al. 2005).

Note: this material was adapted from Ferguson et al. (2007).

Figure 82 – Two **WATCH OUT!** plants for moist-forest sites. Western coneflower (*Rudbeckia occidentalis*) (above) and bracken fern (*Pteridium aquilinum*) (below) are widespread plants in a moist-forest zone. Both species are allelopathic, and they have been found to suppress conifer regeneration when occurring in large numbers (see Box 7). In northern Blue Mountains, western coneflower is associated with the very moist end of grand fir series, specifically the grand fir/oakfern, grand fir/swordfern-ginger, grand fir/false bugbane, and grand fir/queencup beadlily plant associations (Johnson and Clausnitzer 1992).

Although reasons are not completely clear (but soil chemical changes related to aluminum-humus interactions are suspected; see McDaniel and Hipple 2010), forest management practices creating open conditions on moist-forest sites sometimes promote site domination by coneflower or bracken (seldom do both species occur together), resulting in long-term suppression of conifer regeneration; in other instances on these sites, conifer regeneration gets established promptly, and a monoculture of coneflower or bracken does not result.

As time passed, young stands differentiated into a multi-layered condition: western larch became co-dominant with planted ponderosa pine as an overstory layer, eventually over-topping the pine, while subordinate layers containing Douglas-fir, grand fir, Engelmann spruce, and Pacific yew also got established (appendix 2).

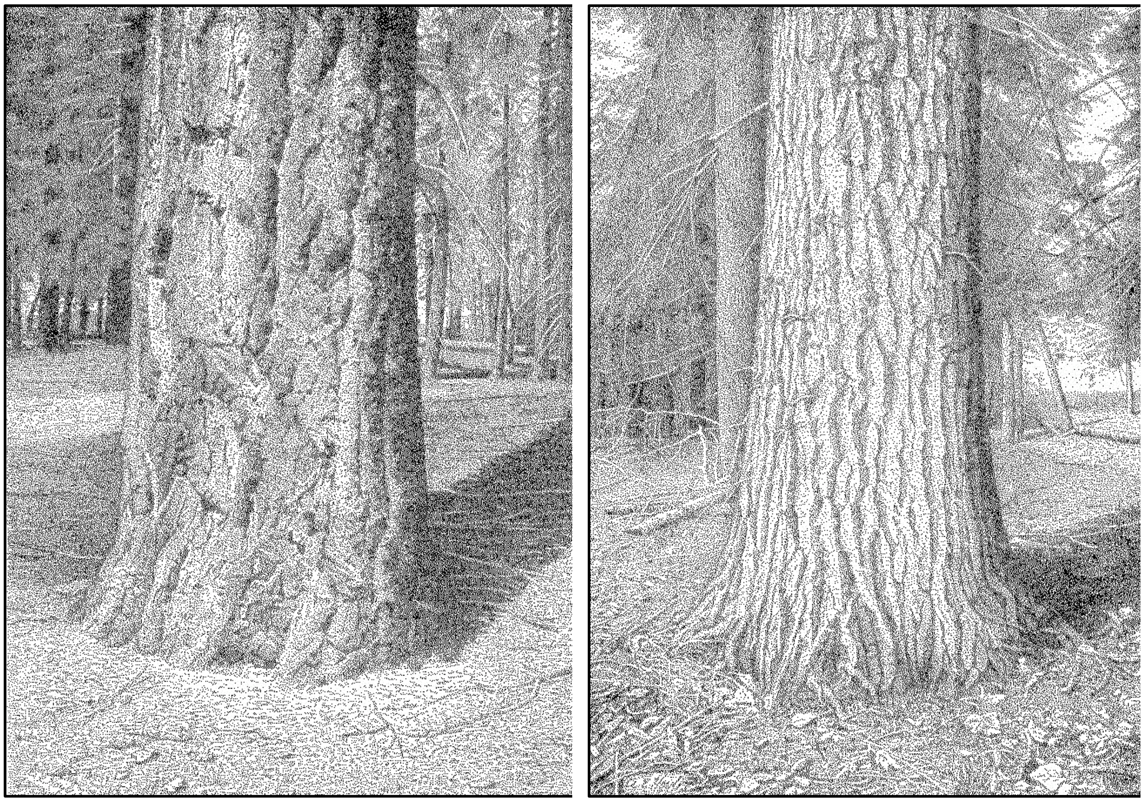
An important management objective for moist forests of northern Blue Mountains is to begin implementing intermediate cutting in these plantations as a way to move them back toward an historically appropriate species composition, with reduced representation of ponderosa pine and increased representation of western larch, western white pine, and Douglas-fir (while maintaining existing successional trends for grand fir, Engelmann spruce, Pacific yew, and Scouler willow) (fig. 83).

Figure 83 – Example of low thinning in a moist mixed-conifer forest, with western larch selected as a primary residual species. Low thinning is defined as removal of trees from lower crown classes or canopy layers to favor trees in upper crown classes or layers. For mixed stands, low thinning can also be used to modify species composition by favoring one species over another. In this example, most residual trees are western larch, a shade-intolerant tree species, and most cut (removed) trees are grand fir and Engelmann spruce, both of which are shade-tolerant species. Thinning is effective for retaining species with low shade tolerance because they will only survive for short periods of intense, competition-induced stress. So, in mixed stands, thinning should target removal of shade-tolerant tree species (fig. 79).

When young stands already have a reasonable proportion of larch, western white pine, and Douglas-fir, intermediate treatments (improvement cuttings and thinnings) will be important in the future to maintain the 'competitive advantage' or 'free-to-grow' status of these species (Jain and Graham 2002, Jain et al. 2004).

Today, a mosaic of young forest patches with heightened fire and insect hazard is interspersed with old-forest patches; thinning in young stands not only accelerates development of large-diameter trees, but it also helps protect old-forest patches from stand-replacing fire or insect outbreaks (Franklin et al. 2002, Swanson et al. 2010). Thinning is also desirable for stands where western white pines with genetic resistance to blister rust could be perpetuated (Jain and Graham 2002). And, finally, thinnings can be used to favor future fire resistance, even for a tree genus, such as *Abies,* with low overall resistance (fig. 84).

A variable-density thinning strategy utilizing skips and gaps could be considered as a strategy for improving or introducing spatial heterogeneity (Aber et al. 2000, Franklin et al. 2007). Treatments introducing gaps into homogeneous plantations can be particularly beneficial from a wildlife perspective (Sallabanks et al. 2002).



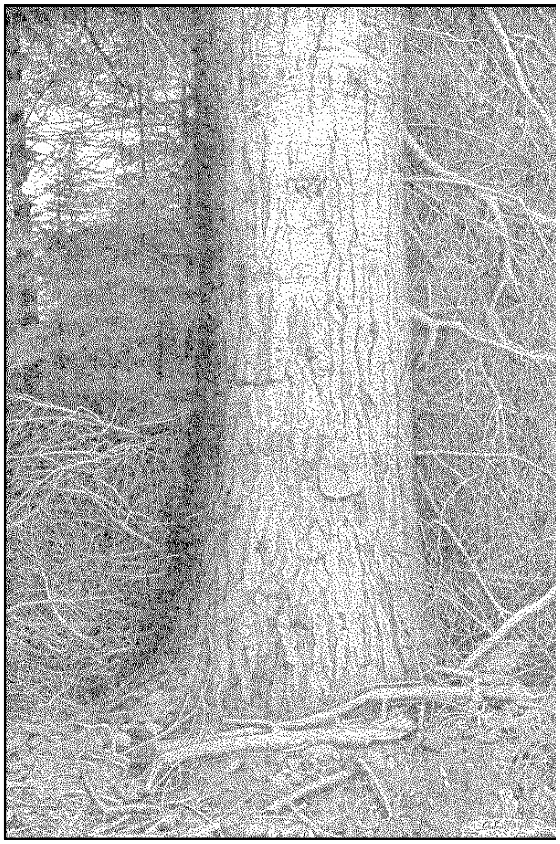


Figure 84 – Bark characteristics for mature examples of white/grand fir (dry sites; upper left), grand fir (moist sites; upper right), and subalpine fir (moist/cold sites, lower right) trees. Upper left image shows deep bark fissures and thick (also corky or spongy) bark plates for dry-site fir of southern Blue Mountains. Upper right image shows shallow bark fissures and narrow, thin bark plates for moist-site grand fir of northern Blue Mountains. Lower right image shows poorly developed bark fissures and plates for subalpine fir of northern Blue Mountains. These images portray morphological differences across a large geographical extent, such as the Blue Mountains, for one species (or hybrid complex) – the two white/grand firs portrayed in upper row, and across the whole *Abies* genus – white/grand firs in upper row compared with subalpine fir of lower right. Images taken from Sudworth (1916).

"The abundance of many bird groups was higher in the gap-dominated than in the continuous forest. Species preferring interior parts of the forest had the most negative association with the presence of gaps but this relationship was not statistically significant. Abundances of many bird groups increased with increasing gap size, while its effect on abundance of some bird groups disappeared quickly. Our review suggests that silvicultural practices that bring about small gaps do not negatively affect the abundances of most forest birds and often even enhance it. However, more studies are needed to examine optimal size and abundance of gaps in a forest and whether emulated small-scale disturbance effectively mimics natural processes" (Forsman et al. 2010).

Other research suggests it's not just gaps that make a difference in terms of avian habitat, particularly for moist grand fir forests of Blue Mountains (Sallabanks et al. 2002, 2006). "That most studies on thinning of coniferous forest have found increased abundance and richness of birds suggests that thinning can provide some positive ecological benefits while increasing timber yields. The dense nature of such stands [unthinned stands dominated by fir or pine species], the similarity of structural forms of individual trees, and low light penetration creates a homogeneous vertical structure with little ground vegetation. Given the importance of structural complexity as a factor influencing habitat selection in birds, it is perhaps not surprising that thinning results in increased bird diversity in coniferous forests" (Bayne and Nielsen 2011, p. 1925).

Thinning For Young Stands (Plantations)

Unlike old forests, young forests change rapidly (Oliver and Larson 1996). Silvicultural intervention can influence speed and direction of this change to accelerate development of desired forest structure, reduce fire risk and, at the same time, produce some utilitarian goods and services desired by society (Keyes 1996). This fact illustrates that silviculture is little more than application of ecological leverage. Thinning, a purposeful application of ecological leverage, is designed to meet a wide variety of management objectives (figs. 76-79, 83, and 85) (Smith et al. 1997). Young stands on moist-forest sites provide many opportunities to apply ecological leverage by influencing future development of composition, structure, and density.

Taking immediate action for young stands is a proactive shift to curtail excessive fire and insect impacts, and a shift away from reactive responses to landscape-scale disturbance. A solution starts with thinnings and understory removals to reduce stand density in overcrowded forests, including young stands (Oliver et al. 1994). **A pessimistic note**: While sound restoration work is being conducted throughout the Blue Mountains to increase terrestrial and aquatic resilience, many indicators suggest the pace and scale of restoration work should be increased dramatically. Lately, it seems as if only a restoration program of unprecedented scope and extent may alter direction of current trends, and it needs to occur quickly to be successful.

Figure 85 – Moist, mixed-conifer forest with western larch (yellow foliage) dominant in the overstory in the Lookout Mountain portion of the northern Blue Mountains. Commercial thinning and other active management treatments could be considered for reducing intertree competition in these mixed stands, and thereby improve the survival potential for larch. For situations where promoting larch is a management objective (such as in response to a range of variation analysis for species composition indicating that larch is under-represented in a moist-forest landscape), two primary approaches are available: favor larch by using improvement cuttings or thinnings to remove competing species for stands where an acceptable component of larch still exists (as is shown here), or complete a regeneration cutting (fig. 76) designed to obtain natural larch regeneration (fig. 86) or create acceptable site conditions for planting larch seedlings.

Note that a successional progression shown here – stands initially containing a predominance of western larch following a stand-initiating disturbance event, but with larch gradually and inexorably being replaced by more shade-tolerant associates such as Engelmann spruce and grand fir – was found to be common across a portion of the interior Columbia River basin coinciding with western larch's natural range (Quigley and Arbelbide 1997). Without active management intervention, which could easily take the form of thinning (instead of a regeneration cutting method) at this point because the proportion of western larch is still relatively high in these successionally advancing stands, then we would expect representation of western larch in this landscape to continue its decline as time passes.

No single silvicultural system, however, can hope to precisely reproduce the inherent variability of a landscape because forests are shaped by a variety of disturbance processes (Voller and Harrison 1998), suggesting that variable-density thinning with skips and gaps, along with aggregated retention, are two approaches worthy of consideration for moist-forest landscapes where intrastand diversity has deteriorated (Franklin et al. 2007).

Figure 86 – Young, natural regeneration of western larch in a regeneration cutting unit on Pomeroy Ranger District (photo provided by Eric Pfeifer). As discussed in fig. 85, western larch is often a common, early-seral tree species in mixed, moist-forest stands. If larch is to persist as a common landscape element, providing Blue Mountain residents with their prime source of autumn color, it is important that these young stands be 'cultured' in such a way as to discourage replacement of western larch by shade-tolerant competitors such as grand fir, Engelmann spruce, subalpine fir, and Douglas-fir.

When trying variable-density thinning with skips and gaps, or similar approaches such as aggregated retention (Franklin et al. 2007) for the first time, consider applying them on a small scale. Be aware of, and understand, the risks being taken, and recognize that mistakes are probably inevitable. Because each implementation is a new learning experience, be creative and innovative to whatever extent the circumstances allow, and remember that we learn best from our mistakes.

<u>The bottom line is</u>: start slowly with a forgiving project area or management context, so the lessons we learn from new approaches like variable-density thinning come with minimal pain and are less likely to foreclose future options for additional learning opportunities.

Some managers are concerned that accelerated thinning of young stands might lead to unacceptable levels of tree wounding and associated stem decay, or perhaps to an increased level of annosus root disease because of the stumps left behind by thinning (annosus spreads by colonizing fresh stumps). Research suggests these concerns might be unfounded, particularly for situations where most of the residual trees are still physiologically young and vigorous.

"In grand fir stands in southern Oregon, artificially inflicted wounds tended to close more rapidly on trees in thinned stands than on trees in unthinned stands. Thus, western conifers that have been thinned and are growing more rapidly appear to develop a smaller percentage of decay than do trees in unthinned stands. Thinning tended to maintain increased tree vigor and diameter growth several years after treatment for stands of grand fir (after 16 yrs), ponderosa pine (after 16 yrs), and lodgepole pine (after 10 yrs). Small-diameter stumps such as those created through precommercial thinning generally do not result in infection of adjacent living pine or fir in the Pacific Northwest. Although thinning may result in some wounding and subsequent stem decay in residual grand firs, the percentage of stem decay is likely to be lower in thinned stands than in unthinned stands" (Filip et al. 1995).

Some of the watersheds containing young stands and plantations are deficient in old-forest structure as a result of their historical timber harvest. How might old forest be restored for watersheds where it is reduced from desired levels? I believe this process could contribute to restoration of old forest when it is deficient:

1. Identify existing old-forest patches and conserve them from timber harvest, so they could serve as a foundation for future old-forest habitat.
2. Identify mid- to late-seral patches (primarily understory reinitiation stands) in close proximity to existing old-forest patches as potential replacements for them.
3. Examine mid- to late-seral patches to determine which old-forest attributes they have, and to determine if management practices (thinning, etc.) could promote missing attributes more quickly than would occur by doing nothing.
4. Identify a desired landscape patch distribution and determine if early-seral stands (stand initiation and stem exclusion), when they occur at a desirable spacing, could be actively managed (thinned, etc.) to produce old-forest attributes more quickly than would occur by doing nothing.
5. When identifying candidates for future 'old forest multi-strata' patches, stands should be selected with the highest potential to survive to the old forest stage – namely areas on north-facing aspects and at high elevations, particularly if they occur within valley bottoms and drainage headwalls. The predicted location of these semi-stable environmental settings could be modeled by using criteria similar to those described in Camp et al. (1997).

Future treatments in young stands could also attempt to rehabilitate existing visual conditions by restoring a natural vegetation pattern. For example, previous clearcut units could be expanded and shaped in such a way as to approximate the pattern, juxtaposition, and size of patches created by historical occurrences of stand-replacing wildfire, particularly for areas within cold and moist forest PVGs (see "Timber Harvest and Landscape Pattern" discussion in section 4). At a minimum, visual rehabilitation should attempt to modify the geometrically simple pattern associated with existing square clearcuts (Perry 1988).

Promoting Limited Vegetation Components: Western White Pine

An ecological approach to forest management encourages analysts to adopt a broad perspective that emphasizes looking beyond site-level conditions to understand ecosystem-specific disturbance regimes at a landscape scale (Benecke 1996).

One potential pitfall of a broad perspective is the risk of overlooking limited vegetation components such as quaking aspen, western white pine, or black cottonwood – many of which have a restricted distribution and are basically indistinguishable at the coarse grain of a landscape scale.

Western white pine, a mid-seral tree species, is sometimes found on cool moist, cool wet, and warm moist sites in the upper montane and lower subalpine vegetation zones (Powell 1998). It was characterized as having a restricted geographical distribution in the Blue Mountains (Haig et al. 1941).

In actuality, western white pine has a fairly wide distribution in the Blues, but it occurs as a minor species and seldom comprises a plurality of the basal area in any individual stand. Due to changes caused by fire suppression, bark-beetle outbreaks, white pine blister rust (*Cronartium ribicola*), and other factors, it is believed that western white pine was more abundant historically in the Blue Mountains than at present.

Over the last 20 years, western white pine has increasingly been used in reforestation plantings because it survives well and has rapid juvenile growth. I recommend that rust-resistant sources of white pine continue to be planted on moist-forest sites where it is ecologically well adapted (see table 25).

In the near future, some historical plantations containing white pine will need to be thinned. Although stocking levels were not developed specifically for western white pine (Cochran et al. 1994, Powell 1999b), I suggest that Douglas-fir stocking levels be used for white pine, as recommended by Seidel and Cochran (1981).

Promoting Limited Vegetation Components: Broadleaved Tree Species

When present in areas dominated by conifer forests, golden yellows or tawny russets of autumn aspen foliage provide a welcome splash of color. Although it may be difficult to quantify, it is likely that quaking aspen was historically more abundant in the Blue Mountains than it is now – fire suppression and ungulate grazing for more than 120 years has undoubtedly reduced its distribution (Shirley and Erickson 2001, Swanson et al. 2010). [Figure 8 also provides aspen background information.]

[When fire regimes were functioning properly, conifers maintained flammability and promoted fire, thereby favoring the vegetative regeneration mode of suckering aspen and allowing aspen clones to persist in a landscape mosaic for many generations of aspen ramets (Cumming 2001).]

Aspen is a clonal species that regenerates primarily by producing suckers from its root system (Schier et al. 1985). Unfortunately, the suckers are highly palatable to elk, deer, and domestic livestock. In order to allow the suckers to persist and eventually grow above the browse height of large ungulates, it is a common practice to fence aspen clones to prevent grazing damage (fig. 87). Relict aspen clones exist sporadically on moist upland forest sites, and they represent a high restoration need (Shirley and Erickson 2001, Swanson et al. 2010).

Figure 87 – Fencing to prevent ungulate browsing of quaking aspen in a moist-forest setting. Buck-and-pole, A-frame-style fencing has been widely used on Umatilla NF for at least 30 years as a way to mitigate some impacts of ungulate herbivory on aspen suckers (Shirley and Erickson 2001). Fencing is often included with other activities in an integrated aspen restoration plan. When conifers need to be removed from aspen stands (because conifers suppress aspen regeneration), selling them for wood products may generate enough revenue to pay for ungulate-exclusion fencing. If some conifers that should be removed are suitable as buck-and-pole fencing material (such as lodgepole pine), they could obviously be retained on-site for this purpose.

An ungulate browsing narrative provided for black cottonwood (figure 88) applies in large measure to quaking aspen (Endress et al. 2012, Riggs et al. 2000). And as also described for black cottonwood (fig. 88), quaking aspen is appropriate for reintroduction on cool moist and cool very moist upland forest sites (table 27, appendix 1, shows potential vegetation types assigned to cool moist and cool very moist plant association groups) (Swanson 2007). Some literature sources (including Endress et al. 2012) also describe deleterious effects of wild and domestic herbivory on several species of upland willow, including Bebb and Scouler willows.

"In stands with very low levels of regeneration and high levels of crown loss and mortality (correlated with high root mortality), silvicultural manipulations may be unsuccessful in maintaining significant aspen cover on the site. Instead, management efforts may be better directed toward stands with intermediate levels of crown loss and/or higher levels of pre-existing regeneration, indicating that roots are still abundant and vigorous enough to respond to disturbance. Where regeneration is marginal to deficient and herbivory is significant, reduction of herbivore pressure will aid in recovery and increase the likelihood of aspen cover in the future" (Worrall et al. 2010).

Figure 88 – Black cottonwood often occurs on moist forest sites. Research suggests that aspen, cottonwood, and willow species were historically a common component of moist upland forest sites (Endress et al. 2012). But due to their low resistance to herbivory effects (Endress et al. 2012, Riggs et al. 2000), these species are now uncommon on upland sites, and contemporary land managers often perceive them to be obligate riparian species because they retreated to moist-site refugia (swales, intermittent stream drainages, moist meadow margins) in response to a century or more of ungulate browsing, fire exclusion, and conifer competition. Other woody plants commonly encountered on moist-forest sites, including Oregon boxleaf (*Paxistima myrsinites*) and Pacific yew (*Taxus brevifolia*), are frequently suppressed or eliminated by high levels of ungulate use (Box 1, earlier in this white paper, describes Pacific yew browsing effects). Cool moist and cool very moist habitats within a Moist Upland Forest potential vegetation group (table 27, appendix 1, shows potential vegetation types assigned to cool moist and cool very moist plant association groups) are appropriate candidates for reintroduction of black cottonwood on upland (non-riparian) sites (Swanson 2007).

Although long-term trend data is unavailable, black cottonwood is another species whose distribution is undoubtedly reduced from historical levels. Herbivory impact, and curtailment of frequent spring flooding, combined with other factors to limit cottonwood regeneration. Perhaps even more so than aspen, black cottonwood occurred frequently as a component in mixed stands on moist upland-forest sites.

Research in the Blue Mountains has consistently shown a strong association between ungulate herbivory levels and the abundance and vigor of woody, broadleaf plant species (Riggs et al. 2000, Swanson et al. 2010, Vavra et al. 2007).

"Ungulate herbivory can dramatically affect the density and structure of aspen, cottonwood, and willow in areas of high recruitment of these species after episodic disturbance. It is likely that in many coniferous forests of western North America, that the absence of aspen, cottonwood, and willow species as understory species may reflect the very short time periods over which these plant species can be substantially reduced or eliminated in response to high ungulate herbivory that typically follows episodic ground disturbances. Our results further suggest that aspen, cottonwood, and willow species are potentially common components of upland forests of grand fir and Douglas-fir communities of interior western North America [fig. 88]. Aspen, cottonwood, and willow species can exist as shrubs or trees, and our results indicate that these species have the potential to be dominant or co-dominant members of upland forest sites previously considered as strictly coniferous forest types" (Endress et al. 2012).

An Endress et al. (2012) quote, above, supports my assertion that we may not know for certain why aspen occurs exclusively as small stands in the Blue Mountains (stands of less than an acre are common here, as compared to stands covering dozens or hundreds of acres in Rocky Mountains of Utah, Wyoming, and Colorado).

I suspect, however, that contemporary Blue Mountain aspen has retreated to moist-site refugia (swales, intermittent stream drainages, moist meadow margins) in response to a century of ungulate browsing, fire exclusion, and conifer invasion (Swanson et al. 2010).

[Domestic livestock grazing levels were quite high from late 1800s through early 1900s; they are much reduced now (Oliver et al. 1994a). But, herbivory from native ungulates (deer, elk) is now much higher than it was in early 1900s, which has compensated somewhat for declines in domestic livestock grazing (Riggs et al. 2000).]

One factor that should be monitored for the future is wolves. Long-standing research in the greater Yellowstone area suggests that wolves initiate trophic cascades that aid recovery of aspen, cottonwood, willows, and berry-producing shrubs (Beschta et al. 2016, 2018; Beschta and Ripple 2012, 2016; Beyer et al. 2007; Fortin et al. 2005; Hollenbeck and Ripple 2007; Painter et al. 2015; Ripple and Beschta 2003, 2004, 2005, 2006, 2007).

Recent improvements noted for an Elk Flat aspen stand (Walla Walla RD) suggest that perhaps wolves are beginning to have an effect (from Wenaha wolf pack?).

Adaptive Management As A Project Planning Framework

Perhaps an ideal management framework for moist forests is adaptive management (fig. 89) (Bormann et al. 1994b). Adaptive management involves participatory planning, and it is informed by iterative learning about the ecological, social, and economic components of sustainability.

It accounts for previous successes and failures, it can promote increases in contemporary resilience (for both socioeconomic and ecological systems), and these in turn can improve an ecosystem's capacity to respond to future changes and threats (Johnson 1999, Lee 1999). The overall goal of adaptive management is not to maintain an optimal condition for a particular resource, but to develop an optimal management capacity (Johnson 1999).

Managing moist forest and other multiple-use landscapes by attempting to balance benefits for humans and a myriad of ecological values is a complicated endeavor, and I believe an adaptive management framework is ideally suited for this task. Although many illustrative models have been proposed for adaptive management, figure 89 provides a simple and commonly used one.

In its simplest form, adaptive management is learning from doing, and since learning occurs by implementing management policies and practices (not just from traditional application of science-based inquiry and associated research findings), adaptive management could provide an ideal framework for moist-forest management. And because the adaptive management model includes monitoring and evaluation in addition to the adjustment (adaptive) phase, it provides more options than simply muddling through or continuing a status quo (business-as-usual) approach.

Although adaptive management is intuitively attractive, it deserves an 'eyes wide open' approach. It "has been hailed as a solution to endless trial and error approaches to complex natural resource management challenges. However, its implementation has failed more often than not. It does not produce easy answers, and it is appropriate in only a subset of natural resource management problems. It is not a panacea for the navigation of 'wicked problems'" (Allen and Gunderson 2011: 1380).

Adaptive management presents an important challenge and opportunity for forestry in the decades to come, particularly in the context of climate change and its associated uncertainties.

Although it is still unclear how climate change might unfold, it obviously presents high risk of management failures; adaptive management could function as an integral component of an overall risk management strategy for dealing with rapid environmental change in the future (Bolte et al. 2009).

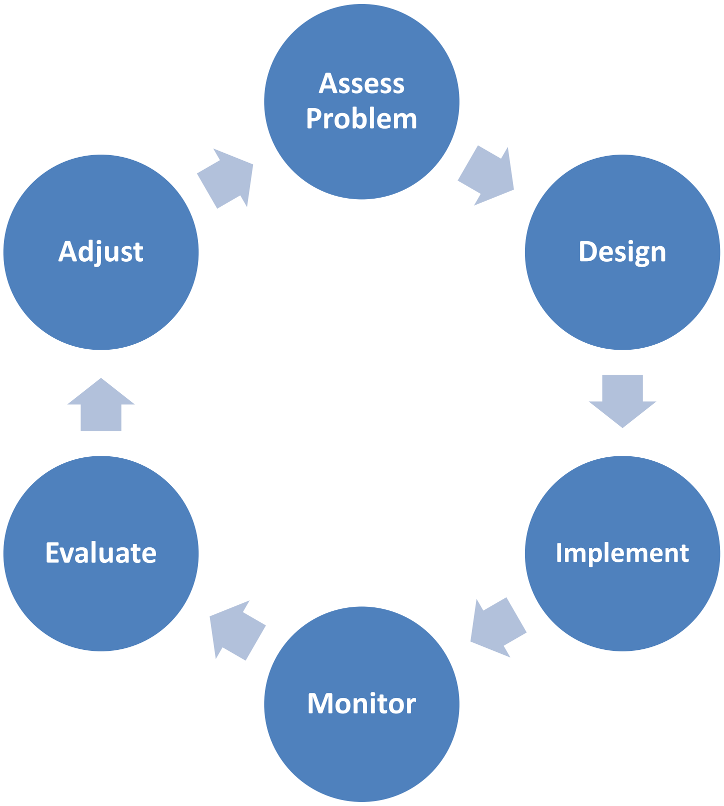


Figure 89 – Adaptive management cycle. Adaptive monitoring efforts must be capable of measuring (detecting) the outcomes of alternative management approaches; this is fundamental to any effective partnership between management and adaptive monitoring. Monitoring implies continuous assessment of one or more variables. It is a set of measures taken as a time series, thereby providing trend data. Monitoring is defined as "the collection and analysis of repeated observations or measurements to evaluate changes in condition and progress toward meeting a management objective" (Elzinga et al. 1998). Muddling through (Lindblom 1959) is sometimes adopted as a management strategy, but its lack of tactical clarity leaves a lot to be desired.

The adaptive management cycle could be especially important for moist-forest management because stakeholder judgments are provisional in response to evolving public opinion – what is acceptable today may change depending on new information or after adopting new management practices. Adaptive management also accounts for seemingly opposite judgments about an issue – stakeholders can support active and passive management simultaneously because of their expectation that each of them would occur in different and carefully selected areas, so both could easily coexist somewhere on an affected landscape (Olsen and Shindler 2010).

Essential to managing for uncertainty of changing climates is use of an adaptive approach where we learn from past and ongoing experiences (Stephens et al. 2010). A desirable goal for active management of moist forest should incorporate flexible strategies considering both historical and future ranges of variation, while "also accommodating knowledge advances and social changes using the principles of adaptive management" (Kuuluvainen and Grenfell 2012).

Restoring ecological integrity for altered moist-forest ecosystems often requires bold management to reinitiate feedback cycles and overcome the inertia of degraded landscapes and biophysical systems (Suding et al. 2004). Adaptive management as a response to climate change could also minimize carbon losses from wildfire.

Recent research concluded that "more carbon is lost from the system if fires are followed by delays in forest regeneration (Keyser et al. 2008), substantially reduced forest density (Kashian et al. 2006), or vegetation conversions to grasslands or shrublands (Savage and Mast 2005)" (Raymond and McKenzie 2012).

[But also consider that historically, moist-forest sites did not always support trees or just conifers – at any particular time, they would have supported at least 5% grass-forb cover types, 5% shrub cover types, and 10% broadleaved tree cover types, as shown in table 18 of section 6.3: Range of Variation. For contemporary moist-forest landscapes that experienced previous simplification and homogenization (Lehmkuhl et al. 1994), I doubt RV objectives for grass-forbs, shrubs, or broadleaved trees could be satisfied at the present time.]

"Ecosystems are moving targets, with multiple potential futures that are uncertain and unpredictable. Therefore management has to be flexible, adaptive, and experimental at scales compatible with the scales of critical ecosystem functions" (Walters 1986).

"Trying to turn the landscape upside down to hack out or cobble together a habitat for one endangered species or another may actually be harder work than reassembling an entire functioning ecological community that supplies the habitat by virtue of landscape-wide processes" (Simberloff 1990).

The insights from Walters (1986) and Simberloff (1990) recognize that moist forest (or any other ecosystem type) seldom exists in isolation. The most common situation is for moist forest to occur in a landscape mosaic – in some areas, it is the dominant landscape element (the matrix) and in others, it exists as patches (discrete stand-scale units) within a much more extensive dry-forest or cold-forest matrix.

A mosaic situation, however, can arise from several sources. In addition to the ecological mosaic described above, administrative land-use designations can create an 'operational' mosaic (Everett and Lehmkuhl 1996, 1999; Everett et al. 1994) where vegetation conditions can be modified for some areas by using timber harvest, whereas other areas can only be changed by applying prescribed fire or another non-harvest activity.

Some portions of the Umatilla National Forest, for example, are reserved from timber harvest (such as Wilderness areas, and designated old-growth units and other Forest Plan management areas where timber harvest is prohibited), while other portions have restrictions limiting harvest (such as Roadless Areas and Riparian Habitat Conservation Areas) (Christensen et al. 2007).

After accounting for reserves and restricted areas, areas remaining for timber harvest can be termed Active Forestry (this classification system is described in Rainville et al. 2008, and in table 3 of white paper F14-SO-WP-Silv-50).

For the Umatilla National Forest, figure 90 shows that approximately 45% of the National Forest System lands are classified as moist forest, with the remainder being nonforest (19%), dry forest (29%), and cold forest (7%). And of the moist forest, figure 90 shows that approximately 44% is reserves and 15% is restricted, neither of which are available for timber management, and that approximately 41% is Active Forestry, representing the proportion on which vegetation conditions could be changed by implementing timber harvest.

I am convinced that the adaptive management concepts discussed in this section are best implemented by using a collaborative approach (fig. 91). Why collaboration? Well, one compelling reason is that research suggests that diversity matters, and new leaps of human logic, innovation, and invention are more likely to arise when diverse people with differing backgrounds and abilities work together toward a common goal (Woolley et al. 2010). And, just as land management is a long-term endeavor requiring flexibility and sustained commitment, so too is collaboration.

“Collaboration among diverse stakeholders is expected to enhance learning, build social legitimacy for decision making, and establish relationships that support learning and adaptation in the long term. Leaders and facilitators of adaptive collaborative management can more effectively manage for productive stakeholder engagement and, thus, social-ecological resilience if they are more tentative in their convictions, more critical of the role of expert knowledge, and more attentive to the knowledge, interests, and power of diverse stakeholders” (Arnold et al. 2012).

Collaborative groups also provide enhanced capacity for the data collection and quality control components of a monitoring program. In some instances, collaborative groups contain scientific expertise comparable or superior to that of an agency, and long-standing groups tend to have in-depth understanding of an agency’s mission and objectives, often allowing them to function at a higher level than other sources of citizen science (Cohn 2008, Dickinson et al. 2012, McKinley et al. 2012).

And not all monitoring needs to include rigorous data collection – camera points (Hall 2002a, 2002b; Powell 2008b) provide effective information and insights (Powell 2014), although they do not provide quantification to the same extent as detailed inventory methods. Camera points and similar methodology are very compatible with the suite of expertise available from collaborative groups. [Figures 75 and 80 show how camera points can be used, and they demonstrate that meaningful trend data can be obtained relatively quickly, and inexpensively, by using camera points.]

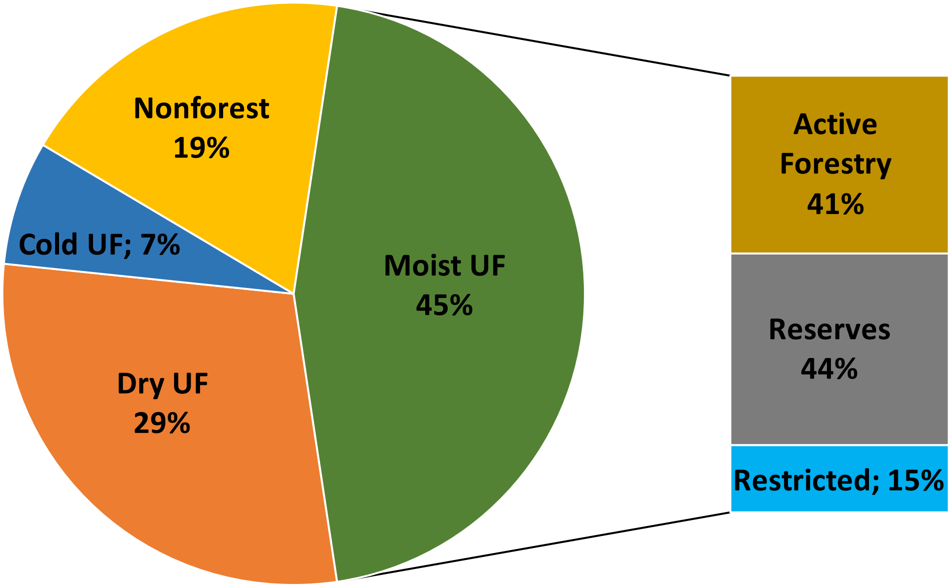


Figure 90 – Land allocation summary for Umatilla National Forest. Not all acres within Umatilla NF are available for timber management – nonforest herblands and shrublands (19%) do not support trees, and some forested lands are reserved from timber harvest, such as Wilderness and Forest Plan management allocations for which timber harvest is not authorized (such as C1 for dedicated old growth). Other forested lands have restrictions limiting harvest, such as Roadless Areas and riparian habitat conservation areas. A round pie chart on the left shows that 45% of the Forest's 1.4-million acre landbase supports moist upland forests. A column chart on the right breaks down moist forest into three land allocation categories – it shows that less than half (41%) of moist upland forest is available for timber harvest (Active Forestry is 41% of moist upland forest biophysical environment; this acreage is about 18% of total, 1.4-million-acre landbase for Umatilla NF). Vegetation modifications for the remaining 59% (reserves and restricted) must be completed by using tools other than timber management.

An important function of collaborative groups is to help keep an agency grounded: "participants in natural resource management organizations are not perceived by stakeholders as being as successful as they themselves think they are (Leach 2002). It reminds us that perceptions may differ from reality. It also suggests a disconnect between agency personnel and stakeholders, which can be problematic if agency personnel perceive they are being successful and stakeholders do not. An agency that perceives itself as successfully implementing a policy is likely to continue on its present course. If stakeholders view the agency's success differently, they may seek to impose change from the outside" (Koontz and Bodine 2008).

Figure 91 – Collaborative group tour involving a moist mixed-conifer forest science synthesis team led by Peter Stine (Stine's science integration group produced a general technical report entitled "The ecology and management of moist mixed-conifer forests in eastern Oregon and Washington: A synthesis of the relevant biophysical science and implications for future land management" (Stine et al. 2014).

This tour stop involved a 'Swamp Creek 1' plantation on the Walla Walla Ranger District; the area was clearcut in 1960 by using a ground-based harvest system. Slash was burned (after being windrowed), and it was planted in 1961 and then noncommercially thinned in 1971. Natural regeneration became established, so the existing stand has a very diverse mix of tree species. For trees greater than 5 inches dbh, the quadratic mean diameter was 12.5" as of the field trip in late August of 2012.

Collaborative field-trip participants were asked to evaluation options for reintroducing or improving stand complexity for Swamp Creek unit 1, and to discuss how any proposed management actions for Swamp Creek 1 might be made compatible with future climate change predictions for the northern Blue Mountains.

The role of trust figures prominently in collaboration because it has a crucial influence on our ability to apply active management, in an adaptive management framework, for moist forests of interior Pacific Northwest. As noted in a recent journal article: "the lack of public trust in agency proposals probably has been the largest single obstruction in moving active management forward on federal forestlands" (Franklin and Johnson 2012).

An important role for collaboration and public involvement in supporting active forest management has long been recognized: "the biggest barrier is social – that is, forging the social consensus on the actions needed to manage NFS [National Forest

System] lands to maintain ecosystem health" (MacCleery 1995, p. 43). Collaboration is a cornerstone of contemporary forest management (Brown 2012), and it is an important component of our Forest Service ethics: "our service ethic is to tell the truth, obey the law, work collaboratively, and use appropriate scientific information in caring for the land and serving people" (Thomas 1995, USDA Forest Service 1994).

Thoughts about concepts and principles relating to effective collaboration:[8]

- Good communication supports effective collaboration.
- Effective collaboration leads to trust.
- Trust is validated with good performance.
- Good performance enables a freedom to manage.
- Freedom to manage fosters creativity.
- Creativity is a privilege, not a right.
- Privileges are valuable and should be safeguarded.

And as a corollary, consequences of ineffective collaboration include these items:

- Poor performance (and follow-through) leads to a loss of trust.
- Loss of trust leads to poor communication.
- Poor communication contributes to conflict.
- Conflict inhibits creativity.
- Lack of creativity leads to process-oriented management.
- Process-oriented management is ineffective (except perhaps to litigants).

Consideration Of Best Available Science

Most Forest Service resource specialists recognize the value of science, and the importance of using 'best available science' as one of their professional responsibilities. Even so, some specialists do not use science as much as they could, sometimes because they believe science is written for other scientists (i.e., too complicated to be useful for land managers), or because workload considerations limit their opportunities to find or use science (e.g., science is left unused due to a lack of time and administrative support for obtaining, reading, and synthesizing it) (Archie et al. 2012).

[Note: one reason for producing white papers, including this one, is to synthesize relevant science information about a topic or issue, and thereby make it more accessible to natural resource professionals. One potential issue with this approach, however, is that science being included in white papers (and this paper includes almost 150 pages of references, not that I'm necessarily proud of this fact) is based on my conception of which references are most important, and which portion of an overall literature base may be most useful for typical natural resource managers working with moist upland forests. Another practitioner could have a different conception of

---

[8] From case studies involving Bridger-Teton NF Wilderness Planning Process; Guiding the Course Consensus Group; Pine-Eagle Consensus Group; and Trout Creek Mountains Working Group. See: Powell, D.C. 1995. Ecosystem analysis in the interior Columbia River basin: a survey of some alternative approaches. Walla Walla, WA: USDA Forest Service, Interior Columbia Basin Ecosystem Management Project. 66 p.

what's most important for this topic, and the references section would then look different than it does for this white paper.]

Professional and technical knowledge is acquired from a continually growing store of scientific information. This knowledge is not learned once in a lifetime – it steadily accumulates as a practitioner keeps abreast of new scientific information and is exposed to new ideas and concepts.

Keeping current with scientific literature is important because science is not a fixed set of beliefs – it is an evolving process. Knowledge we use today is available because we stand on the shoulders of those who preceded us, and we hope our contemporary contributions will allow those who follow us to stand on our shoulders as well.

Science is not absolute or irrefutable – much of what we know in a science context is endlessly evolving. This means that what constitutes high quality science might vary over time and across scientific disciplines. An objective of considering high quality science is for scientists "to provide a meaningful context to scientific information so that its validity might be judged and therefore useful to the policy-maker" (Moghissi et al. 2008). A meaningful context is important because there is not a single scientific truth, but always a set of options from which to choose.

One role of science is to frame or bound the options on which active management can be based, but not to actually make policy or guide management. Science fits uncomfortably in decision making because it can rarely be definitive. Acknowledging this reality is important to avoid misunderstandings and unrealistic expectations about roles and responsibilities for science.

One hallmark of good science is that it is free of value and policy judgments. Therefore, it's very important for practitioners and managers to locate, use, and cite good science (preferably 'best available science') because it functions as a foundation for sustainable natural resource management. When USDA Forest Service completes a planning process to authorize active management for moist forests, we must demonstrate in the administrative record that scientific evidence was given a 'hard look,' and that scientific methodologies were employed that satisfy legal obligations established by National Forest Management Act, National Environmental Policy Act, Endangered Species Act, and others.

These considerations (including cautions and caveats) relate to identification and utilization of best available science (BAS) when planning for active management of moist-forest ecosystems:

1. A primary focus of BAS should be on peer-reviewed literature published in credible outlets such as scientific journals. Although peer review is not a panacea (Bohannon 2013, Neff and Olden 2006), it provides a well-established process to evaluate scientific rigor.

2. Emphasizing journal papers should not overlook relevant scientific information produced by government agencies and non-governmental organizations, or information published in master's theses or Ph.D. dissertations.
3. As a corollary to items 1 and 2, it is incorrect to assume that scientific information provided in non-journal sources (including general technical reports, research papers, and research notes published by USDA Forest Service research stations) has not been peer reviewed, is 'gray' literature, or is otherwise irrelevant or of substandard quality in a BAS context.
4. Specialist reports produced during moist-forest project planning will often contain 'personal-communication' citations (and these may be important when critical analysis assumptions were derived from other interdisciplinary team members rather than from BAS), but most citations in specialist reports should refer to high-quality, peer-reviewed literature.
5. Much powerful science relating to active-management has compelling geographical relevance – citing longleaf pine research from southeastern U.S. in support of a ponderosa pine project in eastern Oregon is sure to raise questions – unless a clear and compelling rationale for doing so is clearly articulated.
6. Production of BAS almost always lags behind emerging issues, sometimes by a considerable time period. This means that reasonable and prudent interpretation of 'related science' will be necessary until BAS has time to catch up.
7. Correlation refers to a statistical relationship between two random variables or two sets of data. Some moist-forest science assumes that correlation between two variables implies causation (e.g., a cause-effect relationship). Cause-effect relationships are valuable because they allow us to fundamentally understand why a particular function or ecosystem state is occurring, and whether we can predict its occurrence elsewhere. However, we should always remember that causation is <u>not</u> established by correlating two sets of observations (comparing one dataset with another, for example), but is proven by a set of replicated experiments covering a wide range of circumstances (Schreuder and Thomas 1991).
8. Uncertainty is more than statistical error – it is increasingly being recognized as a multidimensional concept involving many qualitative (rather than quantitative) aspects, including inexactness related to limitations of methods or protocols, ignorance (despite pretenses or assertions to the contrary, we don't know everything about moist forests), incomplete or unsubstantiated assumptions, and restricted robustness of findings or conclusions.
9. More research does not necessarily reduce uncertainty. In fact, it often reveals unappreciated or heretofore unknown complexities and uncertainties.
10. High quality science does not **require** low levels of uncertainty.
11. Concerns about uncertainty (items 8-10 and 13) demonstrate that science cannot be definitive (at least to an extent often desired by land managers) because statistical uncertainty (and other sources of uncertainty as well) are part-and-parcel of the scientific method.

12. Science is often used strategically by process participants, including selective or biased use of science sources to support a particular policy agenda (e.g., to ‘grind an axe,’ ‘gore an opponent’s ox,’ or otherwise push a particular agenda).
13. As a corollary to the previous point, scientific uncertainty is sometimes magnified and distorted, or occasionally neglected and minimized, and both tactics have been used to support a particular position or agenda.
14. We frequently find ourselves in situations where available scientific evidence leads to more than one credible interpretation, suggesting that scientific consensus is unlikely for many natural resource issues because of their inherent complexities (and inherent complexity renders some issues as ‘wicked’ problems, sensu Carroll et al. 2007, Shindler and Cramer 1999, and Wang 2002).
15. Many issues surrounding interpretation and application of BAS are related to *how a question is framed by land managers* – this can influence science use because a management question is seldom framed exactly like the questions (hypotheses) that science was designed to answer.

Best available science refers to disclosure of relevant science in such a way that evaluators can determine if science was appropriately interpreted and applied during a planning or assessment effort.

A formal evaluation of best available science is called a science consistency review, which evaluates “whether scientific information of appropriate content, rigor, and applicability has been considered, evaluated, and synthesized” (Guldin et al. 2003a, 2003b). A credible science evaluation would also acknowledge responsible opposing scientific viewpoints to those used during preparation of a moist-forest project, including issues of scientific controversy.

And as mentioned above for collaboration, appropriate use and consideration of science is one of four components, four pillars, upon which our Forest Service ethics is based: “our service ethic is to tell the truth, obey the law, work collaboratively, and use appropriate scientific information in caring for the land and serving people” (Thomas 1995, USDA Forest Service 1994).

ACKNOWLEDGMENTS

Don Justice and Andrew Lacey (Supervisor’s Office), Gary Popek (Heppner Ranger District), and Robin Harris (Walla Walla Ranger District) helped compile GIS maps and data used for several of the figures and tables. Their expertise is greatly appreciated.

This paper benefited substantially from technical peer reviews provided by Craig Schmitt, forest pathologist (retired), and Lia Spiegel, forest entomologist, with Forest Health Protection’s Blue Mountains Service Center headquartered in La Grande, Oregon, and Eric Pfeifer, silviculturist with the Walla Walla Ranger District headquartered in Walla Walla, Washington. Their comments greatly enhanced the content and readability of this document.

PHOTO CREDITS

Unless noted otherwise in figure captions, all photographs, diagrams, or images were acquired or prepared by David C. Powell. Many of the photographs are available from the www.forestryimages.org website.

GLOSSARY

Abiotic. Nonliving components of the environment that are currently not part of living organisms, such as soils, rocks, water, air, light, and nutrients (Dunster and Dunster 1996). Compare with: biotic.

Active management. Human intervention into the nature, extent, and timing of disturbance to wildland ecosystems for the purpose of obtaining desired goods and services (Haeussler and Kneeshaw 2003). Also defined as the use of planning, thinning, prescribed fire, timber harvest, and reforestation to intentionally influence the health and resilience of a forest. In a climate-change context, active management refers to responses supporting ecosystem changes related to climate change (such as assisted species migration).

Adaptation. A far-term climate change strategy adopting tactics such as minimizing negative ecosystem effects (reforest now with tree species expected to be tolerant of future droughts), or by exploiting potential opportunities to adapt to future climatic conditions. Sometimes considered to be analogous with resilience. Adaptation and mitigation are important strategies for addressing climate change.

Adaptive management. A dynamic approach to land management in which the effects of treatments and decisions are continually monitored and used, along with research results, to modify management on a continuing basis to ensure that objectives are being met (Helms 1998).

Allelopathy. A competitive strategy in which certain plants produce chemical compounds (allelochemicals) interfering with the germination, growth, or development of other (competing) plants (Dunster and Dunster 1996).

Andisol. Soil order characterized by young soils on volcanic substrates (Chapin et al. 2002).

Anthropogenic. Resulting from or caused by people (Chapin et al. 2002).

Bark beetles. Small, often cylindrical beetles in the family Scolytidae that bore through the bark of host trees to lay their eggs and, as larvae, to tunnel and feed in the inner bark (Doliner and Borden 1984).

Basal area. Surface area of a woody stem, including the bark, as if cut off at breast height (4½ feet above the ground); also, surface area of all stems in a stand and expressed per unit of land area (basal area per acre) (Jennings et al. 2003).

Biological diversity (biodiversity). Variety of all fauna, flora, and microbes, and their habitats. Biodiversity is hierarchical, ranging from genetic diversity to species diversity and then ultimately ecosystem diversity (Powell et al. 2001).

Biological legacies. Remnants of a previous forest ecosystem that persist after a disturbance, including seeds, large wood in the form of standing dead trees (snags) and down logs, and undergrowth plant parts that replace aboveground biomass removed by the disturbance (Foster et al. 1998).

Biophysical environment. Landscape-level unit of vegetation composition and structure, with its associated environmental gradients and processes of change (Powell et al. 2007).

Biotic. Any living component of an ecosystem, including plants and animals (Dunster and Dunster 1996). An entity that is distinct from abiotic physical and chemical components (Allaby 1998). Compare with: abiotic.

Burn severity. Fire severity and burn severity are sometimes used interchangeably. Note that burn severity relates specifically to soils, particularly to the loss of organic matter from, and directly above, the mineral soil (Keeley et al. 2009). Compare with: fire severity.

Climax. The culminating seral stage in plant succession for any given site where, in the absence of catastrophic disturbance, the vegetation has reached a highly stable condition and undergoes change very slowly (Dunster and Dunster 1996). A self-replacing community that is relatively stable over several generations of the dominant plant species, or very persistent in comparison to other seral stages (Kimmins 1997). Also refer to seral stage: potential natural community.

Cohort. A group of trees developing after a single disturbance, commonly consisting of trees of similar age, although one cohort can include a considerable span of ages ranging from seedlings or sprouts to trees that predate the disturbance (Helms 1998). Stands are often characterized as single-cohort or multicohort depending on whether they contain one or several cohorts (Oliver and Larson 1996).

Collaboration or collaborative group. A structured manner in which a collection of people with diverse interests share knowledge, ideas, and resources while working together in an inclusive and cooperative manner toward a common purpose (USDA Forest Service 2012a).

Community. In an ecological context, a community is made up of all of the interacting populations in an environment. Community refers to a group of organisms that tends to occur together under similar environmental conditions, occupying the same habitat or area and interacting with each other (Doliner and Borden 1984). Community is usually considered to be a smaller spatial scale than an ecosystem.

Competition. Negative interactions between individuals of either the same or different species that utilize common and limited resources such as nesting sites, nutrients, or prey (Doliner and Borden 1984). For trees, competition results in a density-related scarcity of certain environmental factors that are important for tree growth and survival (Helms 1998).

Connectivity. Ecological conditions existing at several spatial and temporal scales that provide landscape linkages and permit exchange of flow, sediments, and nutrients; daily and seasonal movements of animals within home ranges; dispersal and genetic interchange between populations; and long distance range shifts of species, such as in response to climate change (USDA Forest Service 2012a).

Cover type. Plant species forming a plurality of the composition across a given land area, e.g., the Engelmann spruce-subalpine fir, ponderosa pine-Douglas-fir, or lodgepole pine forest cover types (Helms 1998). Forest cover types of the United States and Canada are described in Eyre (1980). Rangeland cover types of the United States are described in Shiflet (1994).

Crown class. A tree classification based on crown position relative to adjacent trees within the same canopy stratum; four primary crown classes are recognized:

Dominant – a tree whose crown extends above the general level of the main canopy, receiving full light from above and partial light from the sides.

Codominant – a tree whose crown helps to form the general level of the main canopy, receiving full light from above and limited light from the sides.

Intermediate – a tree whose crown extends into the lower portion of the main canopy but is shorter than the codominants, receiving little direct light from above and virtually none from the sides.

Subcanopy (overtopped) – a tree whose crown is completely overtopped by the crowns of one or more neighboring trees, occurring in a subordinate or submerged position relative to the main canopy.

Crown fire. An intense fire that burns through the upper tree or shrub canopy, spreading from one woody crown to another above the ground. Typically, understory vegetation is also burned. Depending on species, a crown fire may or may not be lethal to dominant vegetation. An example of this are many shrub and broadleaf tree species that sprout from roots, root crowns, or stem bases after their tops are killed. A crown fire may be continuous, or it may occur as patches within a lower severity burn (Sommers et al. 2011). Three types of crown fire are commonly recognized:

Passive crown fire. This crown fire type is characterized by the torching of a small group of trees (Stephens et al. 2012); a solid or continuous flaming front cannot be maintained except for short periods.

Independent crown fire. This crown fire type spreads without the aid of a supporting surface fire (compare with active crown fire) (Sommers et al. 2011).

Active crown fire. This crown fire type is characterized by fire spreading continuously in crown and surface fuels simultaneously (Stephens et al. 2012), which is termed a dependent crown fire, or in just the crown fuels with no surface fuel involvement (independent crown fire). A strongly wind-driven, independent crown fire is sometimes observed in boreal forest during late winter or spring when a snowpack covers surface fuels.

Desired future conditions. A description of land or resource conditions that are believed necessary if goals and objectives are to be fully achieved (Helms 1998).

Disease. Any more or less prolonged disturbance of an organism that interferes with its normal structure or function; the causes of disease are both biotic and abiotic (Doliner and Borden 1984).

Disturbance. A relatively discrete event that disrupts the structure of an ecosystem, community, or population, and changes resource availability or the physical environment. Disturbances include processes such as fires, floods, insect outbreaks, disease epidemics, and windstorms (Dodson et al. 1998).

Disturbance regime. A description of characteristic types of disturbance on a given landscape; and frequency, severity, and size distribution of characteristic disturbance types and their interactions (USDA Forest Service 2012a). Description of a disturbance regime includes characteristics such as the spatial distribution of disturbance events; disturbance frequency (number of disturbance events in a specified time interval, or the probability of a disturbance event occurring within a particular time interval); return interval (average time between successive disturbance events); rotation period (length of time until an area equivalent to the size of an analysis area would be affected in one disturbance event); disturbance size; and the magnitude, or intensity, of a disturbance event (Dodson et al. 1998).

Dominant species. Plant or animal species that competitively exclude subordinate species by capturing a disproportionate share of site resources, thus contributing most to productivity (Ellison et al. 2005).

Ecological amplitude. See: tolerance.

Ecological conditions. Biological and physical environments affecting diversity of plant and animal communities, persistence of native species, and productive capacity of ecological systems. Ecological conditions include habitat and other influences on species and environments. Examples of ecological conditions include abundance and distribution of aquatic and terrestrial habitats, connectivity, roads and other structural developments, human uses, and invasive species (USDA Forest Service 2012a).

Ecological integrity. Quality or condition of an ecosystem when its dominant ecological characteristics (for example, composition, structure, function, connectivity, and species composition and diversity) occur within the range of variation, and can withstand and recover from most perturbations imposed by natural environmental dynamics or human influence (USDA Forest Service 2012a).

Ecological threshold. “An ecological threshold is the point at which there is an abrupt change in an ecosystem quality, property or phenomenon, or where small changes in an environmental driver produce large responses in the ecosystem” (Groffman et al. 2006).

Ecological trajectory. “Sequential expression of an ecosystem over time, as if you could take a time-lapse movie of an ecosystem over several millennia and play it back in a few minutes. The movie will gradually show an extant ecosystem, but its composition and structure can be fluid, especially during periods of rapid environmental change, such as those caused by anthropogenic climate change” (Clewell and Aronson 2013).

Ecosystem. A spatially explicit, relatively homogeneous unit that includes all interacting organisms and elements of the abiotic environment within its boundaries. An ecosystem is commonly described in terms of its: (1) Composition. The biological elements within the different levels of biological organization, from genes and species to communities and ecosystems. (2) Structure. The organization and physical arrangement of biological elements such as, snags and down woody debris, vertical and horizontal distribution of vegetation, stream habitat complexity, landscape pattern, and connectivity. (3) Function. Ecological processes that sustain composition and structure, such as energy flow, nutrient cycling and retention, soil development and retention, predation and herbivory, and natural disturbances such as wind, fire, and floods. (4) Connectivity. (USDA Forest Service 2012a). Also see: connectivity.

Ecosystem management. Management driven by explicit goals, executed by policies, protocols and practices, and made adaptable by monitoring and research based on our best understanding of ecological interactions and processes necessary to sustain ecosystem composition, structure, and function (Christensen et al. 1996).

Ecosystem services. Ecosystem services include provisioning services such as food, water, timber, and fiber; regulating services affecting climate, floods, disease, wastes, and water quality; cultural services providing recreational, aesthetic, and spiritual benefits; and supporting services such as soil formation, photosynthesis, and nutrient cycling (Hassan et al. 2005).

Even-aged stand. A stand of trees composed of a single age class (USDA Forest Service 2012a).

Existing vegetation. Vegetation found at a given location at the time of observation (Jennings et al. 2003). Compare with: potential vegetation.

Fire. A self-sustaining chemical reaction releasing energy in the form of light and heat (Brenner 1998). Four types of fire are commonly recognized (arranged from least intense to most intense):

Ground fire. Fires burning in surface organic materials such as peat or deep duff layers. Ground fires typically undergo a large amount of smoldering combustion and less active flaming than other fire types. They may kill roots of overstory species due to prolonged high temperatures in the rooting zone (Sommers et al. 2011). [Although the terms are often used interchangeably, and incorrectly so, *ground fire is not the same as surface fire*.]

Surface fire. Fires burning only the lowest vegetation layer, which may consist of grasses, herbs, low shrubs, mosses or lichens (live fuels), and dead tree foliage and branchwood cast into the surface fuelbed from the overstory canopy. In forests, woodlands, or savannas, surface fires are generally low to moderate severity, and do not cause extensive overstory mortality (Sommers et al. 2011).

Mixed-severity fire. For this fire regime, fire severity varies between nonlethal understory fire and lethal stand replacement fire, with the variation occurring in space (between polygons) or time (within the same polygon). In some vegetation types, the stage of succession, the understory vegetation structure, the fuel condition, or the weather may determine whether a low or high-severity (or surface or crown) fire occurs. In this scenario, individual fires vary over time between low-severity surface fires and longer-interval stand replacement fires. In other situations, the severity may vary spatially as a function of landscape complexity or vegetation pattern, in which case the result may be a mosaic of young, old, and multi-aged vegetation patches (Sommers et al. 2011).

Stand replacement fire. A fire that is lethal to most of the dominant, aboveground vegetation, with the result that it substantially changes the vegetation structure. Stand replacement fires may occur in forests, woodlands and savannas, annual grasslands, and shrublands. Depending on the vegetation type being affected, stand replacement fire may result from crown fire, high-severity surface fire, or ground fire (Sommers et al. 2011). Also see: crown fire.

Fire behavior. This term relates to the manner in which fire reacts to fuel, weather, and topography; common terms used to describe fire behavior include smoldering, creeping, running, spotting, and torching (Sommers et al. 2011).

Fire exclusion. Areas where wildland fires were eliminated, including areas historically exposed to traditional Native American burning (Rapp 2002b).

Fire frequency. Number of times that fire occurs within a defined geographical area and during a specific time period. Fire frequency is sometimes characterized by using fire return intervals: very frequent (0-25 years between fires); frequent (26-75 years); and infrequent (76-150 or more years) (Sommers et al. 2011).

Fire intensity. Fire intensity describes the physical combustion process of energy release from organic matter. It is often expressed as fireline intensity – the rate of heat transfer per unit length of fireline. Since there is often a consistent relationship between fireline intensity and flame length, flame length may be used as a measure of fireline intensity (Keeley et al. 2009). Three intensity classes are recognized: low (average flame length of less than 3 feet), intermediate (average flame lengths of 3 to 9 feet), and high (flame lengths exceed 9 feet).

Fire regime. A generalized description of the role fire plays in an ecosystem (Agee 1993). When characterizing a fire regime, these attributes are often included: frequency, magnitude (intensity and/or severity), variability, seasonality, synergism, and extent (Agee 1998). Note that many fire regime classification systems exist; a recent one recognizes three primary regimes for <u>forested</u> environments (Brown and Smith 2000): (1) understory – fires are generally nonlethal to dominant vegetation (80% or more survives), and they do not change its structure; (2) mixed severity – fire either causes selective mortality in dominant vegetation (depending on its fire tolerance), or it varies between the understory and stand-replacement modes; and (3) stand replacement – fire kills or consumes the dominant vegetation (80% or more is either killed or consumed), and the forest structure is changed substantially. Compare with: disturbance regime.

Fire return interval. This metric describes the time between fires in a defined area, usually at the scale of a point, stand, or relatively small landscape area. This is called Mean Fire Interval (MFI) in the LANDFIRE system, when it refers to the average number of years between fires in representative stands (Barrett et al. 2010).

Fire rotation. The time required to burn an area equal to the defined area of a landscape. The entire area may not burn during this period; some sites may burn several times, and others not at all. Some authorities consider this term to be synonymous with fire cycle (Sommers et al. 2011).

Fire severity. Fire severity relates to the loss (death) or decomposition of organic matter both aboveground and belowground, including tree mortality as a 'loss' component, but this mortality context is most appropriate for trees lacking any sprouting capacity. Fire severity is correlated with fire intensity (Keeley et al. 2009).

Fire suppression. All activities associated with controlling and extinguishing a fire following its detection (Dunster and Dunster 1996).

Focal species. A small subset of species whose status permits inference to integrity of the larger ecological system to which it belongs, and provides meaningful information regarding effectiveness of a plan in maintaining or restoring ecological conditions to maintain diversity of plant and animal communities in the plan area. Focal species would be commonly selected on the basis of their functional role in ecosystems (USDA Forest Service 2012a).

Forest density management. Cutting or killing trees to increase inter-tree spacing and accelerate growth of remaining trees; the manipulation and control of forest (tree) density to achieve one or more resource objectives. Forest density management is often used to improve forest health, to open the canopy for selected trees, to maintain understory vegetation, or to promote rapid development of late-successional characteristics for biological diversity (Helms 1998).

Forest floor. A general term encompassing the layer of undecomposed organic matter (leaves, twigs, and plant remains in various stages of decomposition) lying on top of the mineral soil (Dunster and Dunster 1996).

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

Forest management. Intentional manipulation of forest ecosystems to influence their composition, structure, or density, and the nature of the products and services they provide (Burger 2009). Also see: active management.

Forest stand. A contiguous group of trees sufficiently uniform in age-class distribution, composition, and structure, and growing on a site of sufficiently uniform quality, to be a distinguishable unit (Helms 1998).

Fragmentation. In a landscape ecology context, fragmentation is a process creating an increasingly complex mosaic of patches resulting from disturbance, including human activities; fragmentation breaks apart a given area into smaller, more geometrically simple pieces (Rochelle et al. 1999, Voller and Harrison 1998).

Fuel. All of the dead and living material in an ecosystem that will burn; fuel includes grasses, dead branches and pine needles on the ground, as well as standing live and dead trees (Brenner 1998). Four types of fuel are commonly recognized (arranged from lowest to highest):

Ground fuel. A fuel component consisting of duff (the Oi soil horizon) and other materials (such as peat) lying on top of a mineral soil surface; ground fuels generally do not contribute to wildfire spread or intensity (Stephens et al. 2012).

Surface fuel. A fuel component including dead and down woody materials, litter, grasses, herbaceous plant material, and short shrubs; surface fuels may be the most hazardous fuel component for some forest types (Stephens et al. 2012).

Ladder fuel. A fuel component consisting of small trees or tall shrubs providing vertical continuity from surface fuels to canopy (crown) fuels (Stephens et al. 2012). Ladder fuels are important for initiating crown fire, but they have little or no influence on crown fire spread.

Crown fuel. A fuel component comprised of overstory tree crowns and canopies (including foliage and small branches); note that the canopy and crowns of small trees (seedlings and saplings) are often included in the ladder-fuels category. Of the three primary fuel components (surface, ladder, crown fuels), fire scientists often consider crown fuels to be the least hazardous (Stephens et al. 2012), but my experience is that this sentiment is seldom shared by managers and practitioners. Crown fuels are typically quantified as canopy bulk density – the mass of available canopy fuel per unit of canopy volume, often expressed in units of kg (mass) per cubic meter (volume).

Fuel load. The amount of combustible material (living and dead organic matter) found in an area (Brenner 1998).

Fuel treatment. Manipulation or removal of fuels to reduce the likelihood of fire ignition, lessen potential fire-caused damage, and improve resistance to control.

Gap. In forestry usage, a gap is the space left in the canopy when one or more trees die, or are removed during timber harvest.

Gap-phase succession. Succession occurring in small holes or gaps within a patch (stand) due to death of individual plants or plant parts (Chapin et al. 2002).

Growing space. An intangible measure of total resources of a site (sunlight, moisture, nutrients, etc.) available for plants (Helms 1998). Growing space refers to availability of all resources needed by a plant to exist on a given site (O'Hara 1996).

Hazard. Stand, tree, and environmental characteristics conducive to an insect outbreak or disease infection (Doliner and Borden 1984). Compare to: susceptibility.

Indicator species. Species used to monitor environmental change or represent specific environmental conditions (Eycott et al. 2007), including plant species conveying information about the ecological nature of a site, such as the nitrogen content, or the alkalinity or acidity of its soils. These plant species have a sufficiently consistent association with a specific environmental condition, or with other species, such that their presence can be used to indicate or predict the environmental condition, or a potential for the other species (Kimmins 1997).

Integrated landscape management. "The integrated planning and assessment of land uses and human activities over whole landscapes to ensure the long-term economic, social, and environmental sustainability of ecosystems and their resources. It is applied at appropriate temporal and spatial scales necessary to achieve multiple management objectives" (Canadian Council of Forest Ministers 2008).

Integrated pest management. The maintenance of biotic disturbance agents, including insects and diseases, at tolerable levels by the planned use of preventive, suppressive, or regulatory strategies (including silvicultural treatments) that are ecologically and socially acceptable.

Keystone species. Species with ecosystem effects that are disproportionately large in comparison to their biomass or number (Eycott et al. 2007). The gopher tortoise, for example, is a keystone species because more than 330 other species use its burrows (Simberloff 1999).

Landscape. A defined area irrespective of ownership or other artificial boundaries, such as a spatial mosaic of terrestrial and aquatic ecosystems, landforms, and plant communities, repeated in similar form throughout such a defined area (USDA Forest Service 2012a).

Landscape ecology. A study of structure, function, and change in a heterogeneous land area composed of interacting ecosystems (Forman and Godron 1986). Some landscape ecologists classify the spatial elements of a landscape into three primary components:

Matrix: the most extensive and most connected landscape element; it plays a dominant role in landscape function. The matrix is the landscape element surrounding a patch.

Patch: a nonlinear land area differing in appearance from its surroundings. Patches are a landscape element distinct from the matrix and isolated from other similar areas (patches).

Corridor: a narrow, linear land feature differing from the matrix on either side. Riparian habitats along streams or rivers often function as corridors (Forman and Godron 1986).

Layer (vegetation). A structural component of a plant community consisting of plants of approximately the same height stature (e.g., tree, shrub, and herb layer); as defined here, synonymous with stratum (Jennings et al. 2003).

Life history traits. Traits of an organism (e.g., seed size and number, potential growth rate, maximum size, and longevity) influencing how quickly a species can get to a site, how quickly it grows, how tall it gets, and how long it survives (Chapin et al. 2002).

Litter. Dead debris (plant material) covering the ground, including cones, needles or other shed foliage, branches, and other material (Brenner 1998).

Management area. A land area identified within the planning area that has the same set of applicable plan components. A management area does not have to be spatially contiguous (USDA Forest Service 2012a).

Management implication. A quantifiable index or attribute used to determine the success of implementing land management planning guidelines. An example is the use of wildlife indicator species (Dunster and Dunster 1996).

Mechanical treatment. Mechanical treatment refers to the use of tractors or other machinery to remove trees in a tree harvest operation (stewardship harvest), or to the use of hand-operated tools (chain saws, axes, etc.) to cut, clear, thin, girdle or prune woody plant species (Powell et al. 2001).

Mitigation. A near-term climate change strategy adopting tactics such as reducing greenhouse gas emissions (by reducing wildfire emissions, for example), or by enhancing carbon uptake and storage. Mitigation is sometimes considered to be analogous to resistance. Near-term mitigation and far-term adaptation are important strategies, in some combination, to address climate change.

Monitoring. A systematic process of collecting information to evaluate effects of management actions, or changes in conditions or relationships (USDA Forest Service 2012a).

Native knowledge. A way of knowing or understanding the world, including traditional ecological and social knowledge of the environment derived from multiple generations of indigenous peoples' interactions, observations, and experiences with their ecological systems. Native knowledge is place-based and culture-based knowledge in which people learn to live in and adapt to their own environment through interactions, observations, and experiences with their ecological system. This knowledge is generally not solely gained, developed by, or retained by individuals, but is rather accumulated over successive generations, and is expressed through oral traditions, ceremonies, stories, dances, songs, art, and other means within a cultural context (USDA Forest Service 2012a).

Native species. An organism that was historically, or currently is, present in a particular ecosystem as a result of natural migratory or evolutionary processes; and not present as a result of accidental or deliberate introduction into the ecosystem.

Natural regeneration. The renewal of a forest community by natural (as compared to human) means, such as tree seedling establishment from seed on-site, from adjacent areas, or seed brought in by wind currents, birds, or animals.

Nature. This term has been used to mean the natural world on Earth as it exists without human beings or civilization, that is, the environment including mountains, plains, rivers, lakes, oceans, air, and rocks, along with all other nonhuman, nondomesticated living things (Botkin 1990a).

Old forest. A forest structural stage characterized by a predominance of large trees (> 21" dbh) in a stand with either one or multiple canopy layers. On warm dry sites that historically featured frequent, low-severity surface fires, a single stratum may be present (old forest single stratum). On cool moist sites where surface fire was relatively uncommon, multi-layer stands with large trees in the uppermost stratum are typically found (old forest multi strata). Compare with: old growth.

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). For national forest system lands in the Pacfic Northwest, characteristics (attributes) of old-growth forests are described in USDA Forest Service (1993a).

Outbreak. A sudden increase in destructiveness or population level of a pest species in a given area; usually used in reference to bark beetles, defoliators, and other forest insects (Doliner and Borden 1984).

Parent material. Rocks or other substrates that generate soils through weathering (Chapin et al. 2002).

Pathogen. Any agent, whether a living organism or abiotic factor, that induces disease (Doliner and Borden 1984).

Plant association. A plant community with similar physiognomy (form and structure) and floristics; commonly it is a climax community (Allaby 1998). It is believed that 1) the individual species in the association are, to some extent, adapted to each other; 2) the association is made up of species that have similar environmental requirements; and 3) the association has some degree of integration (Kimmins 1997). Also see: climax; seral stage: potential natural community.

Plant association group (PAG). Groupings of plant associations (and other potential vegetation types such as plant communities and plant community types) representing similar ecological environments, as defined by using temperature and moisture regimes (Powell et al. 2007). The most common PAG in the Moist Upland Forest PVG is the Cool Moist Upland Forest PAG.

Plant community. A naturally occurring assemblage of plant species living in a defined area or habitat (USDA Forest Service 2012a). In a vegetation classification context: (1) a plant community has no particular successional (seral) status; (2) plant communities represent vegetation types with a restricted geographical distribution; and (3) plant communities have such a small number of sample plots that it is not possible to infer their true successional status (Johnson and Clausnitzer 1992).

Plant community type. An aggregation of all plant communities with similar structure and floristic composition. A vegetation classification unit with no particular successional status implied (Dunster and Dunster 1996).

Plant succession. The process by which a series of different plant communities, along with associated animals and microbes, successively occupy and replace each other over time in a particular ecosystem or landscape location following a disturbance event (Kimmins 1997). The process of development (or redevelopment) of an ecosystem over time (Botkin 1990a).

Potential vegetation. The vegetation that would become established if successional sequences were completed without interference by man or natural disturbance under present climatic and edaphic conditions; the plant community developing if all successional sequences were completed under existing site conditions (Dunster and Dunster 1996). Also see: climax; seral stage: potential natural community.

Potential vegetation group (PVG). An aggregation of plant association groups (PAGs) with similar environmental regimes and dominant plant species. Each PVG includes PAGs representing a similar temperature or moisture influence (Powell et al. 2007). The focus of this white paper is Moist Upland Forest, a PVG characterized by relatively moderate temperature conditions and a moist or mesic moisture regime.

Prescribed fire. Deliberate burning of wildland fuels in either a natural or modified state, and under specified environmental conditions, in order to confine the fire to a predetermined area, and to produce a fireline intensity and rate of spread meeting land management objectives (Powell et al. 2001).

Productivity. The capacity of NFS lands and their ecological systems to provide the various renewable resources in certain amounts in perpetuity (USDA Forest Service 2012a). For this white paper, productivity is an ecological term, not an economic term.

Range of variation. A characterization of fluctuations in ecosystem conditions or processes over time; an analytical technique used to define the bounds of ecosystem behavior that remain relatively consistent through time (Morgan et al. 1994). Values of composition, structure, or another attribute, and falling between upper and lower bounds determined for the attribute (Jennings et al. 2003), are said to be within the range of variation. Attributes whose values occur above the upper bound are said to be 'over-represented;' attributes whose values are below the lower bound are said to be 'under-represented' (see fig. 68). Also see: reference conditions.

Reburn. The repeat burning of an area over which a fire has previously passed, but has left unburnt fuel (Helms 1998).

Reference conditions. A reference ecosystem or reference conditions can serve as a model for planning ecosystem restoration activities. In its simplest form, the reference is an actual site, its written description (such as historical accounts of a reference area), or both (Society for Ecological Restoration 2004). Reference conditions also refer to a range of variation in ecological structures and processes, reflecting recent evolutionary history and the dynamic interplay of biotic and abiotic factors. Reference conditions generally reflect ecosystem properties that are free of major influence by Euro-American humans (Kaufmann et al. 1994).

Resilience. Intrinsic properties allowing the fundamental functions of an ecosystem to persist in the presence of disturbance; the 'bounce-back' capability of a system to recover from disturbance. "Ecological resilience is the capacity of an ecosystem to absorb disturbance and undergo change while maintaining its essential functions, structures, identity, and feedbacks. Resilience is often synonymous with adaptive capacity, i.e., the ability of a system to reconfigure itself in the face of disturbance or stresses without significant decreases in critical aspects such as productivity or composition" (Drever et al. 2006). Resilience recognizes that systems have a capacity to absorb disturbance, but this capacity has limits and when they are exceeded, the system may rapidly transition to a different state or developmental trajectory (Gunderson et al. 2010). In a climate-change context, resilience is sometimes viewed as analogous to adaptation.

Resistance. Resistance refers to the ability of an ecosystem to remain relatively unchanged in the face of external forces such as disturbance (pulse-type changes) or climate change. Resistance is sometimes viewed as being analogous to stability (Holling 1973), but in a climate-change context, it is often viewed as analogous to mitigation.

Restoration. Holistic action taken to modify an ecosystem to achieve desired, healthy, and functioning conditions and processes. This term is generally used to refer to the process of enabling a system to resume acting, or continue to act, following disturbance as if disturbance had not occurred (Powell et al. 2001). Restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. Ecological restoration focuses on reestablishing the composition, structure, pattern, and ecological processes necessary to facilitate terrestrial and aquatic ecosystem sustainability, resilience, and health under current and future conditions (USDA Forest Service 2012a). Two restoration approaches have been described:

Active restoration: an approach involving implementation of active management practices (prescribed fire, thinning, etc.) designed to restore appropriate composition, structure, or density conditions.

Passive restoration: an approach involving removal of stressors causing ecosystem degradation, such as cessation of fire suppression (exclusion) in fire-dependent ecosystems (Rapp 2002b).

Riparian areas. Three-dimensional ecotones of interaction between terrestrial and aquatic ecosystems extending down into the groundwater, up above the canopy, outward across the floodplain, up nearby side-slopes draining to the water, laterally into the terrestrial ecosystem, and along the water course at variable widths (USDA Forest Service 2012a).

Riparian management zone (riparian habitat conservation areas). Portions of a watershed where riparian-dependent resources receive primary emphasis, and for which plans include plan components to maintain or restore riparian functions and ecological functions (USDA Forest Service 2012a).

Risk. A combination of the likelihood that a negative outcome will occur (as related to susceptibility and vulnerability), and severity of the resulting negative consequences (USDA Forest Service 2012a). Note that risk refers to an event with a known occurrence probability, whereas uncertainty refers to an event with an unknown probability.

Secondary forest succession. Succession occurring in an environment that has already been modified by a period of occupancy by living organisms. Secondary succession can be progressive (proceeding from an early stage to a later one) or retrogressive (proceeding from a later stage back to an earlier one) (Hall et al. 1995). Forest clearcuts and abandoned agricultural fields are examples of environments undergoing secondary succession (Kimmins 1997).

Seral stage. The identifiable stages in the development of a sere, from an initial pioneer stage, through various early and mid-seral stages, to late seral, subclimax, and climax stages. The stages are identified by different plant communities, different ages of the dominant vegetation, and by different microclimatic, soil, and forest conditions (Kimmins 1997). Four seral stages are recognized (Hall et al. 1995):

Early Seral: clear dominance of seral species (western larch, ponderosa pine, lodgepole pine, etc.); PNC species are absent or present in very low numbers.

Mid Seral: PNC species are increasing in the forest composition as they actively colonize the site (or as they continue an ongoing developmental process); PNC species are approaching equal proportions with seral species.

Late Seral: PNC species are now dominant, but long-lived early-seral species (ponderosa pine, western larch, etc.) may still persist in the plant community.

Potential Natural Community (PNC): the biotic community that one presumes would be established and maintained over time under present environmental conditions; early- and mid-seral species are scarce or absent in the plant composition.

Severity. Proportion of the organic matter lost from the vegetation and surface soils due to disturbances (Chapin et al. 2002).

Shade tolerance. The capacity of trees to grow satisfactorily in the shade of, and in competition with, other trees (Helms 1998). Also see: tolerance.

Shifting mosaic. Landscape in which patches differ in successional stage, but the landscape as a whole is at steady state (i.e., there is no directional change in the relative proportions of different successional stages or structure classes) (Chapin et al. 2002).

Silvicultural prescription. A planned series of treatments designed to change current forest structure to one meeting the goals and objectives established for an area (Helms 1998). A prescription is a written statement or document defining the outcomes to be attained from silvicultural treatments; outcomes are generally expressed as acceptable ranges of the various indices being used to characterize forest development (Dunster and Dunster 1996).

Silvicultural treatment. An activity, practice, or action that can be applied in a controlled manner, according to the specifications of a silvicultural prescription or forest plan, to improve actual or potential conditions or benefits (Hoffman et al. 1999).

Silviculture. Applying techniques or practices to manipulate forest vegetation by directing stand and tree development, and by creating or maintaining desired conditions. Silviculture is based on an ecosystem concept that emphasizes the need to evaluate the many abiotic and biotic factors influencing the choice and outcome of silvicultural treatments and their sequence over time, and the long-term consequences and sustainability of management regimes. [Definition derived from multiple sources.]

Soil compaction. The process by which soil grains or particles are rearranged, resulting in a decrease in void space and causing closer contact with one another, thereby increasing bulk density (Helms 1998).

Species diversity. Number, evenness, and composition of species in an ecosystem; the total range of biological attributes of all species present in an ecosystem (Chapin et al. 2002).

Stability. Ecosystem stability is the ability of an ecosystem to maintain its existing trajectory (pathway) in spite of stress; it denotes dynamic equilibrium rather than stasis. Stability is related to an ecosystem's capacity for resistance and resilience (SERI 2004).

Stakeholders. Local community leaders, county governments, affected American Indian tribes, industrial organizations, local and national environmental organizations, and involved citizens that have a keen interest, or stake, in what's happening on public lands (Schmidt et al. 1993).

Stewardship. Taking a long-term and integrated view of resource management – air, water, land, plants and animals – recognizing the dependent relationships of humans on the environment, and that environmental health is fundamental to economic and human health (British Columbia Habitat Branch 2000).

Stewardship harvest. Often, a tree harvest operation completed for reasons other than production of timber commodities (Powell et al. 2001). Stewardship harvest also involves situations where the timber volume to be removed by a silvicultural treatment is insufficient to cover treatment costs (logging, transportation, etc.), so a subsidy payment must be made (e.g., cash contributed) to make the project financially viable.

Stressors. Factors that may directly or indirectly degrade or impair ecosystem composition, structure, or ecological process in a manner that may impair its ecological integrity, such as an invasive species, loss of connectivity, or the disruption of a natural disturbance regime (USDA Forest Service 2012a).

Structural stage. A stage or recognizable condition that relates to the physical orientation and arrangement of vegetation; the size and arrangement (both vertical and horizontal) of trees and tree parts. The following structural stages have been described (O'Hara et al. 1996, Oliver and Larson 1996):

> **Stand initiation**: one canopy stratum of seedlings and saplings is present; grasses, forbs, and shrubs typically coexist with the trees.
>
> **Stem exclusion**: one canopy stratum comprised mostly of pole-sized trees (5-8.9″ in diameter) is present. The canopy layer may be open (*stem exclusion open canopy*) on sites where moisture is limiting, or closed (*stem exclusion closed canopy*) on sites where light is a limiting resource.
>
> **Young forest multi strata**: three or more canopy layers are present; the size class of the uppermost stratum is typically small trees (9-20.9″ in diameter). Large trees may be absent or scarce.
>
> **Understory reinitiation**: two canopy strata are present the size class of the uppermost stratum is typically small trees (9-20.9″ in diameter). In this stage, a second tree layer is established under an older overstory. Overstory mortality created growing space for the establishment of understory trees.
>
> **Old forest**: a predominance of large trees (> 21″ in diameter) is present in a stand with one or more canopy strata. On warm dry sites with frequent, low-intensity fires, a single stratum may be present (*old forest single stratum*). On cool moist sites without recurring underburns, multi-layer stands with large trees in the uppermost stratum may be present (*old forest multi strata*).

Successional stage. A stage or recognizable condition of a plant community occurring during its development from bare ground to climax. Determined using two main criteria: tree size class and stand age. Coniferous forests in the Blue Mountains progress through six recognized stages, as defined below (Thomas 1979).

Grass-forb: dominant vegetation is herbaceous (grasses, grass-like plants, and forbs); stand age: less than 10 years; downed logs are present but not decayed.

Shrub-seedling: dominant vegetation is woody shrubs and/or tree seedlings; stand age: less than 10 years; downed logs are present but not decayed.

Pole-sapling: dominated by trees in the sapling size class, pole size class, or both; stand age: 11-39 years; even-height canopy; logs on ground are beginning to decay.

Young: dominated by trees that are no longer poles, but have not yet reached maturity; stand age: 40-79 years; self-thinning beginning; downed logs are moderately decayed; understory vegetation is starting to reappear.

Mature: domination or predominance of mature, vigorous trees; stand age: 80-159 years; self-thinning occurring; both decayed and undecayed logs are on the ground; some snags are present; understory vegetation is well established.

Old Growth: a stand past full maturity and showing decadence – the last stage in forest succession; stand age: 160 years and greater; understory vegetation is well established; snags are present; heart rot and other signs of decadence are common; all tree sizes and ages represented to some extent; abundant decayed and undecayed logs on the ground.

Surrogate species. These species are used to monitor the effects of change in other, less-common species (Eycott et al. 2007).

Susceptibility. This term refers to the probability of an organism being infected or infested by another organism (trees affected by bark beetles, defoliators, etc.), as evaluated by using inherent or intrinsic forest characteristics (species composition, stand density, etc.). The terms susceptibility and hazard are often used interchangeably. Compare with: vulnerability.

Sustainability. The capacity of forests, ranging from stands to ecoregions, to maintain their health, productivity, diversity, and overall integrity, in the long run, and in the context of human activity and use (Helms 1998). The capability to meet the needs of the present generation without compromising the ability of future generations to meet their needs (USDA Forest Service 2012a).

Sustainable forest management. Active "management that maintains and enhances the long-term health of forest ecosystems for the benefit of all living things while providing environmental, economic, social, and cultural opportunities for present and future generations" (Canadian Council of Forest Ministers 2008).

Target species. Specific target species for research or action; generally refers to species identified for conservation action (Eycott et al. 2007).

Thinning. A treatment designed to reduce tree density and thereby improve growth of the residual trees, enhance forest health, or recover potential mortality resulting from intertree competition. Two types of thinning are recognized – commercial thinning where the trees being removed are large enough to have economic value, and noncommercial thinning where trees are too small to be sold for conventional wood products, so the excess trees are cut and generally left on-site (Powell et al. 2001).

Timber harvest. The removal of trees for wood fiber use and other multiple-use purposes (USDA Forest Service 2012a).

Timber production. The purposeful growing, tending, harvesting, and regeneration of regulated crops of trees to be cut into logs, bolts, or other round sections for industrial or consumer use (USDA Forest Service 2012a).

Tolerance. A forestry term expressing the relative ability of a plant (tree) to complete its life history, from seedling to adult, under the cover of a forest canopy and while experiencing competition with other plants (Harlow et al. 1996). In general ecology usage, tolerance refers to the capacity of an organism or biological process to subsist under a given set of environmental conditions. Note that the range of conditions under which an organism can subsist, representing its limits of tolerance, is termed its ecological amplitude (Helms 1998).

Traditional ecological knowledge. See: native knowledge.

Umbrella species. These species are used to represent some of the needs of other species (Eycott et al. 2007).

Undergrowth. Herbaceous and shrubby plants growing beneath a forest canopy; as used in this document, undergrowth does not include small trees such as seedlings or saplings. Compare with: understory.

Understory. All of the vegetation growing under a forest overstory. In some applications, understory is only considered to be small trees (e.g., in a forest comprised of multiple canopy layers, the taller trees form the overstory, the shorter trees the understory); in other instances, understory is assumed to include herbaceous and shrubby plants in addition to trees. When understory is assumed to refer to trees only, other plants (herbs and shrubs) are often called an undergrowth to differentiate between the two (Helms 1998). Compare with: undergrowth.

Viable population. A population of a species that continues to persist over the long term with sufficient distribution to be resilient and adaptable to stressors and likely future environments (USDA Forest Service 2012a).

Vulnerability. This term refers to the probability of tree or forest damage resulting from an infection or infestation by damaging agents (such as bark beetles, defoliators, etc.). Susceptibility reflects the influence of forest or stand conditions (are lodgepole pines in a stand larger than 9 inches in diameter, which renders them susceptible to bark-beetle attack?), whereas vulnerability relates to whether damage will actually occur (is a mountain pine beetle population in close proximity to a lodgepole pine forest containing susceptible trees?).

Watershed. A region or land area drained by a single stream, river, or drainage network; a drainage basin (USDA Forest Service 2012a).

Wildfire. Any fire occurring on wildlands that is not meeting management objectives and thus merits a suppression response (Brenner 1998).

Wildland-urban interface. Areas where human communities are built in proximity to flammable fuels found in wildlands (Brenner 1998).

Wood decay. The decomposition of wood by fungi and other microorganisms, resulting in softening, progressive loss of strength and weight, and often changes in texture and color (Helms 1998). Terms associated with wood decay, and referenced in this white paper, are provided below (unless noted otherwise, term definitions provided by the USDA Forest Service, Forest Products Laboratory).

Bluestain. A deepseated fungal discoloration, predominantly bluish in color but sometimes grey, black or brown, confined mostly to the sapwood. Bluestain does not cause a loss of structural strength (Doliner and Borden 1984).

Brown rot. In wood, any decay in which the fungal attack concentrates on the cellulose and associated carbohydrates rather than on the lignin, which produces a light to dark brown friable residue known variously as 'dry rot' or 'cubical rot'.

Heart rot. Any rot or decay characteristically confined to the heartwood portion of a tree stem. Heart rot generally originates in the living tree (such as rust-red stringy rot caused by the Indian paint fungus).

Incipient decay. An early stage of tree decay that has not proceeded far enough to soften or otherwise perceptibly impair the hardness of wood. It is usually accompanied by a slight discoloration or bleaching of wood tissue.

White rot. In wood, any decay or rot attacking both the cellulose and the lignin, producing a generally whitish residue that may be spongy or stringy rot, or occur as pocket rot (advanced decay appearing in the form of a hole or pocket). White rot tends to produce more complete decomposition of the wood, and its decay products are much shorter lived (in the soil) than decay products produced by brown rots.

EPILOGUE

Whose woods these are, I think I know.
His house is in the village though;
He will not see me stopping here
To watch his woods fill up with snow.

My little horse must think it queer
To stop without a farmhouse near
Between the woods and frozen lake
The darkest evening of the year.

He gives his harness bells a shake
To ask if there is some mistake.
The only other sound's the sweep
Of easy wind and downy flake.

The woods are lovely, dark and deep.
But I have promises to keep,
And miles to go before I sleep,
And miles to go before I sleep.

Robert Frost, "Stopping by Woods on a Snowy Evening,"
from *The Poetry of Robert Frost*, copyright 1923.

> "***Go forth from the house in the village. Watch your woods fill up with snow on the darkest evening of the year. Or with the glory of the sunlight on the brightest morning. Listen to the sweep of easy wind and the faint fall of flakes. Listen to the many voices of the forest, the soft, serene, the violent, and natural sounds we sometimes hear but cannot understand. Let us share the promise and the joy, each in his own way, of the good and sweet earth, the woods and lake***."

Michael Frome, "*Whose Woods These Are: The Story of the National Forests*," published by Doubleday and Company, Inc.; copyright 1962 (360 pages).

APPENDIX 1: POTENTIAL VEGETATION COMPOSITION

The moist upland forest PVG includes moist mixed-conifer forests occurring in the upper montane vegetation zone (see fig. 3). Portions of three potential vegetation series (see fig. 4) are represented in the moist upland forest PVG – subalpine fir, grand fir, and Douglas-fir. Note that lodgepole pine and quaking aspen plant community types, representing successional (non-climax) stages of a plant association, are also common in the moist upland forest PVG – eight lodgepole pine and two aspen types are included in the list below (table 27).

Many users of the Blue Mountains potential vegetation classification system (Powell et al. 2007) have questioned the inclusion of subalpine fir plant associations in the moist upland forest PVG. These users generally assume that any subalpine fir type must occur in the subalpine vegetation zone (see fig. 3) and in the cold upland forest PVG. In response to these questions or concerns, a separate section was added to this white paper specifically to address their questions and concerns. It is included in the Ecological Setting section (section 2), specifically a sub-section called Moist Forest Classification Concepts in section 2.1 (see page 10).

Prime examples of moist forest in the northern Blue Mountains are associated with Level IV ecoregion unit 11c, the Maritime-Influenced Zone (fig. 91).

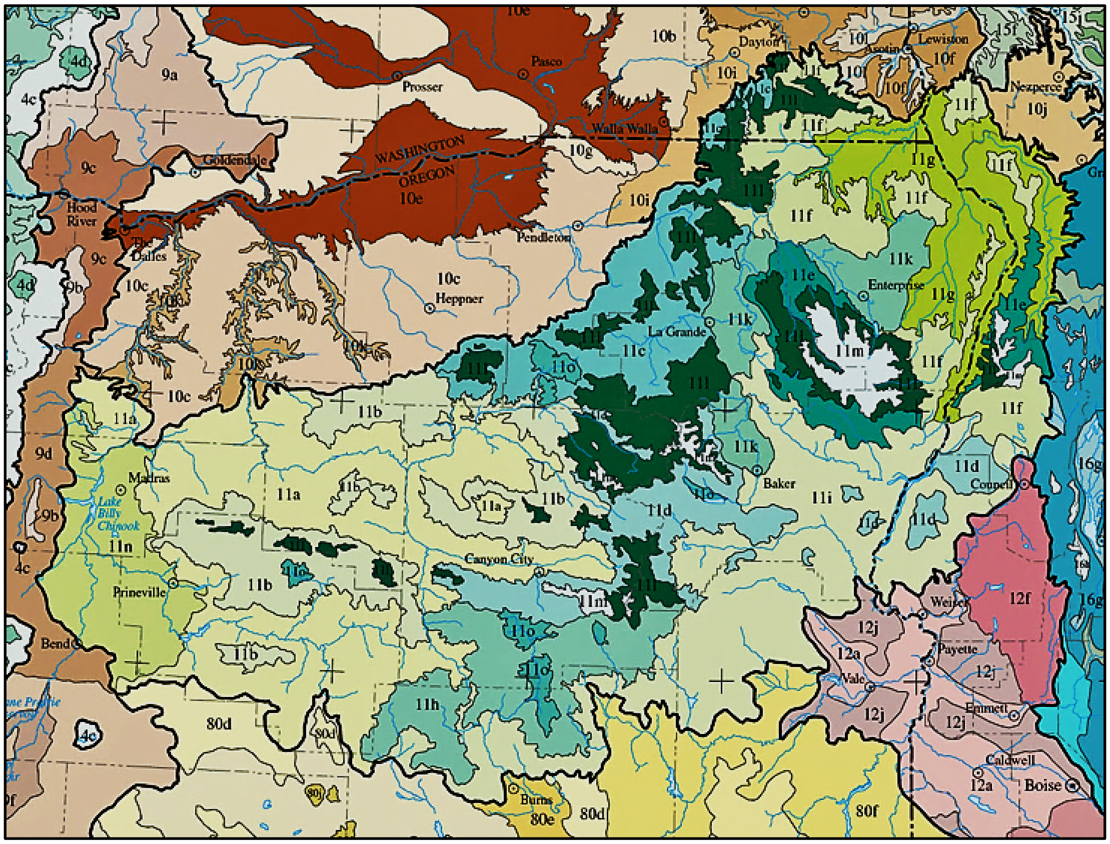


Figure 91 – Level IV ecoregions of the Blue Mountains.

Table 27: Potential vegetation type (PVT) codes and names, and plant association group (PAG) assignments, for the moist upland forest potential vegetation group (PVG).[1]

PVT Code	PVT Name	PAG
ABGR/ACGL	grand fir/Rocky Mountain maple	warm very moist
ABGR/ACGL-PHMA	grand fir/Rocky Mountain maple-mallow ninebark	warm moist
ABGR/BRVU	grand fir/Columbia brome	warm moist
ABGR/CLUN	grand fir/queencup beadlily	cool moist
ABGR/GYDR	grand fir/oakfern	cool very moist
ABGR/LIBO2	grand fir/twinflower	cool moist
ABGR/POMU-ASCA3	grand fir/swordfern-ginger	cool very moist
ABGR/TABR2/CLUN	grand fir/Pacific yew/queencup beadlily	cool wet
ABGR/TABR2/LIBO2	grand fir/Pacific yew/twinflower	cool wet
ABGR/TRCA	grand fir/false bugbane	cool very moist
ABGR/VAME	grand fir/big huckleberry	cool moist
ABGR/VASC-LIBO2	grand fir/grouse huckleberry-twinflower	cool moist
ABGR-CHNO/VAME	grand fir-Alaska yellow cedar/big huckleberry	cool moist
ABLA2/ARCO	subalpine fir/heartleaf arnica	cool moist
ABLA2/CLUN	subalpine fir/queencup beadlily	cool moist
ABLA2/LIBO2	subalpine fir/twinflower	cool moist
ABLA2/STAM	subalpine fir/claspleaf twistedstalk	cool wet
ABLA2/TRCA	subalpine fir/false bugbane	cool moist
ABLA2/VAME	subalpine fir/big huckleberry	cool moist
ABLA-PIEN/ARCO9	subalpine fir-Engelmann spruce/heartleaf arnica	cool moist
ABLA-PIEN/CLUN2	subalpine fir-Engelmann spruce/queencup beadlily	cool moist
ABLA-PIEN/LIBO3	subalpine fir-Engelmann spruce/twinflower	cool moist
ABLA-PIEN/TRCA	subalpine fir-Engelmann spruce/false bugbane	cool moist
PICO(ABGR)/ALSI	lodgepole pine(grand fir)/Sitka alder	cool very moist
PICO(ABGR)/LIBO2	lodgepole pine(grand fir)/twinflower	cool moist
PICO(ABGR)/VAME	lodgepole pine(grand fir)/big huckleberry	cool moist
PICO(ABGR)/VAME/CARU	lodgepole pine(grand fir)/big huckleberry/pinegrass	cool moist
PICO(ABGR)/VAME/PTAQ	lodgepole pine(grand fir)/big huckleberry/bracken fern	cool moist
PICO(ABGR)/VAME-LIBO2	lodgepole pine(grand fir)/big huckleberry-twinflower	cool moist
PICO(ABLA2)/VAME	lodgepole pine(subalpine fir)/big huckleberry	cool moist
PICO(ABLA2)/VAME/CARU	lodgepole pine(subalpine fir)/big huckleberry/pinegrass	cool moist
POTR5/CAGE2	quaking aspen/elk sedge	cool very moist
POTR5(ABGR)/HODI	quaking aspen(grand fir)/oceanspray	warm moist
PSME/ACGL-PHMA	Douglas-fir/Rocky Mountain maple-mallow ninebark	warm moist
PSME/ACGL-SYOR	Douglas-fir/Rocky Mountain maple-mtn. snowberry	warm moist
PSME/HODI	Douglas-fir/oceanspray	warm moist

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

The landscape classification portrayed in figure 91 is based on Omernik (1995). Unlike other systems, the Omernik approach uses only one classification unit – the ecoregion, with scale-based distinctions based on different hierarchical levels. To reach the Blue Mountains hierarchical level, here is the progression: Level I is unit 6.0: Northwestern Forested Mountains (this broad unit stretches from Alaska and the Yukon to New Mexico). Level II is unit 6.2: Western Cordillera (ranging from British Columbia to New Mexico). Level III is unit 6.2.9: Blue Mountains, extending from west-central Idaho (adjoining the Idaho Batholith and Northern Rockies ecoregions) to the Cascade Mountains (adjoining the Eastern Cascades Slopes and Foothills ecoregion). Level IV units are shown in figure 91, and summarized here:

Unit	**Description**
11a	John Day/Clarno Uplands: semi-arid foothills and low mountains surrounding western perimeter of the Blue Mountains.
11b	John Day/Clarno Highlands: dissected hills and low mountains supporting western ponderosa pine forest with a grass and shrub undergrowth.
11c	Maritime-Influenced Zone: portion of Blue Mountains that directly intercepts marine weather systems moving east through the Columbia River gorge.
11d	Mélange: dissected, mid-elevation mountains with a complex geology typical of mélange formations.
11e	Wallowas/Seven Devils Mountains: deeply dissected mountains in the Wallowa and Seven Devils ranges.
11f	Canyons and Dissected Highlands: steeply sloping, upper river canyons and dissected plateaus in the rain shadow of the mountains.
11g	Canyons and Dissected Uplands: deep river canyons and dissected plateaus at an elevation of 1,000 to 5,000 feet.
11h	Continental Zone Highlands: moderately dissected, mountainous volcanic plateau with scattered cinder cones at an elevation of 4,000 to 6,700 feet.
11i	Continental Zone Foothills: foothills, hills, and scattered buttes lying between the Blue and Wallowa mountains and the northwestern Snake River plain.
11k	Blue Mountain Basins: flat to rolling alluvial valleys containing floodplains, fluvial terraces, and scattered buttes in the Wallowa, Grande Ronde, and Baker valleys.
11l	Mesic Forest Zone: dissected volcanic plateau and mid-elevation mountains containing the highest forested areas in the Blue Mountains, western Wallowa Mountains, and western Seven Devils Mountains.
11m	Subalpine-Alpine Zone: high-elevation, glaciated mountains with aretes, cirques, and tarns.
11n	Deschutes River Valley: broad valley with deeply incised streams.
11o	Cold Basins: cold, wet valleys and basins, and wet meadows, at an elevation of 3,600 to 6,000 feet.

Note: Descriptions taken from "Blue Mountains (ecoregion)" article: http://en.wikipedia.org/wiki/Blue_Mountains_(ecoregion) (1/4/2014)

The Blue Mountains ecoregion unit has been divided into subregions or zones (Clarke and Bryce 1997). Good examples of moist forest in the northern Blue Mountains occur in the maritime-influenced zone and the mesic forest zone (fig. 91), both of which are influenced to a greater or lesser extent by marine weather systems moving east from the Pacific Ocean and through a low break in the Cascade Mountains – the Columbia River gorge (Mock 1996).

At a level below the maritime-influenced or mesic-forest zones, a typical moist-forest stand in the northern Blue Mountains (app. elevation 3,850 feet) occupies the plateau variant (instead of the canyons and dissected highlands variant). The plateau variant tends to have deep ash- or loess-dominated soils because they have been able to persist on flat, rolling, and north-facing slope positions (on other slope positions, natural erosion processes have often removed the ash). The soil temperature regime of the plateau variant is frigid, but the soil moisture regime is udic (moist). At slightly higher elevations to the east of this example stand (above 4,000 feet elevation), the soil temperature regime becomes colder in the cryic range.

One reason for delineation of a mesic-forest zone is presence of a soil ash layer above Columbia River basalt parent material. Because the Blue Mountains ecoregion is located south of Columbia Plateau ecoregion, much fine-textured ('ashy') material is actually wind-deposited loess originating from glacial activity or glacial Lake Missoula flood events occurring in or near the Columbia Plateau in eastern Washington. When ash is abundant (rather than loess), it is typically derived from Mount Mazama (Crater Lake) eruptions in southwestern Oregon, but Glacier Peak ash from northeastern Washington is also found (Foit et al. 1993, Fryxell 1965).

At this point, we have arrived hierarchically at the level of a typical moist-forest stand, located in the plateau variant of either the maritime-influenced or mesic forest zones (instead of canyons and dissected highlands, a variant generally found on steeper side slopes lacking deep, fine-textured soil mantles). Our example stand has even or gently rolling landforms, and slope gradients seldom exceeding 20 percent, with much of the stand being either flat or having slope gradients of 5% or less (if slopes greatly exceeded 20%, then the stand would be assigned to the canyons and dissected highlands variant). The stand's lack of topographic variation results in its ecological gradients being gradual and continuous, rather than abrupt and exhibiting sharply defined boundaries.

For a typical moist-forest stand in the northern Blue Mountains, here is the hierarchical progression, moving downward from the broadest unit to the finest unit:

- Northwestern Forested Mountains
  - Western Cordillera
    - Blue Mountains
      - Maritime-Influenced Zone
        - Plateau Variant
          - Moist Upland Forest Stand

APPENDIX 2: REGENERATION MONITORING RESULTS

This appendix provides regeneration monitoring results for moist upland forests of the Umatilla National Forest (Table 28). It summarizes tree density (stems per acre), by species as grouped by seral status, for 102 plots established in plantations located on the Moist Upland Forest potential vegetation group (94 plots are Managed Stand Survey installations; 8 plots were established during a Forest Plan review of regeneration results). The Sources/Notes section at the end of the table provides additional information about the 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 the Moist Upland Forest PVG overall).

This table provides monitoring information to inform moist-forest prescriptions for tree planting (reforestation). Planting is one of the highest-cost activities in the vegetation management realm – when considering both internal costs (contract administration, seed procurement, etc.) and external costs (service contract, seedlings, etc.), total planting costs often run from \$300 to \$500 per acre.

As budgetary resources continue to decline in the future, it is important to consider reforestation options that could be implemented at lower cost. The regeneration monitoring data presented here demonstrates that moist-forest sites tend to support abundant amounts of natural regeneration, and that much of the regeneration has relatively high levels of species diversity.

When considering the Moist Upland Forest PVG as a whole (see "Mean: Moist Upland Forest PVG" row at bottom of table 28), true firs have the highest average density levels (subalpine fir averaged 668 stems per acre and grand fir averaged 650 stems per acre). High amounts of true fir regeneration are not surprising because its seed rain and seedling frequency 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 the representation of true firs can be justified as a counterbalance to their regeneration proficiency.

The National Forest Management Act of 1976 (P.L. 94-588) (NFMA) states that when trees are cut to achieve timber production objectives, the 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)). Although this statement has been interpreted in various ways, it does not mean that reforestation (tree planting) must occur within 5 years of timber harvest (Watrud et al. 2012).

The NFMA statement quoted above means that we should not harvest areas where previous experience suggests that restocking will generally not occur in 5 years, regardless of whether the 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.”

The 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 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 moist upland forest receiving a regeneration cutting method, but it should be informed by the regeneration monitoring results presented in table 28. Those results suggest that natural regeneration is often abundant on moist upland forest sites, but that most of it is comprised of late-seral tree species. Therefore, tree planting could be prescribed to establish an ecologically appropriate forest cover, which is defined as a proper mix of early- and mid-seral tree species in the context of an early stage of plant succession (Swanson et al. 2011).

Note that tree planting recommendations are provided, by plant association group, in table 25 of this white paper.

Forest Vegetation Simulator (FVS) Regeration Modeling Considerations

None of the Pacific Northwest variants of FVS contain a regeneration establishment model. If they did, FVS would periodically interject ‘background’ levels of natural regeneration (e.g., ingrowth), and the composition and amount 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, we need to add it manually by using either the ‘natural’ or ‘plant’ keywords. The regeneration monitoring results presented in this appendix provide a reasonable basis for formulating credible ingrowth scenarios for moist upland forest of the northern Blue Mountains (Umatilla National Forest).

[Note: The Managed Stand Survey (MSS) process was initiated in the late 1980s. Initial installations (1-acre plots) were installed in young, managed stands throughout the Pacific Northwest Region. One reason for initiating the MSS program was to obtain long-term information about ingrowth and young-stand development, and then use it to calibrate FVS. Unfortunately, the MSS plots were never remeasured.]

Table 28: Regeneration monitoring results for moist upland forests of the Umatilla National Forest.

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 —————————>													
2778	GF/ACGL	WVM		52		47	99	40		40		185		185		324
2832	GF/ACGL	WVM		24		64	88	31		31		309		309		428
	Mean: GF/ACGL			38		56	94	36		36		247		247		376
2753	GF/CLUN	ClM		141		29	170				624	213		837		1007
2755	GF/CLUN	ClM		152	81	53	286	191		191	64	2051		2115		2592
2756	GF/CLUN	ClM		41			41	44		44	12	148		160		245
2757	GF/CLUN	ClM			4	40	44	92		92	192	237		429	20	585
2758	GF/CLUN	ClM		64		20	84	60		60	277	1299		1576		1720
2768	GF/CLUN	ClM		79		91	170	8		8		157		157	4	339
2785	GF/CLUN	ClM				4	4				120	536		656		660
2798	GF/CLUN	ClM		16	8	80	104				928	1605	68	2601		2705
2822	GF/CLUN	ClM				20	20	4		4	1095	407		1502		1526
2827	GF/CLUN	ClM				321	321	199	44	243	969	1924		2893	40	3497
2828	GF/CLUN	ClM						20		20	20	556		576		596
	Mean: GF/CLUN			82	31	73	124	77	44	83	430	830	68	1227	21	1407
2010	GF/LIBO2	ClM	20	20	80	804	924	144		144	517	941		1458		2526
2017	GF/LIBO2	ClM		20	408	108	536	28		28	545	425		970		1534
2254	GF/LIBO2	ClM		131		4	135	40		40	28	200		228		403
2257	GF/LIBO2	ClM			581	20	601	560		560		1324		1324		2485
2258	GF/LIBO2	ClM		97	84	8	189	180		180		100		100		469
2261	GF/LIBO2	ClM		119	28	160	307	255		255		1163		1163		1725
2263	GF/LIBO2	ClM				4	4	44		44	4	433		437		485
2268	GF/LIBO2	ClM		336	717		1053	64		64	60	915		975		2092
2512	GF/LIBO2	ClM		20	413	1087	1520	283		283		492		492		2295
2529	GF/LIBO2	ClM			1212	47	1259	104		104		2319		2319		3682
2533	GF/LIBO2	ClM		8	915	184	1107	177		177		4349		4349		5633
2750	GF/LIBO2	ClM		57	32	157	246	20		20		88		88		354
2759	GF/LIBO2	ClM				20	20				85	83		168		188
2760	GF/LIBO2	ClM		68	1004	172	1244	84		84	697	1191		1888		3216

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 ————————>													
2763	GF/LIBO2	ClM				967	967	44		44	665	201		866		1877
2775	GF/LIBO2	ClM				8	8				72	232		304		312
2783	GF/LIBO2	ClM				56	56	80		80	600	779		1379		1515
2784	GF/LIBO2	ClM				8	8	32		32	164	144		308		348
2786	GF/LIBO2	ClM		20		108	128	104		104	567	1117		1684		1916
2818	GF/LIBO2	ClM		195	4	4	203	328		328	400	1145		1545		2076
2824	GF/LIBO2	ClM			8	44	52	84		84	36	1224		1260		1396
2825	GF/LIBO2	ClM		27	24	224	275	39	20	59	95	1631		1726		2060
2829	GF/LIBO2	ClM		16	33	25	74	76		76	4	428		432	4	586
UMA11	GF/LIBO2	ClM		129	86	386	601									601
UMA13	GF/LIBO2	ClM		120	20	200	340	30		30	10	10		20		390
UMA14	GF/LIBO2	ClM		240	40	100	380	150		150						530
UMA16	GF/LIBO2	ClM		75	13	6	94	6		6	88	13		101		201
UMA2	GF/LIBO2	ClM		155	70		225	40		40						265
UMA9	GF/LIBO2	ClM		50	160	120	330	40		40	90	30		120		490
	Mean: GF/LIBO2		20	95	282	186	444	117	20	118	249	807		989	4	1436
2752	GF/TABR/CLUN	ClW				12	12				276	571		847	44	903
2754	GF/TABR/CLUN	ClW				75	75	4		4	204	184		388	4	471
2764	GF/TABR/CLUN	ClM						20		20	47	71		118	28	166
2765	GF/TABR/CLUN	ClM		43		4	47	52		52	180	221		401	884	1384
2767	GF/TABR/CLUN	ClM		24		107	131	225		225	541	1535		2076	1000	3432
2769	GF/TABR/CLUN	ClM		52			52	16		16	40	739		779	456	1303
2770	GF/TABR/CLUN	ClM		227		60	287					848		848	80	1215
2788	GF/TABR/CLUN	ClM		40	20	184	244	72		72	68	627		695		1011
2789	GF/TABR/CLUN	ClM									100	257		357	16	373
	Mean: GF/TABR/CLUN			77	20	74	121	65		65	182	561		723	314	1140
2800	GF/TRCA3	CVM		21		11	32	24		24	28	673	20	721		777
2837	GF/TRCA3	CVM			53		53	80		80	147	712		859		992
	Mean: GF/TRCA3			21	53	11	43	52		52	88	693	20	790		885
2256	GF/VAME	ClM		360	20	20	400	80		80	100	980		1080		1560

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 ————————>													
2262	GF/VAME	ClM		35	180	24	239	1432		1432	985	2721	20	3726		5397
2267	GF/VAME	ClM			16	72	88	97		97	273	109	60	442		627
2269	GF/VAME	ClM		399	204	20	623	364	8	372	500	1720	20	2240		3235
2272	GF/VAME	ClM		19	68	11	98	444		444	224	1875		2099		2641
2273	GF/VAME	ClM			148	4	152	100		100	1124	1127	112	2363		2615
2516	GF/VAME	ClM			60	63	123	168		168	12	737		749		1040
2532	GF/VAME	ClM		60	2484	40	2584	149		149		601		601		3334
2751	GF/VAME	ClM		124	28	40	192	124		124	60	377		437		753
2761	GF/VAME	ClM		153			153	16		16	4	36		40		209
2762	GF/VAME	ClM		23			23					213		213	40	276
2766	GF/VAME	ClM		59		72	131	147		147	4	516		520		798
2771	GF/VAME	ClM		247		4	251	33		33	88	680		768	240	1292
2774	GF/VAME	ClM		160		24	184	64		64	332	1639		1971	40	2259
2787	GF/VAME	ClM				269	269	87		87	532	665		1197		1553
2795	GF/VAME	ClM		204		7	211	55		55	112	1024		1136		1402
2816	GF/VAME	ClM			333	557	890	24		24	1059	239		1298		2212
2820	GF/VAME	ClM			963		963				60	1260		1320		2283
2826	GF/VAME	ClM		140		73	213	120		120		209		209		542
2830	GF/VAME	ClM		131	20	1209	1360	43		43	20	87		107		1510
2833	GF/VAME	ClM		39		24	63				4	88	4	96		159
2834	GF/VAME	ClM		77		11	88	8		8	44	181		225		321
2835	GF/VAME	ClM		4		4	8	53		53	108	729		837		898
	Mean: GF/VAME			131	377	127	405	180	8	181	282	774	43	1029	107	1605
2019	GF/VASC-LIBO2	ClM		27		120	147	60		60	176	736		912	8	1127
UMA12	GF/VASC-LIBO2	ClM		150	30	10	190	50	20	70		30		30		290
	Mean: GF/VASC-LIBO2			89	30	65	169	55	20	65	176	383		471	8	709
2264	SF/VAME	ClM			65		65	68		68	504	60	1096	1660		1793
2265	SF/VAME	ClM			168		168	4		4	1083	1020	1424	3527		3699
2266	SF/VAME	ClM			343	35	378	64		64	124	319	1299	1742		2184
2270	SF/VAME	ClM			4	8	12				103		345	448		460

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 ————————>													
2271	SF/VAME	ClM		40	120	216	376	20		20	1228	371	1287	2886		3282
2773	SF/VAME	ClM			75		75				563	204	591	1358		1433
2776	SF/VAME	ClM		8	28		36				349	4	169	522		558
2777	SF/VAME	ClM			32		32				587	20	2240	2847		2879
2782	SF/VAME	ClM			8	8	16				265	20	1641	1926		1942
2790	SF/VAME	ClM		4	643	108	755				1348		2091	3439		4194
2791	SF/VAME	ClM		4	247	28	279				1376	420	404	2200		2479
2792	SF/VAME	ClM		8	863	4	875	40		40	708	120	248	1076		1991
2793	SF/VAME	ClM		8	72		80				173	60	136	369		449
2794	SF/VAME	ClM			231		231				144		52	196		427
2796	SF/VAME	ClM		4	92	36	132				236	744	235	1215		1347
2797	SF/VAME	ClM		36	128	52	216				1672	20	1419	3111		3327
2799	SF/VAME	ClM			8	52	60				595	904	800	2299		2359
2801	SF/VAME	ClM		8	152	4	164				183	468	1084	1735		1899
2802	SF/VAME	ClM			24	8	32	24		24	264	227	656	1147		1203
2803	SF/VAME	ClM			168	28	196				540	120	873	1533		1729
2817	SF/VAME	ClM			12	119	131				585		1045	1630		1761
2819	SF/VAME	ClM				4	4				65	40	353	458		462
2821	SF/VAME	ClM		153	320	120	593	20		20	2537	360	509	3406		4019
UMA10	SF/VAME	ClM			1020	60	1080						420	420		1500
	Mean: SF/VAME			27	210	52	249	34		34	662	290	851	1715		1974
Mean: Moist Upland Forest PVG			20	88	250	117	312	112	23	113	382	650	668	1147	182	1526

Sources/Notes: **Plot** includes two types of plots: numbers refer to plots from the Managed Stand Survey (MSS), which is a plot-based system (a 5-point plot cluster covering about 1 acre) installed in 1990 in young, managed stands with an average 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 consisting of a 2-digit tree species (GF = grand fir; SF = subalpine fir) and a 4- or 5-digit understory species code (ACGL = Rocky Mountain maple; CLUN = queencup beadlily; LIBO2 = twinflower; TABR = Pacific yew; TRCA3 = false bugbane; VAME = big huckleberry; VASC = grouse huckleberry). **PAG** refers to plant association group (ClVM is cool very moist; ClM is cool moist; WVM is warm very moist; see Powell et al. 2007). Columns are provided for individual tree species (in addition to the 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 the sum of the preceding four species columns; **Mid Seral** is the sum of the preceding two columns; **Late Seral** is the sum of the preceding three columns; other species includes Pacific yew, hawthorn, willow, and paper birch. **Total** is a summed tree density value, as trees per acre, for the 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 the disturbance process (including 'advance' regeneration and mature trees from the previous stand that survived the disturbance process). Note that Mean values are presented for each plant association, and at the end of the table for the Moist 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 the Mean values).

MOIST-FOREST REFERENCES AND LITERATURE CITED

This section includes literature cited in the white paper, along with other references having relevance to the ecology and active management of moist forests in the Blue Mountains of northeastern Oregon and southeastern Washington.

Cautionary note about moist-forest references: active management of moist forest is an emerging issue across the interior Pacific Northwest and northern Rocky Mountains regions, so literature citations relating to moist-forest ecosystems may span this entire geographical area.

I believe it is useful for practitioners to be aware of the full breadth of moist-forest research, but it is also important to recognize that in the context of this white paper, place-based research pertaining specifically to the Blue Mountains and adjacent portions of the interior Northwest qualifies as primary sources, while work from the northern Rockies (particularly for areas located west of the Continental Divide) qualifies as useful secondary sources, and work from elsewhere in the western United States would be tertiary sources.

Note: I believe Sierra Nevada research is more appropriate for the summer-dry Mediterranean climate of the Blue Mountains than moist-forest work from the monsoon climates of the southern Rocky Mountains.

With few exceptions, sources contained in this References section are available from the 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 there.

When preparing a white paper, one of my objectives is to help users locate any of its references or 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 generally provided, although I realize that weblinks have not been stable (USDA Forest Service Treesearch links, however, have been quite stable thus far).

Aber, J.; Christensen, N.; Fernandez, I.; Franklin, J.; Hidinger, L.; Hunter, M.; MacMahon, J.; Mladenoff, D.; Pastor, J.; Perry, D.; Slangen, R.; van Miegroet, H. 2000. Applying ecological principles to management of the U.S. national forests. Issues in Ecology, Number 6. Washington, DC: Ecological Society of America. 20 p. http://www.esa.org/science_resources/issues/FileEnglish/issue6.pdf

Adamowicz, W.L.; Burton, P.J. 2003. Sustainability and sustainable forest management. Chapter 2. In: Burton, P.J.; Messier, C.; Smith, D.W.; Adamowicz, W.L., eds. Towards sustainable management of the boreal forest. Ottawa, ON: NRC Research Press: 41-64. isbn:0-660-18762-0

Adams, J.A. 1978. Long-term aspects of nutrient loss from forest soils and ecosystems. New Zealand Journal of Forestry. 23(1): 10-20. https://pdfs.semanticscholar.org/1d46/e56fe77f74be46ce1a7a3fd4bcc3a7013094.pdf

Adams, D.L. 1980. The northern Rocky Mountain region. In: Barrett, J.W., ed. Regional silviculture of the United States. New York: John Wiley & Sons: 341-389. isbn:0-471-05645-6

Adams, D.L.; Ferguson, D.E. 1990. A forest regeneration mystery–the grand fir mosaic. Focus on Renewable Natural Resources. 15: 21-22.

Adams, D.M.; Latta, G.S. 2005. Costs and regional impacts of restoration thinning programs on the national forests in eastern Oregon. Canadian Journal of Forest Research. 35(6): 1319-1330. doi:10.1139/x05-065

Adams, H.D.; Guardiola-Claramonte, M.; Barron-Gafford, G.A.; Villegas, J.C.; Breshears, D.D.; Zou, C.B.; Troch, P.A.; Huxman, T.E. 2009. Temperature sensitivity of drought-induced tree mortality portends increased regional die-off under global-change-type drought. Proceedings of the National Academy of Sciences. 106(17): 7063-7066. doi:10.1073/pnas.0901438106

Agee, J.K. 1990. The historical role of fire in Pacific Northwest forests. In: Walstad, J.D.; Radosevich, S.R.; Sandberg, D.V., eds. Natural and prescribed fire in Pacific Northwest forests. Corvallis, OR: Oregon State University Press: 25-38. isbn:0-87071-359-0

Agee, J.K. 1993. Fire ecology of Pacific Northwest forests. Washington, DC: Island Press. 493 p. isbn:1-55963-229-1

Agee, J.K. 1994. Fire and weather disturbances in terrestrial ecosystems of the eastern Cascades. Gen. Tech. Rep. PNW-GTR-320. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 52 p. http://www.treesearch.fs.fed.us/pubs/6225

Agee, J.K. 1996a. Achieving conservation biology objectives with fire in the Pacific Northwest. Weed Technology. 10(2): 417-421. http://www.jstor.org/stable/info/3988077

Agee, J.K. 1996b. Fire in the Blue Mountains: a history, ecology, and research agenda. In: Jaindl, R.G.; Quigley, T.M., eds. Search for a solution: sustaining the land, people, and economy of the Blue Mountains. Washington, DC: American Forests: 119-145. https://ecoshare.info/uploads/ccamp/synthesis_paper_tools/huckleberry/Agee_1996.pdf

Agee, J.K. 1996c. The influence of forest structure on fire behavior. In: Proceedings of the seventeenth annual forest vegetation management conference; 1996 January 16-18; Redding, CA: 52-68. http://www.fvmc.org/PDF/FVMCProc17th(1996).pdf

Agee, J.K. 1997a. Fire management for the 21st century. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press: 191-201. isbn:1-55963-399-9

Agee, J.K. 1997b. The severe weather wildfire – too hot to handle? Northwest Science. 71(1): 153-156. The_severe_weather_wildfire_-_Too_hot_to_handle

Agee, J.K. 1998. The landscape ecology of western forest fire regimes. Northwest Science. 72(Special Issue): 24-34. http://digitalcommons.usu.edu/cgi/viewcontent.cgi?article=1305&context=barkbeetles

Agee, J.K. 1999. Fire effects on landscape fragmentation in interior West forests. In: Rochelle, J.A.; Lehmann, L.A.; Wisniewski, J., eds. Forest fragmentation: wildlife and management implications. Leiden, Netherlands: Brill Academic Publishers: 43-60. isbn:90-04-11388-6

Agee, J.K. 2002a. Fire as a coarse filter for snags and logs. In: Laudenslayer, W.F., Jr.; Shea, P.J.; Valentine, B.E.; Weatherspoon, C.P.; Lisle, T.E., eds. Proceedings of the symposium on the ecology and management of dead wood in western forests. Gen. Tech. Rep. PSW-GTR-181. Albany, CA: USDA Forest Service, Pacific Southwest Research Station: 359-368. http://www.treesearch.fs.fed.us/pubs/6718

Agee, J.K. 2002b. The fallacy of passive management: managing for firesafe forest reserves. Conservation Biology In Practice. 3(1): 18-25. doi:10.1111/j.1526-4629.2002.tb00023.x

Agee, J.K. 2003. Historical range of variability in eastern Cascade forests, Washington, USA. Landscape Ecology. 18(8): 725-740. doi:10.1023/B:LAND.0000014474.49803.f9

Agee, J.K. 2005. The complex nature of mixed severity fire regimes. In: Taylor, L.; Zelnik, J.; Cadwallader, S.; Hughes, B., eds. Mixed severity fire regimes: ecology and management. AFE MISC03. Pullman, WA: Washington State University Extension: 1-10.

Agee, J.K.; Maruoka, K.R. 1994. Historical fire regimes of the Blue Mountains. Tech Notes BMNRI-TN-1. La Grande, OR: USDA Forest Service, Blue Mountains Natural Resources Institute. 4 p.

Agee, J.K.; Skinner, C.N. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management. 211(1-2): 83-96. doi:10.1016/j.foreco.2005.01.034

Agee, J.K.; Bahro, B.; Finney, M.A.; Omi, P.N.; Sapsis, D.B.; Skinner, C.N.; van Wagtendonk, J.W.; Weatherspoon, C.P. 2000. The use of shaded fuelbreaks in landscape fire management. Forest Ecology and Management. 127(1-3): 55-66. doi:10.1016/S0378-1127(99)00116-4

Ager, A.A.; McGaughey, R.J. 1997. UTOOLS: microcomputer software for spatial analysis and landscape visualization. Gen. Tech. Rep. PNW-GTR-397. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 15 p. http://www.treesearch.fs.fed.us/pubs/3023

Ager, A.A.; Vaillant, N.M.; Finney, M.A. 2010. A comparison of landscape fuel treatment strategies to mitigate wildland fire risk in the urban interface and preserve old forest structure. Forest Ecology and Management. 259(8): 1556-1570. doi:10.1016/j.foreco.2010.01.032

Ager, A.A.; Vaillant, N.M.; Finney, M.A. 2011. Integrating fire behavior models and geospatial analysis for wildland fire risk assessment and fuel management planning. Journal of Combustion. 2011(572452): 1-19. doi:10.1155/2011/572452

Ager, A.A.; Vaillant, N.M.; Finney, M.A.; Preisler, H.K. 2012. Analyzing wildfire exposure and source–sink relationships on a fire prone forest landscape. Forest Ecology and Management. 267: 271-283. doi:10.1016/j.foreco.2011.11.021

Ager, A.A.; Barros, A.M.G.; Day, M.A.; Preisler, H.K.; Spies, T.A.; Bolte, J. 2018. Analyzing fine-scale spatiotemporal drivers of wildfire in a forest landscape model. Ecological Modelling. 384: 87-102. doi:10.1016/j.ecolmodel.2018.06.018

Ager, A.A.; Houtman, R.M.; Day, M.A.; Ringo, C.; Palaiologou, P. 2019. Tradeoffs between US national forest harvest targets and fuel management to reduce wildfire transmission to the wildland urban interface. Forest Ecology and Management. 434: 99-109. doi:10.1016/j.foreco.2018.12.003

Ager, A.A.; Barros, A.M.G.; Houtman, R.; Seli, R.; Day, M.A. 2020. Modelling the effect of accelerated forest management on long-term wildfire activity. Ecological Modelling. 421: 108962 (13 p). doi:10.1016/j.ecolmodel.2020.108962

Aho, P.E.; Fiddler, G.; Filip, G.M. 1983. How to reduce injuries to residual trees during stand management activities. Gen. Tech. Rep. PNW-156. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 17 p. https://www.fs.usda.gov/treesearch/pubs/7697

Aho, P.E.; Filip, G.M.; Lombard, F.F. 1987. Decay fungi and wounding in advance grand and white fir regeneration. Forest Science. 33(2): 347-355. doi:10.1093/forestscience/33.2.347

Aitken, S.N.; Yeaman, S.; Holliday, J.A.; Wang, T.; Curtis-McLane, S. 2008. Adaptation, migration or extirpation: climate change outcomes for tree populations. Evolutionary Applications. 1(1): 95-111. doi:10.1111/j.1752-4571.2007.00013.x

Alberti, G.; Nock, C.; Fornasier, F.; Scherer-Lorenzen, M.; De Nobili, M.; Peressotti, A.; Hoenig, L.; Bruelheide, H.; Bauhus, J. 2017. Tree functional diversity influences belowground ecosystem functioning. Applied Soil Ecology. 120: 160-168. doi:10.1016/j.apsoil.2017.07.038

Alexander, R.R. 1987. Ecology, silviculture, and management of the Engelmann spruce-subalpine fir type in the central and southern Rocky Mountains. Agric. Handb. No. 659. Washington, DC: USDA Forest Service. 144 p. http://digitalcommons.usu.edu/cgi/viewcontent.cgi?article=1144&context=barkbeetles

Alexander, R.R.; Edminster, C.B. 1980a. Lodgepole pine management in the central Rocky Mountains. Journal of Forestry. 78(4): 196-201. doi:10.1093/jof/78.4.196

Alexander, R.R.; Edminster, C.B. 1980b. Management of spruce-fir in even-aged stands in the central Rocky Mountains. Res. Pap. RM-217. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 14 p. https://www.fs.usda.gov/treesearch/pubs/53277

Alexander, M.E.; Hawksworth, F.G. 1975. Wildland fires and dwarf mistletoes: a literature review of ecology and prescribed burning. Gen. Tech. Rep. RM-14. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 12 p. https://www.fs.usda.gov/treesearch/pubs/40436

Alexander, R.R.; Shepperd, W.D. 1990. *Picea engelmannii* Parry ex. Engelm.; Engelmann spruce. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America; volume 1, conifers. Agric. Handb. 654. Washington, DC: USDA Forest Service: 187-203.
https://www.treesearch.fs.fed.us/pubs/1547
https://www.srs.fs.usda.gov/pubs/misc/ag_654/volume_1/picea/engelmannii.htm

Alfaro, R.; Axelson, J.; Hawkes, B. 2008. Mountain pine beetle increases the complexity of fire-origin lodgepole pine stands in British Columbia, Canada. BC Journal of Ecosystems and Management. 9(3): 85-89. https://jem-online.org/index.php/jem/article/download/402/317

Alfaro, R.I.; Battisti, A.; Carroll, A.; Fleming, R.; Hantula, J. 2010. Forest health in a changing environment. In: Mery, G.; Katila, P.; Galloway, G.; Alfaro, R.I.; Kanninen, M.; Lobovikov, M.; Varjo, J., eds. Forests and society – responding to global drivers of change. IUFRO World Series Volume 25. Vienna, Austria: International Union of Forest Research Organizations: 113-134. https://www.iufro.org/fileadmin/material/publications/iufro-series/ws25/113-134.pdf

Allaby, Michael, ed. 1998. The concise Oxford dictionary of ecology. Second edition. New York: Oxford University Press. 440 p. isbn:0-19-280078-7

Allbrook, R.F. 1986. Effect of skid trail compaction on a volcanic soil in central Oregon. Soil Science Society of America Journal. 50(5): 1344-1346. doi:10.2136/sssaj1986.03615995005000050052x

Allen, C.R.; Gunderson, L.H. 2011. Pathology and failure in the design and implementation of adaptive management. Journal of Environmental Management. 92(5): 1379-1384. doi:10.1016/j.jenvman.2010.10.063

Allen, T.F.H.; Wyleto, E.P. 1983. A hierarchical model for the complexity of plant communities. Journal of Theoretical Biology. 101(4): 529-540. doi:10.1016/0022-5193(83)90014-0

Allen, J.E.; Burns, M.; Sargent, S.C. 2002. Cataclysms on the Columbia: a layman's guide to the features produced by the catastrophic Bretz floods in the Pacific Northwest. Portland, OR: Timber Press. 213 p. isbn:0-88192-215-3

Allen, C.D.; Macalady, A.K.; Chenchouni, H.; Bachelet, D.; McDowell, N.; Vennetier, M.; Kitzberger, T.; Rigling, A.; Breshears, D.D.; Hogg, E.H.; Gonzalez, P.; Fensham, R.; Zhang, Z.; Castro, J.; Demidova, N.; Lim, J.-H.; Allard, G.; Running, S.W.; Semerci, A.; Cobb, N. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management 259(4):660-684. doi:10.1016/j.foreco.2009.09.001

Amaranthus, M.P. 1997. Forest sustainability: an approach to definition and assessment at the landscape level. Gen. Tech. Rep. PNW-GTR-416. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 14 p. https://www.fs.usda.gov/treesearch/pubs/3246

Amaranthus, M.P.; Perry, D.A. 1994. The functioning of ectomycorrhizal fungi in the field: linkages in space and time. Plant and Soil. 159(1): 133-140. doi:10.1007/bf00000102

Amaranthus, M.P.; Page-Dumroese, D.; Harvey, A.; Cazares, E.; Bednar, L.F. 1996. Soil compaction and organic matter affect conifer seedling nonmycorrhizal and ectomycorrizal root tip abundance and diversity. Res. Pap. PNW-RP-494. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 12 p. https://www.fs.usda.gov/treesearch/pubs/26874

Amman, G.D. 1977. The role of the mountain pine beetle in lodgepole pine ecosystems: impact on succession. In: Mattson, W.J., ed. The role of arthropods in forest ecosystems. New York: Springer-Verlag: 3-18.

Amman, G.D., comp. 1989. Proceedings–symposium on the management of lodgepole pine to minimize losses to the mountain pine beetle. Gen. Tech. Rep. INT-262. Ogden, UT: USDA Forest Service, Intermountain Research Station. 119 p. https://archive.org/download/CAT90949101/CAT90949101.pdf

Amman, G.D.; Anhold, J.A. 1989. Preliminary evaluation of hazard and risk rating variables for mountain pine beetle infestations in lodgepole pine stands. In: Amman, G.D., comp. Proceedings–symposium on the management of lodgepole pine to minimize losses to mountain pine beetle. Gen. Tech. Rep. INT-262. Ogden, UT: USDA Forest Service, Intermountain Research Station: 22-27. https://archive.org/download/CAT90949101/CAT90949101.pdf

Amman, G.D.; Safranyik, L. 1985. Insects of lodgepole pine: impacts and control. In: Baumgartner, D.M.; Krebill, R.G.; Arnott, J.T.; Weetman, G.F., comps. Lodgepole pine: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Cooperative Extension, Office of Conferences and Institutes 107-124.

Amman, G.D.; Lessard, G.D.; Rasmussen, L.A.; O'Neil, C.G. 1988. Lodgepole pine vigor, regeneration, and infestation by mountain pine beetle following partial cutting on the Shoshone National Forest, Wyoming. Res. Paper INT-396. Ogden, UT: USDA Forest Service, Intermountain Research Station. https://archive.org/download/lodgepolepinevig396amma/lodgepolepinevig396amma.pdf

Amman, G.D.; McGregor, M.D.; Schmitz, R.F.; Oakes, R.D. 1988. Susceptibility of lodgepole pine to infestation by mountain pine beetles following partial cutting of stands. Canadian Journal of Forest Research. 18(6): 688-695. doi:10.1139/x88-105

Amoroso, M.M.; Daniels, L.D.; Bataineh, M.; Andison, D.W. 2011. Evidence of mixed-severity fires in the foothills of the Rocky Mountains of west-central Alberta, Canada. Forest Ecology and Management. 262(12): 2240-2249. doi:10.1016/j.foreco.2011.08.016

Amoroso, M.M.; Coates, K.D.; Astrup, R. 2013. Stand recovery and self-organization following large-scale mountain pine beetle induced canopy mortality in northern forests. Forest Ecology and Management. 310: 300-311. doi:10.1016/j.foreco.2013.08.037

Anderson, D.W. 1988. The effect of parent material and soil development on nutrient cycling in temperate ecosystems. Biogeochemistry. 5(1): 71-97. doi:10.1007/BF02180318

Anderson, M.K. 1996. Tending the wilderness. Restoration and Management Notes. 14(2): 154-166. https://www.jstor.org/stable/43440247

Anderson, M.D. 2003. *Pinus contorta* var. *latifolia*. In: Fire effects information system [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us/database/feis/plants/tree/pinconl/all.html

Anderson, M.K. 2006. Tending the wild: Native American knowledge and the management of California's natural resources. Berkeley, CA: University of California Press. 526 p. isbn:978-0-520-24851-9

Anderson, J.E.; Romme, W.H. 1991. Initial floristics in lodgepole pine *(Pinus contorta)* forests following the 1988 Yellowstone fires. International Journal of Wildland Fire. 1(2): 119-124. doi:10.1071/WF9910119

Anderson, L.; Carlson, C.E.; Wakimoto, R.H. 1987. Forest fire frequency and western spruce budworm outbreaks in western Montana. Forest Ecology and Management. 22(3-4): 251-260. doi:10.1016/0378-1127(87)90109-5

Anderson-Teixeira, K.J.; Miller, A.D.; Mohan, J.E.; Hudiburg, T.W.; Duval, B.D.; DeLucia, E.H. 2013. Altered dynamics of forest recovery under a changing climate. Global Change Biology. 19(7): 2001-2021. doi:10.1111/gcb.12194

Andrews, P.L.; Butler, B.W. 2006. Fuels management—how to measure success: conference proceedings. Proceedings RMRS-P-41. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 809 p. https://www.fs.usda.gov/treesearch/pubs/24476

Angelstam, P. 1997. Landscape analysis as a tool for the scientific management of biodiversity. Ecological Bulletins. 46: 140-170. https://www.jstor.org/stable/20113213

Angelstam, P.K. 1998. Maintaining and restoring biodiversity in European boreal forests by developing natural disturbance regimes. Journal of Vegetation Science. 9(4): 593-602. doi:10.2307/3237275

Angelstam, P. 2003. Reconciling the linkages of land management with natural disturbance regimes to maintain forest biodiversity in Europe. In: Bissonette, J.A.; Storch, I., eds. Landscape ecology and resource management: linking theory with practice. Washington, DC: Island Press: 193-226. isbn:1-55963-973-3

Anhold, J.A.; Jenkins, M.J. 1987. Potential mountain pine beetle (Coleoptera: Scolytidae) attack of lodgepole pine as described by stand density index. Environmental Entomology. 16(3): 738-742. doi:10.1093/ee/16.3.738

Anhold, J.A.; Jenkins, M.J.; Long, J.N. 1996. Management of lodgepole pine stand density to reduce susceptibility to mountain pine beetle attack. Western Journal of Applied Forestry. 11(2): 50-53. doi:10.1093/wjaf/11.2.50

Ansley, J.-A.S.; Battles, J.J. 1998. Forest composition, structure, and change in an old-growth mixed conifer forest in the northern Sierra Nevada. Journal of the Torrey Botanical Society. 125(4): 297-308. http://www.jstor.org/stable/2997243

Antos, J.A.; Habeck, J.R. 1981. Successional development in *Abies grandis* (Dougl.) Forbes forests in the Swan Valley, western Montana. Northwest Science. 55(1): 26-39.

Aplet, G.H. 1994. Beyond even- vs. uneven-aged management: toward a cohort-based silviculture. Journal of Sustainable Forestry. 2(3-4): 423-433. doi:10.1300/J091v02n03_12

Aplet, G.H.; Keeton, W.S. 1999. Application of historical range of variability concepts to biodiversity conservation. In: Baydack, R.K.; Campa, H., III; Haufler, J.B., eds. Practical approaches to the conservation of biological diversity. Washington, DC: Island Press: 71-86. isbn:1-55963-544-4

Aplet, G.H.; Wilmer, B. 2010. The potential for restoring fire-adapted ecosystems: exploring opportunities to expand the use of wildfire as a natural change agent. Fire Management Today. 70(1): 35-39. http://www.fs.fed.us/fire/fmt/fmt_pdfs/FMT70-1.pdf

Aplet, G.H.; Laven, R.D.; Fiedler, P.L. 1992. The relevance of conservation biology to natural resource management. Conservation Biology. 6(2): 298-300. www.jstor.org/stable/2386253

Archie, K.M.; Dilling, L.; Milford, J.B.; Pampel, F.C. 2012. Climate change and western public lands: A survey of U.S. federal land managers on the status of adaptation efforts. Ecology and Society. 17(4): art20 (24 p). doi:10.5751/ES-05187-170420

Arkle, R.S.; Pilliod, D.S. 2010. Prescribed fires as ecological surrogates for wildfires: a stream and riparian perspective. Forest Ecology and Management. 259(5): 893-903. doi:10.1016/j.foreco.2009.11.029

Arkley, R.J. 1981. Soil moisture use by mixed conifer forest in a summer-dry climate. Soil Science Society of America Journal. 45(2): 423-427. doi:10.2136/sssaj1981.03615995004500020037x

Armleder, H. 1999. Clumpy spacing: juvenile spacing Douglas-fir into clumps to imitate natural stand structure. Exten. Note 32. Victoria, BC: British Columbia Ministry of Forests. 6 p. https://www.for.gov.bc.ca/hfd/pubs/docs/en/en32.pdf

Armstrong, G.W. 1999. A stochastic characterisation of the natural disturbance regime of the boreal mixedwood forest with implications for sustainable forest management. Canadian Journal of Forest Research. 29(4): 424-433. doi:10.1139/x99-010

Armstrong, G.W.; Adamowicz, W.L.; Beck, J.A.; Cumming, S.G.; Schmiegelow, F.K.A. 2003. Coarse filter ecosystem management in a nonequilibrating forest. Forest Science. 49(2): 209-223. doi:10.1093/forestscience/49.2.209

Arno, S.F. 1980. Forest fire history in the northern Rockies. Journal of Forestry. 78(8): 460-465. doi:10.1093/jof/78.8.460

Arno, S.F. 1985. Ecological effects and management implications of Indian fires. In: Lotan, J.E.; Kilgore, B.M.; Fischer, W.C.; Mutch, R.W., tech. coords. Proceedings–symposium and workshop on wilderness fire. Gen. Tech. Rep. INT-182. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 81-86. https://archive.org/download/CAT31118982/CAT31118982.pdf

Arno, S.F.; Davis, D.H. 1980. Fire history of western redcedar/hemlock forests in northern Idaho. In: Stokes, M.A.; Dieterich, J.H., tech. coords. Proceedings of the fire history workshop. Gen. Tech. Rep. RM-81. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station: 21-26. https://www.fs.usda.gov/treesearch/pubs/41408

Arno, S.F.; Fischer, W.C. 1995. *Larix occidentalis*–fire ecology and fire management. In: Schmidt, W.C.; McDonald, K.J., comps. Ecology and management of Larix forests: a look ahead; proceedings of an international symposium. Gen. Tech. Rep. GTR-INT-319. Ogden, UT: USDA Forest Service, Intermountain Research Station: 130-135. https://www.fs.usda.gov/treesearch/pubs/41408

Arno, S.F.; Reinhardt, E.D.; Scott, J.H. 1993. Forest structure and landscape patterns in the subalpine lodgepole pine type: a procedure for quantifying past and present conditions. Gen. Tech. Rep. INT-294. Ogden, UT: USDA Forest Service, Intermountain Research Station. 17 p.

Arno, S.F.; Parsons, D.J.; Keane, R.E. 2000. Mixed-severity fire regimes in the northern

Rocky Mountains: consequences of fire exclusion and options for the future. In: Cole, D.N.; McCool, S.F.; Borrie, W.T.; O'Loughlin, J., eds. Wilderness science in a time of change conference-volume 5: wilderness ecosystems, threats, and management. Proceedings RMRS-P-15-VOL-5. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station: 225-232. http://www.treesearch.fs.fed.us/pubs/21867

Arnold, J.S.; Koro-Ljungberg, M.; Bartels, W.-L. 2012. Power and conflict in adaptive management: Analyzing the discourse of riparian management on public lands. Ecology and Society. 17(1): art19 (12 p). doi:10.5751/ES-04636-170119

Arsenault, A. 2003. A note on the ecology and management of old-growth forests in the Montane Cordillera. Forestry Chronicle. 79(3): 441-454. doi:10.5558/tfc79441-3

Arthaud, G.J.; Rose, D.W. 1996. A methodology for estimating production possibility frontiers for wildlife habitat and timber value at the landscape level. Canadian Journal of Forest Research. 26(12): 2191-2200. doi:10.1139/x26-248

Attiwill, P.M. 1994. The disturbance of forest ecosystems: the ecological basis for conservative management. Forest Ecology and Management. 63(2-3): 247-300. doi:10.1016/0378-1127(94)90114-7

Aubin, I.; Messier, C.; Bouchard, A. 2008. Can plantations develop understory biological and physical attributes of naturally regenerated forests? Biological Conservation. 141(10): 2461-2476. doi:10.1016/j.biocon.2008.07.007

Auclair, A.N.D.; Bedford, J.A. 1994. Conceptual origins of catastrophic forest mortality in the western United States. Journal of Sustainable Forestry. 2(3-4): 249-265. doi:10.1300/J091v02n03_02

Audley, J.P.; Fettig, C.J.; Steven Munson, A.; Runyon, J.B.; Mortenson, L.A.; Steed, B.E.; Gibson, K.E.; Jørgensen, C.L.; McKelvey, S.R.; McMillin, J.D.; Negrón, J.F. 2020. Impacts of mountain pine beetle outbreaks on lodgepole pine forests in the Intermountain West, U.S., 2004–2019. Forest Ecology and Management. 475: 118403 (13 p). doi:10.1016/j.foreco.2020.118403

Aussenac, G.; Granier, A. 1988. Effects of thinning on water stress and growth in Douglas-fir. Canadian Journal of Forest Research. 18(1): 100-105. doi:10.1139/x88-015

Axelson, J.N.; Alfaro, R.I.; Hawkes, B.C. 2009. Influence of fire and mountain pine beetle on the dynamics of lodgepole pine stands in British Columbia, Canada. Forest Ecology and Management. 257(9): 1874-1882. doi:10.1016/j.foreco.2009.01.047

Ayres, M.P.; Lombardero, M.J. 2000. Assessing the consequences of global change for forest disturbance from herbivores and pathogens. Science of the Total Environment. 262(3): 263-286. doi:10.1016/S0048-9697(00)00528-3

Babst, F.; Bouriaud, O.; Poulter, B.; Trouet, V.; Girardin, M.P.; Frank, D.C. 2019. Twentieth century redistribution in climatic drivers of global tree growth. Science Advances. 5(1): eaat4313 (10 p). doi:10.1126/sciadv.aat4313

Bainbridge, E.L.; L Strong, W. 2005. *Pinus contorta* understory vegetation dynamics following clearcutting in west-central Alberta, Canada. Forest Ecology and Management. 213(1-3): 133-150. doi:10.1016/j.foreco.2005.03.014

Baird, M.; Zabowski, D.; Everett, R. 1999. Wildfire effects on carbon and nitrogen in inland coniferous forests. Plant and Soil. 209(2): 233-243. doi:10.1023/A:1004602408717

Baker, F.S. 1929. Effect of excessively high temperatures on coniferous reproduction. Journal of Forestry. 27(8): 949-975. doi:10.1093/jof/27.8.949

Baker, W.L. 2002. Indians and fire in the Rocky Mountains: the wilderness hypothesis renewed. In: Vale, T.R., ed. Fire, native peoples, and the natural landscape. Washington, DC: Island Press: 41-76. isbn:1-55963-889-3

Baker, W.L. 2014. Historical forest structure and fire in Sierran mixed-conifer forests reconstructed from General Land Office survey data. Ecosphere. 5(7): art79 (70 p). doi:10.1890/ES14-00046.1

Baker, M.D.; Lacki, M.J. 1997. Short-term changes in bird communities in response to silvicultural prescriptions. Forest Ecology and Management. 96(1-2): 27-36. doi:10.1016/S0378-1127(97)00052-2

Baker, F.A.; Daniels, S.E.; Parks, C.A. 1996. Inoculating trees with wood decay fungi with rifle and shotgun. Western Journal of Applied Forestry. 11(1): 13-15. doi:10.1093/wjaf/11.1.13

Baker, D.L.; Andelt, W.F.; Burnham, K.P.; Shepperd, W.D. 1999. Effectiveness of Hot Sauce® and Deer Away® repellents for deterring elk browsing of aspen sprouts. Journal of Wildlife Management. 63(4): 1327-1336. doi:10.2307/3802851

Baker, S.C.; Halpern, C.B.; Wardlaw, T.J.; Crawford, R.L.; Bigley, R.E.; Edgar, G.J.; Evans, S.A.; Franklin, J.F.; Jordan, G.J.; Karpievitch, Y.; Spies, T.A.; Thomson, R.J. 2015. Short- and long-term benefits for forest biodiversity of retaining unlogged patches in harvested areas. Forest Ecology and Management. 353: 187-195. doi:10.1016/j.foreco.2015.05.021

Baker, S.C.; Halpern, C.B.; Wardlaw, T.J.; Kern, C.; Edgar, G.J.; Thomson, R.J.; Bigley, R.E.; Franklin, J.F.; Gandhi, K.J.K.; Gustafsson, L.; Johnson, S.; Palik, B.J.; Spies, T.A.; Steel, E.A.; Weslien, J.; Strengbom, J. 2016. A cross-continental comparison of plant and beetle responses to retention of forest patches during timber harvest. Ecological Applications. 26(8): 2495-2506. doi:10.1002/eap.1406

Balch, J.K.; Bradley, B.A.; Abatzoglou, J.T.; Nagy, R.C.; Fusco, E.J.; Mahood, A.L. 2017. Human-started wildfires expand the fire niche across the United States. Proceedings of the National Academy of Sciences of the United States of America. 114(11): 2946-2951. doi:10.1073/pnas.1617394114

Barbour, M.; Kelley, E.; Maloney, P.; Rizzo, D.; Royce, E.; Fites-Kaufmann, J. 2002. Present and past old-growth forests of the Lake Tahoe Basin, Sierra Nevada, US. Journal of Vegetation Science. 13(4): 461-472. doi:10.1111/j.1654-1103.2002.tb02073.x

Barbour, R.J.; Hemstrom, M.; Ager, A.; Hayes, J.L. 2005. Effects of spatial scale on the perception and assessment of risk of natural disturbance in forested ecosystems: examples from northeastern Oregon. Forest Ecology and Management. 211(1-2): 210-225. doi:10.1016/j.foreco.2005.02.034

Barbour, R.J.; Singleton, R.; Maguire, D.A. 2007. Evaluating forest product potential as part of planning ecological restoration treatments on forested landscapes. Landscape and Urban Planning. 80(3): 237-248. doi:10.1016/j.landurbplan.2006.12.003

Baret, M.; Pepin, S.; Pothier, D. 2018. Hydraulic limitations in dominant trees as a contributing mechanism to the age-related growth decline of boreal forest stands. Forest Ecology and Management. 427: 135-142. doi:10.1016/j.foreco.2018.05.043

Baron, J.S.; Julius, S.H.; West, J.M.; Joyce, L.A.; Blate, G.; Peterson, C.H.; Palmer, M.; Keller, B.D.; Kareiva, P.; Scott, J.M.; Griffith, B. 2008. Some guidelines for helping natural resources adapt to climate change. IHDP Update. 2: 46-52. http://www.fs.fed.us/rm/pubs_other/rmrs_2008_baron_j001.pdf

Barrett, S.W. 1980. Indians and fire. Western Wildlands. 6(3): 17-21.

Barrett, S.W. 2004a. Altered fire intervals and fire cycles in the northern Rockies. Fire Management Today. 64(3): 25-29. http://www.fs.fed.us/fire/fmt/fmt_pdfs/FMT64-3.pdf

Barrett, S.W. 2004b. Fire regimes in the northern Rockies. Fire Management Today. 64(2): 32-38. http://www.fs.fed.us/fire/fmt/fmt_pdfs/fmt64-2.pdf

Barrett, S.W.; Arno, S.F. 1982. Indian fires as an ecological influence in the northern Rockies. Journal of Forestry. 80(10): 647-651. doi:10.1093/jof/80.10.647

Barrett, S.W.; Arno, S.F. 1999. Indian fires in the northern Rockies: ethnohistory and ecology. In: Boyd, R., ed. Indians, fire, and the land in the Pacific Northwest. Corvallis, OR: Oregon State University Press: 50-64. isbn:0-87071-459-7

Barrett, J.W.; Roth, L.F. 1985. Response of dwarf mistletoe-infested ponderosa pine to thinning: 1. sapling growth. Res. Pap. PNW-330. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 15 p. http://www.fs.fed.us/pnw/pubs/pnw_rp330.pdf

Barrett, S.; Havlina, D.; Jones, J.; Hann, W.; Frame, C.; Hamilton, D.; Schon, K.; Demeo, T.; Hutter, L.; and Menakis, J. 2010. Interagency Fire Regime Condition Class guidebook. Version 3.0 [Homepage of the Interagency Fire Regime Condition Class website, USDA Forest Service, US Department of the Interior, and The Nature Conservancy]. [Online], Available: www.frcc.gov

Barrett, S.W.; Arno, S.F.; Key, C.H. 1991. Fire regimes of western larch–lodgepole pine forests in Glacier National Park, Montana. Canadian Journal of Forest Research. 21(12): 1711-1720. doi:10.1139/x91-237

Barrett, S.W.; Arno, S.F.; Menakis, J.P. 1997. Fire episodes in the Inland Northwest (1540-1940) based on fire history data. Gen. Tech. Rep. INT-GTR-370. Ogden, UT: USDA Forest Service, Intermountain Research Station. 17 p. https://www.fs.usda.gov/treesearch/pubs/34303

Barrett, S.W.; Swetnam, T.W.; Baker, W.L. 2005. Indian fire use: deflating the legend. Fire Management Today. 65(3): 31-34. http://www.fs.fed.us/fire/fmt/fmt_pdfs/FMT65-3.pdf

Barros, A.M.G.; Ager, A.A.; Day, M.A.; Palaiologou, P. 2019. Improving long-term fuel treatment effectiveness in the National Forest System through quantitative prioritization. Forest Ecology and Management. 433: 514-527. doi:10.1016/j.foreco.2018.10.041

Barsoum, N.; Coote, L.; Eycott, A.E.; Fuller, L.; Kiewitt, A.; Davies, R.G. 2016. Diversity, functional structure and functional redundancy of woodland plant communities: How do mixed tree species plantations compare with monocultures? Forest Ecology and Management. 382: 244-256. doi:10.1016/j.foreco.2016.10.005

Barth, M.A.F.; Larson, A.J.; Lutz, J.A. 2015. A forest reconstruction model to assess changes to Sierra Nevada mixed-conifer forest during the fire suppression era. Forest Ecology and Management. 354: 104-118. doi:10.1016/j.foreco.2015.06.030

Bartlett, A.A. 1994. Reflections on sustainability, population growth, and the environment. Population and Environment. 16(1): 5-35. www.jstor.org/stable/27503374

Bartos, D.L.; Booth, G.D. 1994. Effects of thinning on temperature dynamics and mountain pine beetle activity in a lodgepole pine stand. Res. Pap. INT-RP-479. Ogden, UT: USDA Forest Service, Intermountain Research Station. 9 p. https://www.fs.usda.gov/treesearch/pubs/43373

Bartos, D.L.; Campbell, R.B., Jr. 1998. Water depletion and other ecosystem values forfeited when conifer forests displace aspen communities. In: Potts, D.F., ed. Proceedings of the AWRA specialty conference, rangeland management and water resources. TPS-98-1. Herndon, VA: American Water Resources Association: 427-434.

Baskent, E.Z. 1999. Controlling spatial structure of forested landscapes: a case study towards landscape management. Landscape Ecology. 14(1): 83-97. doi:10.1023/A:1008071307848

Baskent, E.Z.; Jordan, G.A. 1996. Designing forest management to control spatial structure of landscapes. Landscape and Urban Planning. 34(1): 55-74. doi:10.1016/0169-2046(95)00200-6

Baskerville, G.L. 1975. Spruce budworm: super silviculturist. Forestry Chronicle. 51: 138-140. doi:10.5558/tfc51138-4

Bate, L.J.; Wisdom, M.J.; Wales, B.C. 2007. Snag densities in relation to human access and associated management factors in forests of northeastern Oregon, USA. Landscape and Urban Planning. 80(3): 278-291. doi:10.1016/j.landurbplan.2006.10.008

Bates, C.G. 1926. Some relations of plant ecology to silvicultural practice. Ecology. 7(4): 469-480. doi:10.2307/1931172

Bates, C.G.; Hilton, H.C.; Krueger, T. 1929. Experiments in the silvicultural control of natural reproduction of lodgepole pine in the central Rocky Mountains. Journal of Agricultural Research. 38(4): 229-243. https://naldc-legacy.nal.usda.gov/naldc/download.xhtml?id=IND43967667&content=PDF

Battles, J.J.; Shlisky, A.J.; Barrett, R.H.; Heald, R.C.; Allen-Diaz, B.H. 2001. The effects of forest management on plant species diversity in a Sierran conifer forest. Forest Ecology and Management. 146(1-3): 211-222. doi:10.1016/S0378-1127(00)00463-1

Battles, J.J.; Robards, T.; Das, A.; Waring, K.; Gilless, J.K.; Biging, G.; Schurr, F. 2008. Climate change impacts on forest growth and tree mortality: a data-driven modeling study in the mixed-conifer forest of the Sierra Nevada, California. Climatic Change. 87(1): 193-213. doi:10.1007/s10584-007-9358-9

Bauhus, J.; Puettmann, K.; Messier, C. 2009. Silviculture for old-growth attributes. Forest Ecology and Management. 258(4): 525-537. doi:10.1016/j.foreco.2009.01.053

Baumgartner, D.M.; Lotan, J.E., comps. 1991. Interior Douglas-fir: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Department of Natural Resource Sciences, Cooperative Extension. 301 p.

Baumgartner, D.M.; Mitchell, R., comps. 1984. Silvicultural management strategies for pests of the interior Douglas-fir and grand fir forest types. Pullman, WA: Washington State University, Cooperative Extension. 199 p.

Baumgartner, D.M.; Krebill, R.G.; Arnott, J.T.; Weetman, G.F., comps. 1985. Lodgepole pine: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Cooperative Extension, Office of Conferences and Institutes. 381 p.

Baumgartner, D.M.; Lotan, J.E.; Tonn, J.R., comps. 1994. Interior cedar-hemlock-white pine forests: ecology and management. Pullman, WA: Washington State University, Department of Natural Resource Sciences. 365 p.

Baydack, R.K.; Campa, H.I.; Haufler, J.B., eds. 1999. Practical approaches to the conservation of biological diversity. Washington, DC: Island Press. 313 p. isbn:1-55963-544-4

Bayne, E.; Nielsen, B. 2011. Temporal trends in bird abundance in response to thinning of lodgepole pine (*Pinus contorta*). Canadian Journal of Forest Research. 41(10): 1917-1927. doi:10.1139/x11-113

Bazzaz, F.A. 1979. The physiological ecology of plant succession. Annual Review of Ecology and Systematics. 10: 351-371. doi:10.1146/annurev.es.10.110179.002031

Bazzaz, F.A.; Chiariello, N.R.; Coley, P.D.; Pitelka, L.F. 1987. Allocating resources to reproduction and defense. BioScience. 37(1): 58-67. doi:10.2307/1310178

Beatty, J.S.; Geils, B.W.; Lundquist, J.E. 1995. Disturbance and canopy gaps as indicators of forest health in the Blue Mountains of Oregon. In: Eskew, L.G., comp. Forest health through silviculture: proceedings of the 1995 National Silviculture Workshop. Gen. Tech. Rep. RM-GTR-267. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station: 74-78. https://www.fs.usda.gov/treesearch/pubs/23487

Beatty, J.S.; Filip, G.M.; Mathiasen, R.L. [1997]. Larch dwarf mistletoe. Forest Insect & Disease Leaflet 169; R6-FID-PR-009-97. [Place of publication unknown]: USDA Forest Service. 7 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043215.pdf

Beechie, T.J.; Sear, D.A.; Olden, J.D.; Pess, G.R.; Buffington, J.M.; Moir, H.; Roni, P.; Pollock, M.M. 2010. Process-based principles for restoring river ecosystems. BioScience. 60(3): 209-222. doi:10.1525/bio.2010.60.3.7

Beechie, T.; Imaki, H.; Greene, J.; Wade, A.; Wu, H.; Pess, G.; Roni, P.; Kimball, J.; Stanford, J.; Kiffney, P.; Mantua, N. 2013. Restoring salmon habitat for a changing climate. River Research and Applications. 29(8): 939-960. doi:10.1002/rra.2590

Behan, R.W. 1997. Scarcity, simplicity, separatism, science–and systems. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press: 411-417. isbn:1-55963-399-9

Beier, P.; Brost, B. 2010. Use of land facets to plan for climate change: Conserving the arenas, not the actors. Conservation Biology. 24(3): 701-710. doi:10.1111/j.1523-1739.2009.01422.x

Bell, T.L.; Adams, M.A. 2011. Attack on all fronts: functional relationships between aerial and root parasitic plants and their woody hosts and consequences for ecosystems. Tree Physiology. 31: 3-15. doi:10.1093/treephys/tpq108

Bellard, C.; Bertelsmeier, C.; Leadley, P.; Thuiller, W.; Courchamp, F. 2012. Impacts of climate change on the future of biodiversity. Ecology Letters. 15(4): 365-377. doi:10.1111/j.1461-0248.2011.01736.x

Belote, R.T.; Dietz, M.S.; Aplet, G.H. 2015. Allocating untreated “controls” in the National Wilderness Preservation System as a climate adaptation strategy: a case study from the Flathead National Forest, Montana. Northwest Science. 89(3): 239-254. doi:10.3955/046.089.0311

Belote, R.T.; Larson, A.J.; Dietz, M.S. 2015. Tree survival scales to community-level effects following mixed-severity fire in a mixed-conifer forest. Forest Ecology and Management. 353: 221-231. doi:10.1016/j.foreco.2015.05.033

Benda, L.E.; Poff, N.L.; Tague, C.; Palmer, M.A.; Pizzuto, J.; Cooper, S.; Stanley, E.; Moglen, G. 2002. How to avoid train wrecks when using science in environmental problem solving. BioScience. 52(12): 1127-1136. doi:10.1641/0006-3568(2002)052[1127:HTATWW]2.0.CO;2

Benecke, U. 1996. Ecological silviculture: the application of age-old methods. New Zealand Forestry. 41(2): 27-33. Ecological Silviculture

Bengston, D.N. 1994. Changing forest values and ecosystem management. Society and Natural Resources. 7(6): 515-533. doi:10.1080/08941929409380885

Bengtsson, J.; Nilsson, S.G.; Franc, A.; Menozzi, P. 2000. Biodiversity, disturbances, ecosystem function and management of European forests. Forest Ecology and Management. 132(1): 39-50. doi:10.1016/S0378-1127(00)00378-9

Bentz, B. 2009. Bark beetle outbreaks in western North America: causes and consequences. Salt Lake City, UT: University of Utah Press. 42 p. isbn:978-0-87480-965-7

Bentz, B.J.; Régnière, J.; Fettig, C.J.; Hansen, E.M.; Hayes, J.L.; Hicke, J.A.; Kelsey, R.G.; Negrón, J.F.; Seybold, S.J. 2010. Climate change and bark beetles of the western United States and Canada: direct and indirect effects. BioScience. 60(8): 602-613. doi:10.1525/bio.2010.60.8.6

Bergeron, Y.; Harvey, B. 1997. Basing silviculture on natural ecosystem dynamics: an approach applied to the southern boreal mixedwood forest of Quebec. Forest Ecology and Management. 92(1-3): 235-242. doi:10.1016/S0378-1127(96)03924-2

Bergeron, Y.; Leduc, A. 1998. Relationships between change in fire frequency and mortality due to spruce budworm outbreak in the southeastern Canadian boreal forest. Journal of Vegetation Science. 9(4): 492-500. doi:10.2307/3237264

Bergeron, Y.; Richard, P.J.H.; Carcaillet, C.; Gauthier, S.; Flannigan, M.; Prairie, Y.T. 1998. Variability in fire frequency and forest composition in Canada's southern boreal forest: a challenge for sustainable forest management. Conservation Ecology. 2(2): 1-10. http://www.consecol.org/vol2/iss2/art6

Bergeron, Y.; Harvey, B.; Leduc, A.; Gauthier, S. 1999. Forest management guidelines based on natural disturbance dynamics: stand- and forest-level considerations. Forestry Chronicle. 75(1): 49-54. doi:10.5558/tfc75049-1

Bergeron, Y.; Leduc, A.; Harvey, B.D.; Gauthier, S. 2002. Natural fire regime: a guide for sustainable management of the Canadian boreal forest. Silva Fennica. 36(1): 81-95. doi:10.14214/sf.553

Bergeron, Y.; Cyr, D.; Drever, C.R.; Flannigan, M.; Gauthier, S.; Kneeshaw, D.; Lauzon, E.; Leduc, A.; Le Goff, H.; Lesieur, D.; Logan, K. 2006. Past, current, and future fire frequencies in Quebec's commercial forests: implications for the cumulative effects of harvesting and fire on age-class structure and natural disturbance-based management. Canadian Journal of Forest Research. 36(11): 2737-2744. doi:10.1139/X06-177

Bergeron, Y.; Cyr, D.; Girardin, M.P.; Carcaillet, C. 2010. Will climate change drive 21st century burn rates in Canadian boreal forest outside of its natural variability: collating global climate model experiments with sedimentary charcoal data. International Journal of Wildland Fire. 19(8): 1127-1139. doi:10.1071/WF09092

Berkes, F. 2008. Sacred ecology. New York, NY: Routledge. 313 p. isbn:0-415-95829-6

Berkes, F.; Colding, J.; Folke, C. 2000. Rediscovery of traditional ecological knowledge as adaptive management. Ecological Applications. 10(5): 1251-1262. doi:10.1890/1051-0761(2000)010[1251:ROTEKA]2.0.CO;2

Berlik, M.M.; Kittredge, D.B.; Foster, D.R. 2002. The illusion of preservation: a global environmental argument for the local production of natural resources. Journal of Biogeography. 29(10-11): 1557-1568. doi:10.1046/j.1365-2699.2002.00768.x

Berryman, A.A. 1969. Responses of *Abies grandis* to attack by *Scolytus ventralis* (Coleoptera: Scolytidae). Canadian Entomologist. 101(10): 1033-1041. doi:10.4039/Ent1011033-10

Berryman, A.A. 1972. Resistance of conifers to invasion by bark beetle-fungus associations. BioScience. 22(10): 598-602. doi:10.2307/1296206

Berryman, A.A. 1973. Population dynamics of the fir engraver, *Scolytus ventralis* (Coleoptera: Scolytidae): I, analysis of population behavior and survival from 1964-1971. Canadian Entomologist. 105(11): 1465-1488. doi:10.4039/Ent1051465-11

Berryman, A.A. 1976. Theoretical explanation of mountain pine beetle dynamics in lodgepole pine forests. Environmental Entomology. 5(6): 1225-1233. doi:10.1093/ee/5.6.1225

Berryman, A.A. 1982. Biological control, thresholds, and pest outbreaks. Environmental Entomology. 11(3): 544-549. doi:10.1093/ee/11.3.544

Berryman, A.A. 1991. Population theory: an essential ingredient in pest prediction, management and policy-making. American Entomologist. 37(3): 138-142. doi:10.1093/ae/37.3.138

Berryman, A.A. 1996. What causes population cycles of forest Lepidoptera? Trends in Ecology and Evolution. 11(1): 28-32. doi:10.1016/0169-5347(96)81066-4

Berryman, A.A.; Ashraf, M. 1970. Effects of *Abies grandis* resin on the attack behavior and brood survival of *Scolytus ventralis* (Coleoptera: Scolytidae). Canadian Entomologist. 102(10): 1229-1236. doi:10.4039/Ent1021229-10

Berryman, A.A.; Stark, R.W. 1985. Assessing the risk of lodgepole pine stand destruction by pests. In: Baumgartner, D.M.; Krebill, R.G.; Arnott, J.T.; Weetman, G.F., comps. Lodgepole pine: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Cooperative Extension, Office of Conferences and Institutes: 163-169.

Berryman, A.A.; Stenseth, N.C.; Wollkind, D.J. 1984. Metastability of forest ecosystems infested by bark beetles. Researches on Population Ecology. 26(1): 13-29. doi:10.1007/BF02515505

Berryman, A.A.; Stenseth, N.C.; Isaev, A.S. 1987. Natural regulation of herbivorous forest insect populations. Oecologia. 71(2): 174-184. doi:10.1007/BF00377282

Beschta, R.L.; Ripple, W.J. 2012. Berry-producing shrub characteristics following wolf reintroduction in Yellowstone National Park. Forest Ecology and Management. 276: 132-138. doi:10.1016/j.foreco.2012.03.035

Beschta, R.L.; Ripple, W.J. 2016. Riparian vegetation recovery in Yellowstone: The first two decades after wolf reintroduction. Biological Conservation. 198: 93-103. doi:10.1016/j.biocon.2016.03.031

Beschta, R.L.; Ripple, W.J. 2020. Large carnivore extirpation linked to loss of overstory aspen in Yellowstone. Food Webs. 22: e00140 (7 p). doi:10.1016/j.fooweb.2019.e00140

Beschta, R.L.; Donahue, D.L.; DellaSala, D.A.; Rhodes, J.J.; Karr, J.R.; O'Brien, M.H.; Fleischner, T.L.; Deacon Williams, C. 2013. Adapting to climate change on western public lands: Addressing the ecological effects of domestic, wild, and feral ungulates. Environmental Management. 51(2): 474-491. doi:10.1007/s00267-012-9964-9

Beschta, R.L.; Painter, L.E.; Levi, T.; Ripple, W.J. 2016. Long-term aspen dynamics, trophic cascades, and climate in northern Yellowstone National Park. Canadian Journal of Forest Research. 46(4): 548-556. doi:10.1139/cjfr-2015-0301

Beschta, R.L.; Painter, L.E.; Ripple, W.J. 2018. Trophic cascades at multiple spatial scales shape recovery of young aspen in Yellowstone. Forest Ecology and Management. 413: 62-69. doi:10.1016/j.foreco.2018.01.055

Bessie, W.C.; Johnson, E.A. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. Ecology. 76(3): 747-762. doi:10.2307/1939341

Beyer, H.L.; Merrill, E.H.; Varley, N.; Boyce, M.S. 2007. Willow on Yellowstone's northern range: Evidence for a trophic cascade? Ecological Applications. 17(6): 1563-1571. doi:10.1890/06-1254.1

Bigelow, S.W.; North, M.P.; Salk, C.F. 2011. Using light to predict fuels-reduction and group-selection effects on succession in Sierran mixed-conifer forest. Canadian Journal of Forest Research. 41(10): 2051-2063. doi:10.1139/x11-120

Bigelow, S.W.; North, M.P. 2012. Microclimate effects of fuels-reduction and group-selection silviculture: implications for fire behavior in Sierran mixed-conifer forests. Forest Ecology and Management. 264: 51-59. doi:10.1016/j.foreco.2011.09.031

Bigelow, S.W.; Papaik, M.J.; Caum, C.; North, M.P. 2014. Faster growth in warmer winters for large trees in a Mediterranean-climate ecosystem. Climatic Change. 123(2): 215-224. doi:10.1007/s10584-014-1060-0

Biggs, R.; Carpenter, S.R.; Brock, W.A. 2009. Turning back from the brink: Detecting an impending regime shift in time to avert it. Proceedings of the National Academy of Sciences. 106(3): 826-831. doi:10.1073/pnas.0811729106

Bigio, E.R.; Swetnam, T.W.; Baisan, C.H. 2016. Local-scale and regional climate controls on historical fire regimes in the San Juan Mountains, Colorado. Forest Ecology and Management. 360: 311-322. doi:10.1016/j.foreco.2015.10.041

Binkley, D.; Stape, J.L.; Ryan, M.G. 2004. Thinking about efficiency of resource use in forests. Forest Ecology and Management. 193(1-2): 5-16. doi:10.1016/j.foreco.2004.01.019

Birdsey, R.; Pan, Y. 2011. Drought and dead trees. Nature Climate Change. 1(9): 444-445. doi:10.1038/nclimate1298

Birdsey, R.A.; Dugan, A.J.; Healey, S.P.; Dante-Wood, K.; Zhang, F.; Mo, G.; Chen, J.M.; Hernandez, A.J.; Raymond, C.L.; McCarter, J. 2019. Assessment of the influence of disturbance, management activities, and environmental factors on carbon stocks of U.S. national forests. Gen. Tech. Rep. RMRS-GTR-402. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 116 p. https://www.fs.usda.gov/treesearch/pubs/59157

Bisson, P.A.; Rieman, B.E.; Luce, C.; Hessburg, P.F.; Lee, D.C.; Kershner, J.L.; Reeves, G.H.;

Gresswell, R.E. 2003. Fire and aquatic ecosystems of the western USA: current knowledge and key questions. Forest Ecology and Management. 178(1-2): 213-229. doi:10.1016/S0378-1127(03)00063-X

Bisson, P.A.; Dunham, J.B.; Reeves, G.H. 2009. Freshwater ecosystems and resilience of Pacific salmon: Habitat management based on natural variability. Ecology and Society. 14(1): 45 (18 p). doi:10.5751/ES-02784-140145

Bissonette, J.A.; Storch, I., eds. 2003. Landscape ecology and resource management: linking theory with practice. Washington, DC: Island Press. 463 p. isbn:1-55963-973-3

Biswas, S.R.; Mallik, A.U. 2010. Disturbance effects on species diversity and functional diversity in riparian and upland plant communities. Ecology. 91(1): 28-35. doi:10.1890/08-0887.1

Black, H.C. 1992. Silvicultural approaches to animal damage management in Pacific Northwest forests. Gen. Tech. Rep. PNW-GTR-287. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 422 p. https://www.fs.usda.gov/treesearch/pubs/25665

Black, H.C. 1994. Animal damage management handbook. Gen. Tech. Rep. PNW-GTR-332. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 236 p. https://www.fs.usda.gov/treesearch/pubs/3067

Black, T.A.; Nnyamah, U.; Tan, C.S. 1980. Transpiration rate of Douglas fir trees in thinned and unthinned stands. Canadian Journal of Soil Science. 60(4): 625-631. doi:10.4141/cjss80-071

Black, A.E.; Morgan, P.; Hessburg, P.F. 2003. Social and biophysical correlates of change in forest landscapes of the interior Columbia basin, USA. Ecological Applications. 13(1): 51-67. doi:10.1890/1051-0761(2003)013[0051:SABCOC]2.0.CO;2

Blanchette, R.A.; Shaw, C.G. 1978. Management of forest residues for rapid decay. Canadian Journal of Botany. 56(22): 2904-2909. doi:10.1139/b78-348

Blate, G.M.; Joyce, L.A.; Littell, J.S.; McNulty, S.G.; Millar, C.I.; Moser, S.C.; Neilson, R.P.; O'Halloran, K.; Peterson, D.L. 2009. Adapting to climate change in United States national forests. Unasylva 231/232. 60(1-2): 57-62. ftp://ftp.fao.org/docrep/fao/011/i0670e/i0670e00.pdf

Blatner, K.A.; Keegan, C.E.; O'Laughlin, J.; Adams, D.L. 1994. Forest health management policy: A case study in southwestern Idaho. Journal of Sustainable Forestry. 2(3-4): 317-337. doi:10.1300/J091v02n03_07

Bleiker, K.P.; Uzunovic, A. 2004. Fast- and slow-growing subalpine fir produce lesions of different sizes in response to inoculation with a blue-stain fungus associated with *Dryocoetes confusus* (Coleoptera: Scolytidae). Canadian Journal of Botany. 82(6): 735-741. doi:10.1139/b04-053

Blicharska, M.; Mikusinski, G. 2014. Incorporating social and cultural significance of large old trees in conservation policy. Conservation Biology. 28(6): 1558-1567. doi:10.1111/cobi.12341

Bliss, J.C. 2000. Public perceptions of clearcutting. Journal of Forestry. 98(12): 4-9. doi:10.1093/jof/98.12.4

Bliss, J.; Aplet, G.; Hartzell, C.; Harwood, P.; Jahnige, P.; Kittredge, D.; Lewandowski, S.; Soscia, M.L. 2001. Community-based ecosystem monitoring. Journal of Sustainable Forestry. 12(3-4): 143-167. doi:10.1300/J091v12n03_07

Blomdahl, E.M.; Kolden, C.A.; Meddens, A.J.H.; Lutz, J.A. 2019. The importance of small fire refugia in the central Sierra Nevada, California, USA. Forest Ecology and Management. 432: 1041-1052. doi:10.1016/j.foreco.2018.10.038

Blumer, J.C. 1910. Fire as a biological factor. Plant World. 13(2): 42-44. http://www.jstor.org/stable/43477366

Blunden, J.; Arndt, D.S., eds. 2013. State of the climate in 2012. Bulletin of American Meteorological Society. 94(8): S1-S238. doi:10.1175/2013BAMSStateoftheClimate.1

Bloomberg, W.J. 1950. Fire and spruce. Forestry Chronicle. 26(2): 157-161. doi:10.5558/tfc26157-2

Boag, A.E.; Hamilton, L.C.; Hartter, J.; Stevens, F.R.; Palace, M.W.; Ducey, M.J. 2016. Shifting environmental concern in rural eastern Oregon: the role of demographic and place-based factors. Population and Environment. 38(2): 207-216. doi:10.1007/s11111-016-0261-z

Boag, A.E.; Hartter, J.; Hamilton, L.C.; Christoffersen, N.D.; Stevens, F.R.; Palace, M.W.; Ducey, M.J. 2018. Climate change beliefs and forest management in eastern Oregon: implications for individual adaptive capacity. Ecology and Society. 23(4): 1 (21 p). doi:10.5751/ES-10355-230401

Boag, A.E.; Ducey, M.J.; Palace, M.W.; Hartter, J. 2020. Topography and fire legacies drive variable post-fire juvenile conifer regeneration in eastern Oregon, USA. Forest Ecology and Management. 474: 118312 (13 p). doi:10.1016/j.foreco.2020.118312

Bodin, P.; Wiman, B.L.B. 2007. The usefulness of stability concepts in forest management when coping with increasing climate uncertainties. Forest Ecology and Management. 242(2-3): 541-552. doi:10.1016/j.foreco.2007.01.066

Boerker, R.H. 1916. A historical study of forest ecology: its development in the fields of botany and forestry. Forestry Quarterly. 14(3): 380-432. doi:10.1093/jof/14.3.380

Bohannon, J. 2013. Who's afraid of peer review? Science. 342(6154): 60-65. doi:10.1126/science.342.6154.60

Boise Cascade Corporation. 1996. Forest ecosystem management: a graphic overview. La Grande, OR: Boise Cascade Corporation, Timber and Wood Products Division. 52 p.

Boisvenue, C.; Running, S.W. 2010. Simulations show decreasing carbon stocks and potential for carbon emissions in Rocky Mountain forests over the next century. Ecological Applications. 20(5): 1302-1319. doi:10.1890/09-0504.1

Bollenbacher, B.L.; Graham, R.T.; Reynolds, K.M. 2014. Regional forest landscape restoration priorities: Integrating historical conditions and an uncertain future in the northern Rocky Mountains. Journal of Forestry. 112(5): 474-483. doi:10.5849/jof.13-086

Bolon, N.A. 1994. Estimates of the values of elk in the Blue Mountains of Oregon and Washington: evidence from the existing literature. Gen. Tech. Rep. PNW-GTR-316. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 38 p. https://www.fs.usda.gov/treesearch/pubs/9059

Bolte, A.; Ammer, C.; Löf, M.; Madsen, P.; Nabuurs, G.-J.; Schall, P.; Spathelf, P.; Rock, J. 2009. Adaptive forest management in central Europe: climate change impacts, strategies and integrative concept. Scandinavian Journal of Forest Research. 24: 473-482. doi:10.1080/02827580903418224

Bonar, R.L.; Lougheed, H.; Andison, D.W. 2003. Natural disturbance and old-forest management in the Alberta Foothills. Forestry Chronicle. 79(3): 455-461. doi:10.5558/tfc79455-3

Bond, W.J.; Woodward, F.I.; Midgley, G.F. 2005. The global distribution of ecosystems in a world without fire. New Phytologist. 165(2): 525-538. doi:10.1111/j.1469-8137.2004.01252.x

Bonnicksen, T.M. 2000a. America's ancient forests: from the ice age to the age of discovery. New York: John Wiley & Sons. 594 p. isbn:0-471-13622-0

Bonnicksen, T.M. 2000b. Our forests need 'active management.' 21st Century. 13(2): 68-69. https://21sci-tech.com/Subscriptions/Archive/2000_F.pdf

Bonnicksen, T.M.; Anderson, M.K.; Lewis, H.T.; Kay, C.E.; Knudson, R. 1999. Native American influences on the development of forest ecosystems. In: Szaro, R.C.; Johnson, N.C.;

Sexton, W.T.; Malk, A.J., eds. Ecological stewardship: a common reference for ecosystem management. Volume II. Oxford, UK: Elsevier Science, Ltd: 439-470. isbn:0-08-043206-9

Boring, L.R.; Swank, W.T.; Waide, J.B.; Henderson, G.S. 1988. Sources, fates, and impacts of nitrogen inputs to terrestrial ecosystems: review and synthesis. Biogeochemistry. 6(2): 119-159. doi:10.1007/BF00003034

Bormann, B.T.; Gordon, J.C. 1989. Can intensively managed forest ecosystems be self-sufficient in nitrogen? Forest Ecology and Management. 29(1-2): 95-103. doi:10.1016/0378-1127(89)90058-3

Bormann, F.H.; Likens, G.E. 1979. Pattern and process in a forested ecosystem. New York: Springer-Verlag. 264 p.

Bormann, B.T.; Brookes, M.H.; Ford, E.D.; Klester, A.R.; Oliver, C.D.; Weigand, J.F. 1994a. Volume V: a framework for sustainable ecosystem management. Gen. Tech. Rep. PNW-GTR-331. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 61 p. https://www.fs.usda.gov/treesearch/pubs/7080

Bormann, B.T.; Cunningham, P.G.; Brookes, M.H.; Manning, V.W.; Collopy, M.W. 1994b. Adaptive ecosystem management in the Pacific Northwest. Gen. Tech. Rep. PNW-GTR-341. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 22 p. https://www.fs.usda.gov/treesearch/pubs/8965

Bormann, B.T.; Homann, P.S.; Darbyshire, R.L.; Morrissette, B.A. 2008. Intense forest wildfire sharply reduces mineral soil C and N: the first direct evidence. Canadian Journal of Forest Research. 38(11): 2771-2783. doi:10.1139/X08-136

Bose, A.K.; Weiskittel, A.; Kuehne, C.; Wagner, R.G.; Turnblom, E.; Burkhart, H.E. 2018. Tree-level growth and survival following commercial thinning of four major softwood species in North America. Forest Ecology and Management. 427: 355-364. doi:10.1016/j.foreco.2018.06.019

Bothwell, K.N. 2019. Practicing collaborative natural resource management with federal agencies: Keys to success across partnership structures. Journal of Forestry. 117(3): 226-233. doi:10.1093/jofore/fvz010

Botkin, D.B. 1990a. Discordant harmonies: a new ecology for the twenty-first century. New York: Oxford University Press. 241 p. isbn:0-19-507469-6

Botkin, D.B. 1990b. Fire in the forest: managing living resources. Chapter 10. In: Botkin, D.B. Discordant harmonies: a new ecology for the twenty-first century. New York: Oxford University Press: 153-167. isbn:0-19-507469-6

Botkin, D.B. 1994. Preface. In: Sampson, R.N.; Adams, D.L., eds. Assessing forest ecosystem health in the Inland West. Binghamton, NY: Food Products Press (Haworth Press): xxi-xxiv. isbn:1-56022-052-X

Botkin, D.B. 1995. Our natural history: the lessons of Lewis and Clark. New York: G.P. Putnam's Sons. 300 p. isbn:0-399-14048-4

Bottero, A.; D'Amato, A.W.; Palik, B.J.; Bradford, J.B.; Fraver, S.; Battaglia, M.A.; Asherin, L.A. 2017. Density-dependent vulnerability of forest ecosystems to drought. Journal of Applied Ecology. 54(6): 1605-1614. doi:10.1111/1365-2664.12847

Bouchard, M.; Aquilué, N.; Périé, C.; Lambert, M.-C. 2019. Tree species persistence under warming conditions: A key driver of forest response to climate change. Forest Ecology and Management. 442: 96-104. doi:10.1016/j.foreco.2019.03.040

Bourgeron, P.S.; Humphries, H.C.; Jensen, M.E. 1994. Landscape characterization: A framework for ecological assessment at regional and local scales. Journal of Sustainable Forestry. 2(3-4): 267-281. doi:10.1300/J091v02n03_03

Bowman, D.M.J.S.; Balch, J.K.; Artaxo, P.; Bond, W.J.; Carlson, J.M.; Cochrane, M.A.; D'Antonio, C.M.; DeFries, R.S.; Doyle, J.C.; Harrison, S.P.; Johnston, F.H.; Keeley, J.E.;

Krawchuk, M.A.; Kull, C.A.; Marston, J.B.; Moritz, M.A.; Prentice, I.C.; Roos, C.I.; Scott, A.C.; Swetnam, T.W.; van der Werf, G.R.; Pyne, S.J. 2009. Fire in the Earth system. Science. 324(5926): 481-484. doi:10.1126/science.1163886

Bowyer, J.L.; Shmulsky, R.; Haygreen, J.G. 2007. Forest products and wood science: an introduction. 5th edition. Oxford, UK: Blackwell Publishing. 558 p. isbn:978-0-8138-2036-1

Boyce, R.L. 2009. Invasive shrubs and forest tree regeneration. Journal of Sustainable Forestry. 28(1): 152-217. doi:10.1080/10549810802626449

Boyce, M.S.; Haney, A., eds. 1997. Ecosystem management: applications for sustainable forest and wildlife resources. New Haven, CT: Yale University Press. 361 p. isbn:0-300-06902-2

Boyd, R., ed. 1999. Indians, fire, and the land in the Pacific Northwest. Corvallis, OR: Oregon State University Press. 313 p. isbn:0-87071-459-7

Boyden, S.; Montgomery, R.; Reich, P.B.; Palik, B. 2012. Seeing the forest for the heterogeneous trees: stand-scale resource distributions emerge from tree-scale structure. Ecological Applications. 22(5): 1578-1588. doi:10.1890/11-1469.1

Boyle, J.R.; Phillips, J.J.; Ek, A.R. 1973. “Whole tree” harvesting: nutrient budget evaluation. Journal of Forestry. 71(12): 760-762. doi:10.1093/jof/71.12.760

Bradford, J.B.; Bell, D.M. 2017. A window of opportunity for climate-change adaptation: Easing tree mortality by reducing forest basal area. Frontiers in Ecology and the Environment. 15(1): 11-17. doi:10.1002/fee.1445

Bradshaw, F.J. 1992. Quantifying edge effect and patch size for multiple-use silviculture – a discussion paper. Forest Ecology and Management. 48(3-4): 249-264. doi:10.1016/0378-1127(92)90148-3

Brawn, J.D.; Robinson, S.K.; Thompson, F.R. III. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics. 32: 251-276. doi:10.1146/annurev.ecolsys.32.081501.114031

Braziunas, K.H.; Hansen, W.D.; Seidl, R.; Rammer, W.; Turner, M.G. 2018. Looking beyond the mean: Drivers of variability in postfire stand development of conifers in Greater Yellowstone. Forest Ecology and Management. 430: 460-471. doi:10.1016/j.foreco.2018.08.034

Bréda, N.; Huc, R.; Granier, A.; Dreyer, E. 2006. Temperate forest trees and stands under severe drought: a review of ecophysiological responses, adaptation processes and long-term consequences. Annals of Forest Science. 63(6): 625-644. doi:10.1051/forest:2006042

Brenner, J., ed. 1998. Fire in Florida’s ecosystems; educator’s guide, grades 4-8. Tallahassee, FL: Florida Department of Agriculture and Consumer Services, Forest Protection Bureau, Division of Forestry. 62 p.

Breshears, D.D.; Cobb, N.S.; Rich, P.M.; Price, K.P.; Allen, C.D.; Balice, R.G.; Romme, W.H.; Kastens, J.H.; Floyd, M.L.; Belnap, J.; Anderson, J.J.; Myers, O.B.; Meyer, C.W. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences of the United States of America. 102(42): 15144-15148. doi:10.1073/pnas.0505734102

Breshears, D.D.; López-Hoffman, L.; Graumlich, L.J. 2011. When ecosystem services crash: Preparing for big, fast, patchy climate change. AMBIO: A Journal of the Human Environment. 40(3): 256-263. doi:10.1007/s13280-010-0106-4

Brewer, J.S.; Bertz, C.A.; Cannon, J.B.; Chesser, J.D.; Maynard, E.E. 2012. Do natural disturbances or the forestry practices that follow them convert forests to early-successional communities? Ecological Applications. 22(2): 442-458. doi:10.1890/11-0386.1

Bright, G.A.; Powell, D.C. (ed.). 2008. An extensive reconnaissance of the Wenaha National Forest in 1913. Tech. Pub. F14-SO-06-08. Pendleton, OR: USDA Forest Service, Pacific

Northwest Region, Umatilla National Forest. 65 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015522.pdf

British Columbia Habitat Branch. 2000. Environmental risk assessment (ERA): an approach for assessing and reporting environmental conditions. Tech. Bull. 1. [Place of publication unknown]: British Columbia Ministry of Environment, Lands and Parks, Habitat Branch. 70 p. http://www.env.gov.bc.ca/wld/documents/era.pdf

British Columbia Ministry of Forests. 1995a. Bark beetle management guidebook. Victoria, BC: British Columbia Forest Service. 57 p. fpc/fpcguide/beetle/

British Columbia Ministry of Forests. 1995b. Biodiversity guidebook. Victoria, BC: British Columbia Forest Service. 99 p. https://www.for.gov.bc.ca/hfd/library/documents/bib19715.pdf

British Columbia Ministry of Forests. 1995c. Defoliator management guidebook. Victoria, BC: British Columbia Forest Service. 72 p. fpc/fpcguide/defoliat/

British Columbia Ministry of Forests. 1995d. Dwarf mistletoe management guidebook. Victoria, BC: British Columbia Forest Service. 20 p.

Brix, H. 1979. Effects of plant water stress on photosynthesis and survival of four conifers. Canadian Journal of Forest Research. 9(2): 160-165. doi:10.1139/x79-030

Brockley, R.P.; Trowbridge, R.L.; Ballard, T.M.; Macadam, A.M. 1992. Nutrient management in interior forest types. In: Chappell, H.N.; Weetman, G.F.; Miller, R.E., eds. Forest fertilization: sustaining and improving nutrition and growth of western forests. Contribution No. 73. Seattle, WA: University of Washington, College of Forest Resources: 43-64.

Brodie, L.C.; Harrington, C.A. 2020. Guide to variable-density thinning using skips and gaps. Gen. Tech. Rep. PNW-GTR-989. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 37 p. https://www.fs.usda.gov/treesearch/pubs/60596

Brodribb, T.J.; Powers, J.; Cochard, H.; Choat, B. 2020. Hanging by a thread? Forests and drought. Science. 368(6488): 261-266. doi:10.1126/science.aat7631

Brookes, M.H.; Campbell, R.W., eds. 1978. The Douglas-fir tussock moth: a synthesis. Tech. Bull. 1585. Washington, DC: USDA Forest Service, Science and Education Agency. 331 p. The_Douglas_fir_Tussock_Moth

Brookes, M.H.; Colbert, J.J.; Mitchell, R.G.; Stark, R.W. 1985. Managing trees and stands susceptible to western spruce budworm. Tech. Bull. No. 1695. Washington, DC: USDA Forest Service, Cooperative State Research Service. 111 p.

Brookes, M.H.; Campbell, R.W.; Colbert, J.J.; Mitchell, R.G.; Stark, R.W. 1987. Western spruce budworm. Tech. Bull. No. 1694. Washington, DC: USDA Forest Service, Cooperative State Research Service. 198 p. https://digitalcommons.usu.edu/cgi/viewcontent.cgi?article=1318&context=barkbeetles

Brooks, J.R.; Barnard, H.R.; Coulombe, R.; McDonnell, J.J. 2010. Ecohydrologic separation of water between trees and streams in a Mediterranean climate. Nature Geoscience. 3: 100-104. doi:10.1038/NGE0722

Brown, J.K. 1975. Fire cycles and community dynamics in lodgepole pine forests. In: Baumgartner, D.M., ed. Management of lodgepole pine ecosystems; symposium proceedings. Pullman, WA: Washington State University, Cooperative Extension Service: 429-456

Brown, V.K. 1985. Insect herbivores and plant succession. Oikos. 44(1): 17-22. doi:10.2307/3544037

Brown, H. 2005. Ecological restoration in Montana's western larch. Fire Management Today. 65(4): 28-35. http://www.fs.fed.us/sites/default/files/legacy_files/fire-management-today/65-4.pdf

Brown, S.J.M. 2012. The Soda Bear Project and the Blue Mountains Forest Partners/USDA Forest Service collaboration. Journal of Forestry. 110(8): 446-447. doi:10.5849/jof.12-078

Brown, V.K.; Gange, A.C. 1992. Secondary plant succession: How is it modified by insect herbivory? Vegetatio. 101(1): 3-13. http://www.jstor.org/stable/20046173

Brown, J.K.; Smith, J.K., eds. 2000. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42-volume 2. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station. 257 p. http://www.treesearch.fs.fed.us/pubs/4554

Brown, J.K.; Reinhardt, E.D.; Fischer, W.C. 1991. Predicting duff and woody fuel consumption in northern Idaho prescribed fires. Forest Science. 37(6): 1550-1566. doi:10.1093/forestscience/37.6.1550

Brown, J.K.; Reinhardt, E.D.; Kramer, K.A. 2003. Coarse woody debris: managing benefits and fire hazard in the recovering forest. Gen. Tech. Rep. RMRS-GTR-105. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station. 16 p. http://www.treesearch.fs.fed.us/pubs/5585

Brown, R.T.; Agee, J.K.; Franklin, J.F. 2004a. Forest restoration and fire: principles in the context of place. Conservation Biology. 18(4): 903-912. doi:10.1111/j.1523-1739.2004.521_1.x

Brown, T.J.; Hall, B.L.; Westerling, A.L. 2004b. The impact of twenty-first century climate change on wildland fire danger in the western United States: an applications perspective. Climatic Change. 62(1-3): 365-388. doi:10.1023/B:CLIM.0000013680.07783.de

Brubaker, L.B. 1978. Effects of defoliation by Douglas-fir tussock moth on ring sequences of Douglas-fir and grand fir. Tree-Ring Bulletin. 38: 49-60. http://www.treeringsociety.org/TRBTRR/TRBTRR.htm#Vol38

Brubaker, L.B.; Greene, S.K. 1979. Differential effects of Douglas-fir tussock moth and western spruce budworm defoliation on radial growth of grand fir and Douglas-fir. Canadian Journal of Forest Research. 9(1): 95-105. doi:10.1139/x79-016

Brudvig, L.A.; Wagner, S.A.; Damschen, E.I. 2012. Corridors promote fire via connectivity and edge effects. Ecological Applications. 22(3): 937-946. doi:10.1890/11-1026.1

Brundtland, G.H., Khalid, M. 1987. Report of the World Commission on Environment and Development: "Our Common Future." New York: United Nations.

Buckhorn, W.J. 1947. Second memorandum on the Douglas fir tussock moth outbreak near Troy, Oregon. Unpublished Typescript Report. Portland, OR: USDA Bureau of Entomology and Plant Quarantine, Forest Insect Laboratory. 8 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015606.pdf

Buckner, C.H. 1966. The role of vertebrate predators in the biological control of forest insects. Annual Review of Entomology. 11: 449-470. doi:10.1146/annurev.en.11.010166.002313

Budiansky, S. 1995. Nature's keepers: the new science of nature management. New York: The Free Press. 310 p. isbn:0-02-904915-6

Buhyoff, G.J.; Leuschner, W.A. 1978. Estimating psychological disutility from damaged forest stands. Forest Science. 24(3): 424-432. doi:10.1093/forestscience/24.3.424

Bulaon, B.; Sturdevant, N. 2006. Determining stand susceptibility to western spruce budworm and potential damaging effects. Numb. Rep. 06-07. Missoula, MT: USDA Forest Service, Northern Region, Forest Health Protection. 9 p.

Bull, E.L.; Heater, T.W. 2000. Resting and denning sites of American martens in northeastern Oregon. Northwest Science. 74(3): 179-185. http://hdl.handle.net/2376/1018

Bull, E.L.; Twombly, A.D.; Quigley, T.M. 1980. Perpetuating snags in managed mixed conifer forests of the Blue Mountains, Oregon. In: DeGraff, R.M.; Tilghman, N.G., comps. Workshop proceedings: management of western forest and grasslands for nongame birds. Gen. Tech. Rep. INT-86. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 325-336. https://www.fs.fed.us/rm/pubs_int/int_gtr086.pdf

Bull, E.L.; Torgersen, T.R.; Blumton, A.K.; McKenzie, C.M.; Wyland, D.S. 1995. Treatment of an old-growth stand and its effects on birds, ants, and large woody debris: a case study. Gen. Tech. Rep. PNW-GTR-353. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 12 p. http://www.treesearch.fs.fed.us/pubs/5621

Bull, E.L.; Parks, C.G.; Torgersen, T.R. 1997. Trees and logs important to wildlife in the interior Columbia River basin. Gen. Tech. Rep. PNW-GTR-391. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 55 p. https://www.fs.usda.gov/treesearch/pubs/3051

Bull, E.L.; Clark, A.A.; Shepherd, J.F. 2005. Short-term effects of fuel reduction on pileated woodpeckers in northeastern Oregon – a pilot study. Res. Pap. PNW-RP-564. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 17 p. http://www.treesearch.fs.fed.us/pubs/8915

Buma, B.; Wessman, C.A. 2013. Forest resilience, climate change, and opportunities for adaptation: A specific case of a general problem. Forest Ecology and Management. 306: 216-225. doi:10.1016/j.foreco.2013.06.044

Bunnell, F.L.; Johnson, J.F., eds. 2000. Policy and practices for biodiversity in managed forests: the living dance. Vancouver, BC: UBC Press. 162 p. isbn:0-7748-0691-5

Bunnell, F.L.; Kremsater, L.L. 1990. Sustaining wildlife in managed forests. Northwest Environmental Journal. 6(2): 243-269. Sustaining_wildlife_in_managed_forests

Burger, J.A. 2009. Management effects on growth, production and sustainability of managed forest ecosystems: past trends and future directions. Forest Ecology and Management. 258(10): 2335-2346. doi:10.1016/j.foreco.2009.03.015

Burgoyne, T.A.; DeLuca, T.H. 2009. Short-term effects of forest restoration management on non-symbiotic nitrogen-fixation in western Montana. Forest Ecology and Management. 258(7): 1369-1375. doi:10.1016/j.foreco.2009.06.048

Burke, H.E. 1990. Northeastern Oregon bark beetle control project 1910-11. Wickman, B.E., ed. Gen. Tech. Rep. PNW-GTR-249. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 46 p. http://www.treesearch.fs.fed.us/pubs/26285

Burns, R.M., comp. 1989. The scientific basis for silvicultural and management decisions in the national forest system. Gen. Tech. Rep. WO-55. Washington, DC: USDA Forest Service. 180 p. https://archive.org/download/CAT31125176/CAT31125176.pdf

Burns, M.; Cheng, A.S. 2007. Framing the need for active management for wildfire mitigation and forest restoration. Society and Natural Resources. 20(3): 245-259. doi:10.1080/08941920601117348

Burns, R.M.; Honkala, B.H. 1990a. Silvics of North America; Volume 1, conifers. Agric. Handb. 654. Washington, DC: USDA Forest Service. 675 p. https://www.treesearch.fs.fed.us/pubs/1547

Burns, R.M.; Honkala, B.H. 1990b. Silvics of North America; Volume 2, hardwoods. Agric. Handb. 654. Washington, DC: USDA Forest Service. 877 p. https://www.treesearch.fs.fed.us/pubs/1548

Burton, P.J.; Balisky, A.C.; Coward, L.P.; Cumming, S.G.; Kneeshaw, D.D. 1992. The value of managing for biodiversity. Forestry Chronicle. 68: 225-237. doi:10.5558/tfc68225-2

Burton, P.J.; Messier, C.; Smith, D.W.; Adamowicz, W.L., eds. 2003. Towards sustainable management of the boreal forest. Ottawa, ON: NRC Research Press. 1039 p. isbn:0-660-18762-0

Burton, P.J.; Parisien, M.; Hicke, J.A.; Hall, R.J.; Freeburn, J.T. 2008. Large fires as agents of ecological diversity in the North American boreal forest. International Journal of Wildland Fire. 17(6): 754-767. doi:10.1071/WF07149

Buse, L.J.; Perera, A.H., comps. 2002. Emulating natural forest landscape disturbances: concepts and applications. Popular summaries. Forest Research Infor. Pap. No. 149. Sault Ste. Marie, ON: Ontario Ministry of Natural Resources, Ontario Forest Research Institute. 100 p. http://www.ontla.on.ca/library/repository/mon/30010/225303.pdf

Busing, R.T.; White, P.S. 1993. Effects of area on old-growth forest attributes: implications for the equilibrium landscape concept. Landscape Ecology. 8(2): 119-126. doi:10.1007/BF00141591

Busse, M.D.; Hubbert, K.R.; Fiddler, G.O.; Shestak, C.J.; Powers, R.F. 2005. Lethal soil temperatures during burning of masticated forest residues. International Journal of Wildland Fire. 14(3): 267-276. doi:10.1071/WF04062

Butler, K.F.; Koontz, T.M. 2005. Theory into practice: Implementing ecosystem management objectives in the USDA Forest Service. Environmental Management. 35(2): 138-150. doi:10.1007/s00267-003-0312-y

Byler, J.W.; Zimmer-Grove, S. 1991. A forest health perspective on interior Douglas-fir management. In: Baumgartner, D.M.; Lotan, J.E., comps. Interior Douglas-fir: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Department of Natural Resource Sciences, Cooperative Extension: 103-108.

Cahill, D.B. 1960. The relationship of diameter to height of attack in lodgepole pine infested by mountain pine beetle. Res. Note No. 78. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 4 p. https://archive.org/download/relationshipofdi78cahi/relationshipofdi78cahi.pdf

Callicott, J.B.; Mumford, K. 1997. Ecological sustainability as a conservation concept. Conservation Biology. 11(1): 32-40. doi:10.1046/j.1523-1739.1997.95468.x

Camarero, J.J.; Sánchez-Salguero, R.; Sangüesa-Barreda, G.; Matías, L. 2018. Tree species from contrasting hydrological niches show divergent growth and water-use efficiency. Dendrochronologia. 52: 87-95. doi:10.1016/j.dendro.2018.10.003

Camp, A.E. 1999. Age structure and species composition changes resulting from altered disturbance regimes on the eastern slopes of the Cascades Range, Washington. Journal of Sustainable Forestry. 9(3/4): 39-67. doi:10.1300/J091v09n03_03

Camp, A.; Oliver, C.; Hessburg, P.; Everett, R. 1995. Spatial changes in forest landscape patterns from altered disturbance regimes on the eastern slope of the Washington Cascades. In: Brown, J.K.; Mutch, R.W.; Spoon, C.W.; Wakimoto, R.H., tech. coords. Proceedings: Symposium on fire in wilderness and park management. Gen. Tech. Rep. INT-GTR-320. Ogden, UT: USDA Forest Service, Intermountain Research Station: 169-172. https://archive.org/download/CAT10714714/CAT10714714.pdf

Camp, A.; Oliver, C.; Hessburg, P.; Everett, R. 1997. Predicting late-successional fire refugia pre-dating European settlement in the Wenatchee Mountains. Forest Ecology and Management. 95(1): 63-77. doi:10.1016/S0378-1127(97)00006-6

Campbell, S.; Liegel, L., tech. coords. 1996. Disturbance and forest health in Oregon and Washington. Gen. Tech. Rep. PNW-GTR-381. Portland, OR: USDA Forest Service, Pacific Northwest Research Station; Pacific Northwest Region; Oregon Department of Forestry; Washington Department of Natural Resources. 105 p. http://www.treesearch.fs.fed.us/pubs/25383

Campbell, R.W.; Torgersen, T.R. 1983. Compensatory mortality in defoliator population dynamics. Environmental Entomology. 12(3): 630-632. doi:10.1093/ee/12.3.630

Campbell, R.W.; Torgersen, T.R.; Srivastava, N. 1983. A suggested role for predaceous birds and ants in the population dynamics of the western spruce budworm. Forest Science. 29(4): 779-790. doi:10.1093/forestscience/29.4.779

Campbell, S.; Azuma, D.; Weyermann, D. 2003. Forests of eastern Oregon: an overview. Gen.

Tech. Rep. PNW-GTR-578. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 31 p. http://www.treesearch.fs.fed.us/pubs/5343

Campbell, R.; Smith, D.J.; Arsenault, A. 2005. Dendroentomological and forest management implications in the interior Douglas-fir zone of British Columbia, Canada. Dendrochronologia. 22(3): 135-140. doi:10.1016/j.dendro.2005.05.003

Campbell, R.; Smith, D.J.; Arsenault, A. 2006. Multicentury history of western spruce budworm outbreaks in interior Douglas-fir forests near Kamloops, British Columbia. Canadian Journal of Forest Research. 36(7): 1758-1769. doi:10.1139/x06-069

Canadell, J.G.; Raupach, M.R. 2008. Managing forests for climate change mitigation. Science. 320(5882): 1456-1457. doi:10.1126/science.1155458

Canadian Council of Forest Ministers. 2008. A vision for Canada's forests: 2008 and beyond. Ottawa, ON: Natural Resources Canada, Communications Branch. 15 p. https://www.ccfm.org/pdf/Vision_EN.pdf

Canelles, Q.; Aquilué, N.; Duane, A.; Brotons, L. 2019. From stand to landscape: modelling post-fire regeneration and species growth. Ecological Modelling. 404: 103-111. doi:10.1016/j.ecolmodel.2019.05.001

Canham, C.D.; Denslow, J.S.; Platt, W.J.; Runkle, James R.; Spies, Tom A.; White, Peter S. 1990. Light regimes beneath closed canopies and tree-fall gaps in temperate and tropical forest. Canadian Journal of Forest Research. 20(5): 620-631. doi:10.1139/x90-084

Canham, C.D.; Papaik, M.J.; Latty, E.F. 2001. Interspecific variation in susceptibility to windthrow as a function of tree size and storm severity for northern temperate tree species. Canadian Journal of Forest Research. 31(1): 1-10. doi:10.1139/x00-124

Cannell, M.G.R.; Dewar, R.C. 1994. Carbon allocation in trees: A review of concepts for modelling. Advances in Ecological Research. 25: 59-104. doi:10.1016/S0065-2504(08)60213-5

Cannon, J.B.; Henderson, S.K.; Bailey, M.H.; Peterson, C.J. 2019. Interactions between wind and fire disturbance in forests: Competing amplifying and buffering effects. Forest Ecology and Management. 436: 117-128. doi:10.1016/j.foreco.2019.01.015

Cansler, C.A.; McKenzie, D. 2014. Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade Range, USA. Ecological Applications. 24(5): 1037-1056. doi:10.1890/13-1077.1

Cantor, L.F.; Whitham, T.G. 1989. Importance of belowground herbivory: pocket gophers may limit aspen to rock outcrop refugia. Ecology. 70(4): 962-970. doi:10.2307/1941363

Caon, L.; Vallejo, V.R.; Ritsema, C.J.; Geissen, V. 2014. Effects of wildfire on soil nutrients in Mediterranean ecosystems. Earth-Science Reviews. 139: 47-58. doi:10.1016/j.earscirev.2014.09.001

Caraher, D.; Knapp, W.H. 1994. Assessing ecosystem health in the Blue Mountains. In: Foley, L.H., comp. Silviculture: from the cradle of forestry to ecosystem management; proceedings of the National Silviculture Workshop. Gen. Tech. Rep. SE-88. Asheville, NC: USDA Forest Service, Southeastern Forest Experiment Station: 34-41. https://www.fs.usda.gov/treesearch/pubs/132

Caraher, D.L.; Henshaw, J.; Hall, F.; Knapp, W.H.; McCammon, B.P.; Nesbitt, J.; Pedersen, R.J.; Regenovitch, I.; Tietz, C. 1992. Restoring ecosystems in the Blue Mountains: a report to the Regional Forester and the Forest Supervisors of the Blue Mountain forests. Portland, OR: USDA Forest Service, Pacific Northwest Region. 14 p (plus 5 appendices). http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015660.pdf

Carey, A.B. 1998. Ecological foundations of biodiversity: lessons from natural and managed forests of the Pacific Northwest. Northwest Science. 72(Special Issue 2): 127-133. https://www.fs.usda.gov/treesearch/pubs/5526

Carey, A.B. 2003. Restoration of landscape function: reserves or active management? Forestry. 76(2): 221-230. doi:10.1093/forestry/76.2.221

Carey, A.B. 2007. AIMing for healthy forests: active, intentional management for multiple values. Gen. Tech. Rep. PNW-GTR-721. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 447 p. https://www.fs.usda.gov/treesearch/pubs/29208

Carey, A.B.; Lippke, B.R.; Sessions, J. 1999. Intentional systems management: managing forests for biodiversity. Journal of Sustainable Forestry. 9(3): 83-125. doi:10.1300/J091v09n03_05

Carlson, C.E. 1989. Influence of habitat types on forest pests of the northern Rocky Mountains and classification based on vegetation: applications for resource management. In: Ferguson, D.E.; Morgan, P.; Johnson, F.D., comps. Proceedings; land classifications based on vegetation: applications for resource management. Gen. Tech. Rep. INT-257. Ogden, UT: USDA Forest Service, Intermountain Research Station: 186-192. https://archive.org/download/CAT91960227/CAT91960227.pdf

Carlson, C.E.; Cates, R.G. 1991. Thinning, foliage chemistry, and defoliation by budworm in immature northern Rocky Mountain Douglas-fir stands: a preliminary assessment. In: Baumgartner, D.M.; Lotan, J.E., comps. Interior Douglas-fir: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 135-140.

Carlson, C.E.; Lotan, J.E. 1988. Using stand culture techniques against defoliating insects. In: Schmidt, W.C., comp. Proceedings–future forests of the mountain west: a stand culture symposium. Gen. Tech. Rep. INT-243. Ogden, UT: USDA Forest Service, Intermountain Research Station: 275-277. https://archive.org/download/CAT88897341/CAT88897341.pdf

Carlson, M.; Kurz, W.A. 2007. Approximating natural landscape pattern using aggregated harvest. Canadian Journal of Forest Research. 37(10): 1846-1853. doi:10.1139/X07-059

Carlson, C.E.; Wulf, N.W. 1989. Silvicultural strategies to reduce stand and forest susceptibility to the western spruce budworm. Agric. Handb. 676. Washington, DC: USDA Forest Service, Cooperative State Research Service. 31 p. https://naldc.nal.usda.gov/download/CAT89916903/PDF

Carlson, C.E.; Fellin, D.G.; Schmidt, W.C. 1983. The western spruce budworm in northern Rocky Mountain forests: a review of ecology, insecticidal treatments and silvicultural practices. In: O'Loughlin, J.; Pfister, R.D., eds. Management of second-growth forests: the state of knowledge and research needs. Missoula, MT: Montana Forest and Conservation Experiment Station, School of Forestry, University of Montana: 76-103.

Carlson, C.E.; Pfister, R.D.; Theroux, L.J.; Fiedler, C.E. 1985a. Release of a thinned budworm-infested Douglas-fir/ponderosa pine stand. Res. Pap. INT-349. Ogden, UT: USDA Forest Service, Intermountain Research Station. 8 p. https://archive.org/download/releaseofthinned349carl/releaseofthinned349carl.pdf

Carlson, C.E.; Schmidt, W.C.; Fellin, D.C. 1985b. The role of fire in western spruce budworm dynamics: Is wilderness a factor? In: Lotan, J.E.; Kilgore, B.M.; Fischer, W.C.; Mutch, R.W., tech. coords. Proceedings–symposium and workshop on wilderness fire. Gen. Tech. Rep. INT-182. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 318-322. https://archive.org/download/CAT31118982/CAT31118982.pdf

Carolin, V.M.; Coulter, W.K. 1971. Trends of western spruce budworm and associated insects in Pacific Northwest forests sprayed with DDT. Journal of Economic Entomology. 64(1): 291-297. doi:10.1093/jee/64.1.291

Carpenter, S.; Walker, B.; Anderies, J.M.; Abel, N. 2001. From metaphor to measurement: Resilience of what to what? Ecosystems. 4(8): 765-781. doi:10.1007/s10021-001-0045-9

Carroll, M.S.; Blatner, K.A.; Alt, F.J.; Schuster, E.G.; Findley, A.J. 2000. Adaptation strategies of displaced Idaho woods workers: Results of a longitudinal panel study. Society and Natural Resources. 13(2): 95-113. doi:10.1080/089419200279135

Carroll, A.L.; Taylor, S.W.; Régniere, J.; Safranyik, L. 2004. Effects of climate change on range expansion by the mountain pine beetle in British Columbia. In: Shore, T.L.; Brooks, J.E.; Stone, J.E., eds. Mountain pine beetle symposium: challenges and solutions. Infor. Rep. BC-X-399. Victoria, BC: Natural Resources Canada, Canadian Forest Service, Pacific Forestry Centre: 223-232. https://cfs.nrcan.gc.ca/publications?id=25159

Carroll, M.S.; Blatner, K.A.; Cohn, P.J.; Morgan, T. 2007. Managing fire danger in the forests of the US inland northwest: A classic wicked problem in public land policy. Journal of Forestry. 105(5): 239-244. doi:10.1093/jof/105.5.239

Carson, R. 1962. Silent spring. Boston, MA: Houghton Mifflin.

Carson, R.J. 2018. The Blues: Natural history of the Blue Mountains of northeastern Oregon and southeastern Washington. Sandpoint, ID: Keokee Books. 208 p. isbn:978-1-879628-54-0

Carter, S.W., Jr. 1976. Northeast Oregon lodgepole pine harvest and rehabilitation plan; mountain pine beetle. Unpub. Rep. [Place of publication unknown]: USDA Forest Service, Pacific Northwest Region, Malheur, Umatilla, and Wallowa-Whitman national forests. 47 p.

Carter, G.A.; Smith, W.K.; Hadley, J.L. 1988. Stomatal conductance in three conifer species at different elevations during summer in Wyoming. Canadian Journal of Forest Research. 18(2): 242-246. doi:10.1139/x88-035

Carvell, K.L.; Perkey, A.W. 1997. Using diagnostic plants to evaluate site class. NA-TP-03-97. Morgantown, WV: USDA Forest Service, Northeastern Area, State & Private Forestry. 28 p. https://www.fs.usda.gov/naspf/sites/default/files/index.pdf

Cary, A. 1936. White pine and fire. Journal of Forestry. 34(1): 62-65. doi:10.1093/jof/34.1.62

Case, R.L.; Kauffman, J.B. 1997. Wild ungulate influences on the recovery of willows, black cottonwood and thin-leaf alder following cessation of cattle grazing in northeastern Oregon. Northwest Science. 71(2): 115-126. http://hdl.handle.net/2376/1249

Castello, J.D.; Teale, S.A. 2011. Forest health: an integrated perspective. New York: Cambridge University Press. 392 p. isbn:978-0-521-74741-7

Castello, J.D.; Leopold, D.J.; Smallidge, P.J. 1995. Pathogens, patterns, and processes in forest ecosystems. BioScience. 45(1): 16-24. doi:10.2307/1312531

Cates, R.G.; Redak, R.A.; Henderson, C.B. 1983. Patterns in defensive natural product chemistry: Douglas fir and western spruce budworm interactions. In: Hedin, P.A., ed. Plant resistance to insects. ACS Symp. Series, Vol. 208, American Chemical Society: 3-19.

Cates, R.G.; Zou, J.; Carlson, C. 1991. The role of variation in Douglas-fir foliage quality in the silvicultural management of the western spruce budworm. In: Baumgartner, D.M.; Lotan, J.E., comps. Interior Douglas-fir: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 115-127.

Cavard, X.; Bergeron, Y.; Chen, H.Y.H.; Paré, D.; Laganière, J.; Brassard, B. 2011. Competition and facilitation between tree species change with stand development. Oikos. 120(11): 1683-1695. doi:10.1111/j.1600-0706.2011.19294.x

Chambers, J.C.; Allen, C.R.; Cushman, S.A. 2020. Operationalizing the concepts of resilience and resistance for managing ecosystems and species at risk. Lausanne, Switzerland: Frontiers Media SA. 223 p. doi:10.3389/978-2-88963-867-3

Chabot, B.F.; Hicks, D.J. 1982. The ecology of leaf life spans. Annual Review of Ecology and Systematics. 13: 229-259. doi:10.1146/annurev.es.13.110182.001305

Chapin, F.S. III; Van Cleve, K. 1981. Plant nutrient absorption and retention under differing fire regimes. In: Mooney, H.A.; Bonnicksen, T.M.; Christensen, N.L.; Lotan, J.E.; Reiners, W.A., tech. coords. Fire regimes and ecosystem properties; proceedings of the conference. Gen. Tech. Rep. WO-26. [Washington, DC]: USDA Forest Service: 301-321. https://archive.org/download/CAT83781017/CAT83781017.pdf

Chapin, F.S. III; Torn, M.S.; Tateno, M. 1996. Principles of ecosystem sustainability. American Naturalist. 148(6): 1016-1037. doi:10.2307/2463560

Chapin, F.S. III; Matson, P.A.; Mooney, H.A. 2002. Principles of terrestrial ecosystem ecology. New York: Springer-Verlag. 436 p. isbn:0-387-95443-0

Chapin, F.S. III; Kofinas, G.P.; Folke, C., eds. 2009. Principles of ecosystem stewardship: resilience-based natural resource management in a changing world. New York: Springer. 401 p. isbn:978-0-387-73032-5

Chapin, F.S. III; Carpenter, S.R.; Kofinas, G.P.; Folke, C.; Abel, N.; Clark, W.C.; Olsson, P.; Smith, D.M.S.; Walker, B.; Young, O.R.; Berkes, F.; Biggs, R.; Grove, J.M.; Naylor, R.L.; Pinkerton, E.; Steffen, W.; Swanson, F.J. 2010. Ecosystem stewardship: sustainability strategies for a rapidly changing planet. Trends in Ecology & Evolution. 25(4): 241-249. doi:10.1016/j.tree.2009.10.008

Chapin, F.S.; Pickett, S.T.A.; Power, M.E.; Jackson, R.B.; Carter, D.M.; Duke, C. 2011. Earth stewardship: a strategy for social–ecological transformation to reverse planetary degradation. Journal of Environmental Studies and Sciences. 1(1): 44-53. doi:10.1007/s13412-011-0010-7

Chapman, W.K.; Schellenberg, B.; Newsome, T.A. 2011. Assessment of armillaria root disease infection in stands in south-central British Columbia with varying levels of overstory retention, with and without pushover logging. Canadian Journal of Forest Research. 41(8): 1598-1605. doi:10.1139/x11-085

Chappell, H.N.; Weetman, G.F.; Miller, R.E., eds. 1992. Forest fertilization: sustaining and improving nutrition and growth of western forests. Contribution No. 73. Seattle, WA: University of Washington, College of Forest Resources. 302 p.

Charnley, S.; Fischer, A.P.; Jones, E.T. 2007. Integrating traditional and local ecological knowledge into forest biodiversity conservation in the Pacific Northwest. Forest Ecology and Management. 246(1): 14-28. doi:10.1016/j.foreco.2007.03.047

Chase, A. 2001. In a dark wood: the fight over forests and the myths of nature. New Brunswick, NJ: Transaction Publishers. 535 p. isbn:0-7658-0752-1

Chauvin, T.; Cochard, H.; Segura, V.; Rozenberg, P. 2019. Native-source climate determines the Douglas-fir potential of adaptation to drought. Forest Ecology and Management. 444: 9-20. doi:10.1016/j.foreco.2019.03.054

Chazdon, R.L.; Pearcy, R.W. 1991. The importance of sunflecks for forest understory plants. BioScience. 41(11): 760-766. doi:10.2307/1311725

Chen, Z.; Kolb, T.E.; Clancy, K.M.; Hipkins, V.D.; DeWald, L.E. 2001a. Allozyme variation in interior Douglas-fir: association with growth and resistance to western spruce budworm herbivory. Canadian Journal of Forest Research. 31(10): 1691-1700. doi:10.1139/x01-107

Chen, Z.; Kolb, T.E.; Clancy, K.M. 2001b. Mechanisms of Douglas-fir resistance to western spruce budworm defoliation: bud burst phenology, photosynthetic compensation and growth rate. Tree Physiology. 21(16): 1159-1169. doi:10.1093/treephys/21.16.1159

Chen, Z.; Kolb, T.E.; Clancy, K.M. 2002. The role of monoterpenes in resistance of Douglas fir to western spruce budworm defoliation. Journal of Chemical Ecology. 28(5): 897-920. doi:10.1023/a:1015297315104

Chen, H.Y.H.; Klinka, K.; Mathey, A.-H.; Wang, X.; Varga, P.; Chourmouzis, C. 2003. Are mixed-species stands more productive than single-species stands: an empirical test of three forest types in British Columbia and Alberta. Canadian Journal of Forest Research. 33(7): 1227-1237. doi:10.1139/x03-048

Cheng, A.S.; Daniels, S.E. 2003. Examining the interaction between geographic scale and ways of knowing in ecosystem management: a case study of place-based collaborative planning. Forest Science. 49(6): 841-854. doi:10.1093/forestscience/49.6.841

Childs, T.W.; Bedwell, J.L.; Englerth, G.H. 1938. Blister rust infection on *Pinus albicaulis* in the northwest. Plant Disease Reporter. 22(8): 139-140. Childs et al 1938

Chmura, D.J.; Anderson, P.D.; Howe, G.T.; Harrington, C.A.; Halofsky, J.E.; Peterson, D.L.; Shaw, D.C.; St.Clair, J.B. 2011. Forest responses to climate change in the northwestern United States: ecophysiological foundations for adaptive management. Forest Ecology and Management. 261(7): 1121-1142. doi:10.1016/j.foreco.2010.12.040

Choat, B.; Jansen, S.; Brodribb, T.J.; Cochard, H.; Delzon, S.; Bhaskar, R.; Bucci, S.J.; Feild, T.S.; Gleason, S.M.; Hacke, U.G.; Jacobsen, A.L.; Lens, F.; Maherali, H.; Martinez-Vilalta, J.; Mayr, S.; Mencuccini, M.; Mitchell, P.J.; Nardini, A.; Pittermann, J.; Pratt, R.B.; Sperry, J.S.; Westoby, M.; Wright, I.J.; Zanne, A.E. 2012. Global convergence in the vulnerability of forests to drought. Nature. 491(7426): 752-755. doi:10.1038/nature11688

Choi, Y.D. 2007. Restoration ecology to the future: a call for new paradigm. Restoration Ecology. 15(2): 351-353. doi:10.1111/j.1526-100X.2007.00224.x

Chojnacky, D.C.; Bentz, B.J.; Logan, J.A. 2000. Mountain pine beetle attack in ponderosa pine: comparing methods for rating susceptibility. Res. Pap. RMRS-RP-26. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station. 10 p. https://www.fs.usda.gov/treesearch/pubs/4621

Christensen, N.L., Jr. 1997. Managing for heterogeneity and complexity on dynamic landscapes. In: Pickett, S.T.A.; Ostfeld, R.S.; Shachak, M.; Likens, G.E., eds. The ecological basis of conservation: heterogeneity, ecosystems, and biodiversity. New York: Chapman & Hall: 167-186. isbn:0-412-09851-2

Christensen, N.L.; Bartuska, A.M.; Brown, J.H.; Carpenter, S.; D'Antonio, C.; Francis, R.; Franklin, J.F.; MacMahon, J.A.; Noss, R.F.; Parsons, D.J.; Peterson, C.H.; Turner, M.G.; Woodmansee, R.G. 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecological Applications. 6(3): 665-691. doi:10.2307/2269460

Christensen, N.L., Jr.; Gregory, S.V.; Hagenstein, P.R.; Heberlein, T.A.; Hendee, J.C.; Olson, J.T.; Peek, J.M.; Perry, D.A.; Schowalter, T.D.; Sullivan, K.; Tilman, G.D.; Vogt, K.A. 2000. Environmental issues in Pacific Northwest forest management. Washington, DC: National Academy Press. 259 p. isbn:0-309-05328-5

Christensen, G.A.; Dunham, P.; Powell, D.C.; Hiserote, B. 2007. Forest resources of the Umatilla National Forest. Res. Bull. PNW-RB-253. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 38 p. http://www.treesearch.fs.fed.us/pubs/27656

Christiansen, E.; Waring, R.H.; Berryman, A.A. 1987. Resistance of conifers to bark beetle attack: searching for general relationships. Forest Ecology and Management. 22(1-2): 89-106. doi:10.1016/0378-1127(87)90098-3

Ciccarese, L.; Mattsson, A.; Pettenella, D. 2012. Ecosystem services from forest restoration: thinking ahead. New Forests. 43(5-6): 543-560. doi:10.1007/s11056-012-9350-8

Cissel, J.H.; Swanson, F.J.; McKee, W.A.; Burditt, A.L. 1994. Using the past to plan the future in the Pacific Northwest. Journal of Forestry. 92(8): 30-46. doi:10.1093/jof/92.8.30

Clancy, K.M.; Itami, J.K.; Huebner, D.P. 1993. Douglas-fir nutrients and terpenes: potential resistance factors to western spruce budworm defoliation. Forest Science. 39(1): 78-94. doi:10.1093/forestscience/39.1.78

Clark, J.S. 1991. Disturbance and tree life history on the shifting mosaic landscape. Ecology. 72(3): 1102-1118. doi:10.2307/1940609

Clark, J.; Bobbe, T. 2007. Using remote sensing to map and monitor fire damage in forest ecosystems. In: Wulder, M.A.; Franklin, S.E., eds. Understanding forest disturbance and spatial pattern: Remote sensing and GIS approaches. Boca Raton, FL: CRC Press: 113-131. isbn:0-8493-3425-X

Clark, L.R.; Sampson, R.N. 1995. Forest ecosystem health in the inland west: a science and policy reader. Washington, DC: American Forests, Forest Policy Center. 37 p.

Clarke, S.E.; Bryce, S.A., eds. 1997. Hierarchical subdivisions of the Columbia Plateau and Blue Mountains ecoregions, Oregon and Washington. Gen. Tech. Rep. PNW-GTR-395. Portland (OR): USDA Forest Service, Pacific Northwest Research Station. 114 p. https://www.fs.usda.gov/treesearch/pubs/7595

Clausnitzer, R.R. 1993. The grand fir series of northeastern Oregon and southeastern Washington: successional stages and management guide. Tech. Pub. R6-ECO-TP-050-93. Baker City, OR: USDA Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest. 193 p. https://ir.library.oregonstate.edu/downloads/8s45qb336

Cleary, M.; van der Kamp, B.; Morrison, D. 2008. Armillaria root disease stand establishment decision aid. BC Journal of Ecosystems and Management 9(2): 60-65. https://jem-online.org/index.php/jem/article/download/397/312

Clewell, A.; Aronson, J. 2013. The SER primer and climate change. Ecological Management and Restoration. 14(3): 182-186. doi:10.1111/emr.12062

Climate Central. 2012. The age of western wildfires. Princeton, NJ: Climate Central (www.climatecentral.org). 18 p.

Clinton, W.J. 1999. Executive order 13112 of February 3, 1999 – invasive species. 7 p.

Coates, K.D. 2002. Tree recruitment in gaps of various size, clearcuts and undisturbed mixed forest of interior British Columbia, Canada. Forest Ecology and Management. 155(1-3): 387-398. doi:10.1016/S0378-1127(01)00574-6

Coates, K.D.; Burton, P.J. 1997. A gap-based approach for development of silvicultural systems to address ecosystem management objectives. Forest Ecology and Management. 99(3): 337-354. doi:10.1016/S0378-1127(97)00113-8

Cobb, D.F.; O'Hara, K.L.; Oliver, C.D. 1993. Effects of variations in stand structure on development of mixed-species stands in eastern Washington. Canadian Journal of Forest Research. 23(3): 545-552. doi:10.1139/x93-072

Cobb, R.C.; Meentemeyer, R.K.; Rizzo, D.M. 2010. Apparent competition in canopy trees determined by pathogen transmission rather than susceptibility. Ecology. 91(2): 327-333. doi:10.1890/09-0680.1

Cochran, P.H. 1992. Stocking levels and underlying assumptions for uneven-aged ponderosa pine stands. Res. Note PNW-RN-509. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 10 p. http://www.treesearch.fs.fed.us/pubs/25110

Cochran, P.H. 1998. Examples of mortality and reduced annual increments of white fir induced by drought, insects, and disease at different stand densities. Res. Note PNW-RN-525. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 19 p. http://www.treesearch.fs.fed.us/pubs/3036

Cochran, P.H.; Barrett, J.W. 1993. Long-term response of planted ponderosa pine to thinning in Oregon's Blue Mountains. Western Journal of Applied Forestry. 8(4): 126-132. doi:10.1093/wjaf/8.4.126

Cochran, P.H.; Berntsen, C.M. 1973. Tolerance of lodgepole and ponderosa pine seedlings to low night temperatures. Forest Science. 19(4): 272-280. doi:10.1093/forestscience/19.4.272

Cochran, P.H.; Dahms, W.G. 1998. Lodgepole pine development after early spacing in the Blue Mountains of Oregon. Res. Pap. PNW-RP-503. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 24 p. http://www.treesearch.fs.fed.us/pubs/2882

Cochran, P.H.; Dahms, W.G. 2000. Growth of lodgepole pine thinned to various densities on two sites with differing productivities in central Oregon. Res. Pap. PNW-RP-520. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 59 p. https://www.fs.usda.gov/treesearch-beta/pubs/2933

Cochran, P.H.; Seidel, K.W. 1995. Growth of western larch under controlled levels of stocking. In: Schmidt, W.C.; McDonald, K.J., comps. Ecology and management of Larix forests: a look ahead; proceedings of an international symposium. Gen. Tech. Rep. GTR-INT-319. Ogden, UT: USDA Forest Service, Intermountain Research Station: 285-292. https://archive.org/download/CAT10699418/CAT10699418.pdf

Cochran, P.H.; Seidel, K.W. 1999. Growth and yield of western larch under controlled levels of stocking in the Blue Mountains of Oregon. Res. Pap. PNW-RP-517. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 35 p. http://www.treesearch.fs.fed.us/pubs/2912

Cochran, P.H.; Geist, J.M.; Clemens, D.L.; Clausnitzer, R.R.; Powell, D.C. 1994. Suggested stocking levels for forest stands in northeastern Oregon and southeastern Washington. Res. Note PNW-RN-513. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 21 p. http://www.treesearch.fs.fed.us/pubs/25113

Cochrane, M.A.; Moran, C.J.; Wimberly, M.C.; Baer, A.D.; Finney, M.A.; Beckendorf, K.L.; Eidenshink, J.; Zhu, Z. 2012. Estimation of wildfire size and risk changes due to fuels treatments. International Journal of Wildland Fire. 21(4): 357-367. doi:10.1071/WF11079

Cohn, J.P. 2008. Citizen science: Can volunteers do real research? BioScience. 58(3): 192-197. doi:10.1641/B580303

Cohn, G.M.; Parsons, R.A.; Heyerdahl, E.K.; Gavin, D.G.; Flower, A. 2014. Simulated western spruce budworm defoliation reduces torching and crowning potential: a sensitivity analysis using a physics-based fire model. International Journal of Wildland Fire. 23(5): 709-720. doi:10.1071/WF13074

Cole, W.E. 1973. Interaction between mountain pine beetle and dynamics of lodgepole pine stands. Res. Note INT-170. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 6 p. https://archive.org/download/interactionbetwe170cole/interactionbetwe170cole.pdf

Cole, D.M. 1978a. Feasibility of silvicultural practices for reducing losses to the mountain pine beetle in lodgepole pine forests. In: Kibbee, D.L.; Berryman, A.A.; Amman, G.D.; Stark, R.W., tech. eds. Theory and practice of mountain pine beetle management in lodgepole pine forest. Moscow, ID: University of Idaho, Forest, Wildlife and Range Experiment Station: 140-147. Theory_and_Practice_of_Mountain_Pine_Beetle

Cole, D.M. 1978b. Management strategies for preventing mountain pine beetle epidemics in lodgepole pine stands: Based on empirical models. In: Kibbee, D.L.; Berryman, A.A.; Amman, G.D.; Stark, R.W., tech. eds. Theory and practice of mountain pine beetle management in lodgepole pine forest. Moscow, ID: University of Idaho, Forest, Wildlife and Range Experiment Station: 87-97. Theory_and_Practice_of_Mountain_Pine_Beetle

Cole, D.M. 1983. Canopy development in lodgepole pine: Implications for wildlife studies and multiple resource management. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 13 p. Canopy_Development_in_Lodgepole_Pine

Cole, D.M. 1984. Crop-tree thinning a 50-year-old western larch stand: 25-year results. Res. Pap. INT-328. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 9 p. Crop_tree_Thinning_a_50_year_old_Western

Cole, D.M. 1986. An indirect method for determining the order of expected thinning response among overstocked lodgepole pine stands. Canadian Journal of Forest Research. 16(5): 875-879. doi:10.1139/x86-156

Cole, D.M. 1987. Ranking thinning potential of lodgepole pine stands. Gen. Tech. Rep. INT-229. Ogden, UT: USDA Forest Service, Intermountain Research Station. 14 p. Ranking_Thinning_Potential_of_Lodgepole.pine

Cole, D.M. 1993a. Problems in lodgepole pine thinnngs: basal live limbs and fill-in regeneration. Res. Pap. INT-466. Ogden, UT: USDA Forest Service, Intermountain Research Station. 6 p. https://archive.org/download/problemsinlodgep466cole/problemsinlodgep466cole.pdf

Cole, D.M. 1993b. Trials of mixed-conifer plantings for increasing diversity in the lodgepole pine type. Res. Note INT-412. Ogden, UT: USDA Forest Service, Intermountain Research Station. 9 p. https://archive.org/download/trialsofmixedcon412cole/trialsofmixedcon412cole.pdf

Cole, W.E.; Amman, G.D. 1969. Mountain pine beetle infestations in relation to lodgepole pine diameters. Res. Note INT-95. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 7 p. https://www.fs.usda.gov/treesearch/pubs/24634

Cole, W.E.; Cahill, D.B. 1976. Cutting strategies can reduce probabilities of mountain pine beetle epidemics in lodgepole pine. Journal of Forestry. 74(5): 294-297. doi:10.1093/jof/74.5.294

Cole, W.E.; Guymon, E.P.; Jensen, C.E. 1981. Monoterpenes of lodgepole pine phloem as related to mountain pine beetles. Res. Pap. INT-281. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 10 p. https://www.biodiversitylibrary.org/itempdf/137084

Cole, W.E.; Amman, G.D.; Jensen, C.E. 1985. Mountain pine beetle dynamics in lodgepole pine forests. Part III: Sampling and modeling of mountain pine beetle populations. Gen. Tech. Rep. INT-188. Ogden, UT: USDA Forest Service, Intermountain Research Station. 46 p. https://www.biodiversitylibrary.org/itempdf/179286

Coley, P.D.; Bryant, J.P.; Chapin, F.S. 1985. Resource availability and plant antiherbivore defense. Science. 230(4728): 895-899. doi:10.1126/science.230.4728.895

Collins, B.M.; Stephens, S.L. 2010. Stand-replacing patches within a 'mixed severity' fire regime: quantitative characterization using recent fires in a long-established natural fire area. Landscape Ecology. 25(6): 927-939. doi:10.1007/s10980-010-9470-5

Collins, B.M.; Omi, P.N.; Chapman, P.L. 2006. Regional relationships between climate and wildfire-burned area in the interior West, USA. Canadian Journal of Forest Research. 36(3): 699-709. doi:10.1139/x05-264

Collins, B.M.; Miller, J.D.; Thode, A.E.; Kelly, M.; van Wagtendonk, J.W.; Stephens, S.L. 2009. Interactions among wildland fires in a long-established Sierra Nevada natural fire area. Ecosystems. 12(1): 114-128. doi:10.1007/s10021-008-9211-7

Collins, B.M.; Stephens, S.L.; Moghaddas, J.J.; Battles, J. 2010a. Challenges and approaches in planning fuel treatments across fire-excluded forested landscapes. Journal of Forestry. 108(1): 24-31. doi:10.1093/jof/108.1.24

Collins, B.J.; Rhoades, C.C.; Underhill, J.; Hubbard, R.M. 2010b. Post-harvest seedling recruitment following mountain pine beetle infestation of Colorado lodgepole pine stands: a comparison using historic survey records. Canadian Journal of Forest Research. 40(12): 2452-2456. doi:10.1139/X10-172

Collins, B.M.; Everett, R.G.; Stephens, S.L. 2011a. Impacts of fire exclusion and recent managed fire on forest structure in old growth Sierra Nevada mixed-conifer forests. Ecosphere. 2(4): art51. doi:10.1890/ES11-00026.1

Collins, B.J.; Rhoades, C.C.; Hubbard, R.M.; Battaglia, M.A. 2011b. Tree regeneration and future stand development after bark beetle infestation and harvesting in Colorado lodgepole pine stands. Forest Ecology and Management. 261(11): 2168-2175. doi:10.1016/j.foreco.2011.03.016

Collins, B.J.; Rhoades, C.C.; Battaglia, M.A.; Hubbard, R.M. 2012. The effects of bark beetle outbreaks on forest development, fuel loads and potential fire behavior in salvage logged and untreated lodgepole pine forests. Forest Ecology and Management. 284: 260-268. doi:10.1016/j.foreco.2012.07.027

Collins, B.M.; Kramer, H.A.; Menning, K.; Dillingham, C.; Saah, D.; Stine, P.A.; Stephens, S.L. 2013. Modeling hazardous fire potential within a completed fuel treatment network in the northern Sierra Nevada. Forest Ecology and Management. 310: 156-166. doi:10.1016/j.foreco.2013.08.015

Collins, B.M.; Lydersen, J.M.; Fry, D.L.; Wilkin, K.; Moody, T.; Stephens, S.L. 2016. Variability in vegetation and surface fuels across mixed-conifer-dominated landscapes with over 40 years of natural fire. Forest Ecology and Management. 381: 74-83. doi:10.1016/j.foreco.2016.09.010

Collins, B.M.; Fry, D.L.; Lydersen, J.M.; Everett, R.; Stephens, S.L. 2017. Impacts of different land management histories on forest change. Ecological Applications. 27(8): 2475-2486. doi:10.1002/eap.1622

Coman, W.E. 1911. Did the Indian protect the forest? Pacific Monthly. 26(3): 300-306.

Comer, P.; Faber-Langendoen, D.; Evans, R.; Gawler, S.; Josse, C.; Kittel, G.; Menard, S.; Pyne, M.; Reid, M.; Schulz, K.; Snow, K.; Teague, J. 2003. Ecological systems of the United States: a working classification of U.S. terrestrial ecosystems. Arlington, VA: NatureServe. 75 p.

Commoner, B. 1971. The closing circle: nature, man and technology. In: Cahn, M.A.; O'Brien, R., eds. Thinking about the environment. Armonk, NY: M.E. Sharpe, Inc.: 161-166.

Concannon, J.A.; Shafer, C.L.; DeVelice, R.L.; Sauvajot, R.M.; Boudreau, S.L.; DeMeo, T.E.; Dryden, J. 1999. Describing landscape diversity: a fundamental tool for ecosystem management. In: Szaro, R.C.; Johnson, N.C.; Sexton, W.T.; Malk, A.J., eds. Ecological stewardship: a common reference for ecosystem management. Volume II. Oxford, UK: Elsevier Science, Ltd: 195-218. isbn:0-08-043206-9

Conedera, M.; Tinner, W.; Neff, C.; Meurer, M.; Dickens, A.F.; Krebs, P. 2009. Reconstructing past fire regimes: methods, applications, and relevance to fire management and conservation. Quaternary Science Reviews. 28(5-6): 555-576. doi:10.1016/j.quascirev.2008.11.005

Connell, J.H.; Slatyer, R.O. 1977. Mechanisms of succession in natural communities and their role in community stability and organization. American Naturalist. 111(982): 1119-1144. www.jstor.org/stable/2460259

Conner, R.N.; Miller, O.K., Jr.; Adkisson, C.S. 1976. Woodpecker dependence on trees infected by fungal heart rots. Wilson Bulletin. 88(4): 575-581. https://sora.unm.edu/sites/default/files/journals/wilson/v088n04/p0575-p0581.pdf

Converse, S.J.; White, G.C.; Farris, K.L.; Zack, S. 2006. Small mammals and forest fuel reduction: National-scale responses to fire and fire surrogates. Ecological Applications. 16(5): 1717-1729. doi:10.1890/1051-0761(2006)016[1717:Smaffr]2.0.Co;2

Cook, S.F. 1955. The epidemic of 1830-1833 in California and Oregon. University of California Publications in American Archaeology and Ethnology. 43(3): 303-326. Cook 1955

Cook, J.E. 1996. Implications of modern successional theory for habitat typing: a review. Forest Science. 42(1): 67-75. doi:10.1093/forestscience/42.1.67

Cook, J.G.; Irwin, L.L.; Bryant, L.D.; Riggs, R.A.; Thomas, J.W. 1998. Relations of forest cover and condition of elk: a test of the thermal cover hypothesis in summer and winter. Wildlife Monographs. 141: 1-61. http://www.jstor.org/stable/3830857

Cook, E.R.; Woodhouse, C.A.; Eakin, C.M.; Meko, D.M.; Stahle, D.W. 2004. Long-term aridity changes in the western United States. Science. 306(5698): 1015-1018. doi:10.1126/science.1102586

Cooke, B.J.; Nealis, V.G.; Régnière, J. 2007. Insect defoliators as periodic disturbances in northern forest ecosystems. In: Johnson, E.A.; Miyanishi, K., eds. Plant disturbance ecology: the process and the response. Amsterdam: Academic Press: 487-525. isbn:978-0-12-088778-1

Coomes, D.A.; Grubb, P.J. 2000. Impacts of root competition in forests and woodlands: a theoretical framework and review of experiments. Ecological Monographs. 70(2): 171-207. doi:10.1890/0012-9615(2000)070[0171:IORCIF]2.0.CO;2

Cooper, C.F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecological Monographs. 30(2): 129-164. doi:10.2307/1948549

Cooper, S.V.; Neiman, K.E.; Roberts, D.W. 1991. Forest habitat types of northern Idaho: a second approximation. Gen. Tech. Rep. INT-236. Ogden, UT: USDA Forest Service, Intermountain Research Station. 143 p. https://www.fs.usda.gov/treesearch/pubs/24623

Cooper, L.A.; Reed, C.C.; Ballantyne, A.P. 2018. Mountain pine beetle attack faster growing lodgepole pine at low elevations in western Montana, USA. Forest Ecology and Management. 427: 200-207. doi:10.1016/j.foreco.2018.05.048

Coops, N.; Waring, R. 2011. A process-based approach to estimate lodgepole pine (*Pinus contorta*; Dougl.) distribution in the Pacific Northwest under climate change. Climatic Change. 105(1): 313-328. doi:10.1007/s10584-010-9861-2

Coops, N.C.; Waring, R.H.; Wulder, M.A.; White, J.C. 2009. Prediction and assessment of bark beetle-induced mortality of lodgepole pine using estimates of stand vigor derived from remotely sensed data. Remote Sensing of Environment. 113(5): 1058-1066. doi:10.1016/j.rse.2009.01.013

Corlett, R.T. 2015. The Anthropocene concept in ecology and conservation. Trends in Ecology and Evolution. 30(1): 36-41. doi:10.1016/j.tree.2014.10.007

Costanza, R.; Cornwell, L. 1992. The 4P approach to dealing with scientific uncertainty. Environment. 34(9): 12-42. doi:10.1080/00139157.1992.9930930

Couillard, P.-L.; Tremblay, J.; Lavoie, M.; Payette, S. 2019. Comparative methods for reconstructing fire histories at the stand scale using charcoal records in peat and mineral soils. Forest Ecology and Management. 433: 376-385. doi:10.1016/j.foreco.2018.11.015

Courty, P.-E.; Buée, M.; Diedhiou, A.G.; Frey-Klett, P.; Le Tacon, F.; Rineau, F.; Turpault, M.-P.; Uroz, S.; Garbaye, J. 2010. The role of ectomycorrhizal communities in forest ecosystem processes: new perspectives and emerging concepts. Soil Biology and Biochemistry. 42(5): 679-698. doi:10.1016/j.soilbio.2009.12.006

Covington, W.W.; Everett, R.L.; Steele, R.; Irwin, L.L.; Daer, T.A.; Auclair, A.N.D. 1994. Historical and anticipated changes in forest ecosystems of the inland west of the United States. Journal of Sustainable Forestry. 2(1/2): 13-63. doi:10.1300/J091v02n01_02

Covington, W.; Niering, W.A.; Starkey, E.; Walker, J. 1999. Ecosystem restoration and management: scientific principles and concepts. In: Szaro, R.C.; Johnson, N.C.; Sexton, W.T.; Malk, A.J., eds. Ecological stewardship: a common reference for ecosystem management. Volume II. Oxford, UK: Elsevier Science, Ltd: 599-617. isbn:0-08-043206-9

Cowlin, R.W.; Briegleb, P.A.; Moravets, F.L. 1942. Forest resources of the ponderosa pine region of Washington and Oregon. Misc. Pub. No. 490. Washington, DC: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 99 p. https://archive.org/download/forestresourceso490cowl/forestresourceso490cowl.pdf

Coyea, M.R.; Margolis, H.A. 1994. The historical reconstruction of growth efficiency and its relationship to tree mortality in balsam fir ecosystems affected by spruce budworm. Canadian Journal of Forest Research. 24(11): 2208-2221. doi:10.1139/x94-284

Crane, M.F.; Fischer, W.C. 1986. Fire ecology of the forest habitat types of central Idaho. Gen. Tech. Rep. INT-218. Ogden, UT: USDA Forest Service, Intermountain Research Station. 86 p. http://www.treesearch.fs.fed.us/pubs/32956

Cronon, W., ed. 1996. Uncommon ground: rethinking the human place in nature. New York: W.W. Norton & Company. 561 p. isbn:0-393-31511-8

Crookston, N.L.; Rehfeldt, G.E.; Dixon, G.E.; Weiskittel, A.R. 2010. Addressing climate change in the forest vegetation simulator to assess impacts on landscape forest dynamics. Forest Ecology and Management. 260(7): 1198-1211. doi:10.1016/j.foreco.2010.07.013

Crotteau, J.S.; Morgan Varner Iii, J.; Ritchie, M.W. 2013. Post-fire regeneration across a fire severity gradient in the southern Cascades. Forest Ecology and Management. 287: 103-112. doi:10.1016/j.foreco.2012.09.022

Crotteau, J.S.; Keyes, C.R.; Hood, S.M.; Larson, A.J.; Sutherland, E.K.; Wright, D.K.; Egan, J.M. 2018. Stand dynamics 11 years after retention harvest in a lodgepole pine forest. Forest Ecology and Management. 427: 169-181. doi:10.1016/j.foreco.2018.05.013

Crouch, G.L. 1982. Pocket gophers and reforestation on western forests. Journal of Forestry. 80(10): 662-664. doi:10.1093/jof/80.10.662

Crouch, G.L.; Frank, L.R. 1979. Poisoning and trapping pocket gophers to protect conifers in northeastern Oregon. Res. Pap. PNW-261. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 8 p. https://archive.org/download/CAT92272904/CAT92272904.pdf

Crow, T.R.; Gustafson, E.J. 1997. Ecosystem management: managing natural resources in time and space. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press: 215-228. isbn:1-55963-399-9

Crow, T.R.; Perera, A.H. 2004. Emulating natural landscape disturbance in forest management – an introduction. Landscape Ecology. 19(3): 231-233. doi:10.1023/B:LAND.0000030762.86156.5d

Crowe, E.A.; Clausnitzer, R.R. 1997. Mid-montane wetland plant associations of the Malheur, Umatilla and Wallowa-Whitman National Forests. Tech. Pap. R6-NR-ECOL-TP-22-97. Baker City, OR: USDA Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest. 299 p. MidMontaneWetlandPlantAssociationsWallowaWhitnf

Cruickshank, M.G.; Morrison, D.J.; Lalumière, A. 2009. The interaction between competition in interior Douglas-fir plantations and disease caused by *Armillaria ostoyae* in British Columbia. Forest Ecology and Management. 257(2): 443-452. doi:10.1016/j.foreco.2008.09.013

Cui, W.; Perera, A.H. 2008. What do we know about forest fire size distribution, and why is this knowledge useful for forest management? International Journal of Wildland Fire. 17(2): 234-244. doi:10.1071/WF06145

Cullen, S.J.; Montagne, C.; Ferguson, H. 1991. Timber harvest trafficking and soil compaction in western Montana. Soil Science Society of America Journal. 55(5): 1416-1421. doi:10.2136/sssaj1991.03615995005500050036x

Cumming, S.G. 2001. Forest type and wildfire in the Alberta boreal mixedwood: What do fires burn? Ecological Applications. 11(1): 97-110. doi:10.1890/1051-0761(2001)011[0097:FTAWIT]2.0.CO;2

Cumming, G.S. 2011. Spatial resilience: integrating landscape ecology, resilience, and sustainability. Landscape Ecology. 26(7): 899-909. doi:10.1007/s10980-011-9623-1

Cumming, S.G.; Burton, P.J.; Mapili, M.; Prahacs, S. 1994. Potential conflicts between timber supply and habitat protection in the boreal mixedwood of Alberta, Canada: a simulation study. Forest Ecology and Management. 68(2-3): 281-302. doi:10.1016/0378-1127(94)90052-3

Cumming, G.S.; Olsson, P.; Chapin, F.S., III; Holling, C.S. 2013. Resilience, experimentation, and scale mismatches in social-ecological landscapes. Landscape Ecology. 28(6): 1139-1150. doi:10.1007/s10980-012-9725-4

Cundill, G.; Rodela, R. 2012. A review of assertions about the processes and outcomes of social learning in natural resource management. Journal of Environmental Management. 113: 7-14. doi:10.1016/j.jenvman.2012.08.021

Curran, M.; Green, P.; Maynard, D. 2007. Volcanic ash soils: sustainable soil management practices, with examples of harvest effects and root disease trends. In: Page-Dumroese, D.S.; Miller, R.E.; Mital, J.; McDaniel, P.; Miller, D., tech. eds. Volcanic-ash-derived forest soils of the inland northwest: properties and implications for management and restoration. Proceed. RMRS-P-44. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station: 97-119. https://www.fs.usda.gov/treesearch/pubs/26202

Cushman, S.A.; McKelvey, K.S.; Flather, C.H.; McGarigal, K. 2008. Do forest community types provide a sufficient basis to evaluate biological diversity? Frontiers in Ecology and the Environment. 6(1): 13-17. doi:10.1890/070039

Dahl, H.A.; Nicholson, L.A. 1970. Management decisions relating to off-site plantations. In: Hermann, R.K., comp. Regeneration of ponderosa pine. Paper 681. Corvallis, OR: Oregon State University, School of Forestry: 26-27.

Dahlgren, R.A.; Saigusa, M.; Ugolini, F.C. 2004. The nature, properties and management of volcanic soils. Advances in Agronomy. 82: 113-182. doi:10.1016/S0065-2113(03)82003-5

Daigneault, A.; Sohngen, B.; Sedjo, R. 2012. Economic approach to assess the forest carbon implications of biomass energy. Environmental Science & Technology. 46(11): 5664-5671. doi:10.1021/es2030142

Daily, G.C. 1999. Developing a scientific basis for managing earth's life support systems. Conservation Ecology. 3(2): article 14. http://www.consecol.org/vol3/iss2/art14/

Daily, G.C.; Alexander, S.; Ehrlich, P.R.; Goulder, L.; Lubchenco, J.; Matson, P.A.; Mooney, H.A.; Postel, S.; Schneider, S.H.; Tilman, D.; Woodwell, G.M. 1997. Ecosystem services: benefits supplied to human societies by natural ecosystems. Issues in Ecology. 2: 1-16. http://www.esa.org/science_resources/issues/FileEnglish/issue2.pdf

Dale, L. 2006. Wildfire policy and fire use on public lands in the United States. Society and Natural Resources. 19(3): 275-284. doi:10.1080/08941920500460898

Dale, V.H.; Lugo, A.E.; MacMahon, J.A.; Pickett, S.T.A. 1998. Ecosystem management in the context of large, infrequent disturbances. Ecosystems. 1(6): 546-557. doi:10.1007/s100219900050

Dale, V.H.; Brown, S.; Haeuber, R.A.; Hobbs, N.T.; Huntly, N.; Naiman, R.J.; Riebsame, W.E.; Turner, M.G.; Valone, T.J. 2000a. Ecological principles and guidelines for managing the use of land. Ecological Applications. 10(3): 639-670. doi:10.1890/1051-0761(2000)010[0639:EPAGFM]2.0.CO;2

Dale, V.H.; Joyce, L.A.; McNulty, S.; Neilson, R.P. 2000b. The interplay between climate change, forests, and disturbances. Science of the Total Environment. 262(3): 201-204. doi:10.1016/S0048-9697(00)00522-2

Dale, V.H.; Joyce, L.A.; McNulty, S.; Neilson, R.P.; Ayres, M.P.; Flannigan, M.D.; Hanson, P.J.; Irland, L.C.; Lugo, A.E.; Peterson, C.J.; Simberloff, D.; Swanson, F.J.; Stocks, B.J.; Wotton, B.M. 2001. Climate change and forest disturbances. BioScience. 51(9): 723-734. doi:10.1641/0006-3568(2001)051[0723:CCAFD]2.0.CO;2

Dale, V.H.; Tharp, M.L.; Lannom, K.O.; Hodges, D.G. 2010. Modeling transient response of forests to climate change. Science of the Total Environment. 408(8): 1888-1901. doi:10.1016/j.scitotenv.2009.11.050

Dale, V.; Efroymson, R.; Kline, K. 2011. The land use–climate change–energy nexus. Landscape Ecology. 26(6): 755-773. doi:10.1007/s10980-011-9606-2

Dalton, M.M.; Mote, P.W.; Snover, A.K. 2013. Climate change in the Northwest: Implications for our landscapes, waters, and communities. Washington, DC: Island Press. 230 p. https://islandpress.org/books/climate-change-northwest

Daniel, T.W.; Schmidt, J. 1972. Lethal and nonlethal effects of the organic horizons of forested soils on the germination of seeds from several associated conifer species of the Rocky Mountains. Canadian Journal of Forest Research. 2(3): 179-184. doi:10.1139/x72-031

Daniel, T.W.; Helms, J.A.; Baker, F.S. 1979. Principles of silviculture. 2nd edition. New York: McGraw-Hill Book Company. 500 p. isbn:0-07-015297-7

Daniel, T.C.; Carroll, M.S.; Moseley, C.; Raish, C. 2007. People, fire, and forests: a synthesis of wildfire social science. Corvallis, OR: Oregon State University Press. 226 p. isbn:978-0-87071-184-8

Danielson, J.; Sampson, N., chair. [1995.] Proceedings of the conference: forest health and fire danger in inland western forests. [Place of publication unknown]: [Publisher unknown]. 235 p.

D'Arrigo, R.; Villalba, R.; Wiles, G. 2001. Tree-ring estimates of Pacific decadal climate variability. Climate Dynamics. 18(3): 219-224. doi:10.1007/s003820100177

D'Arrigo, R.; Cook, E.R.; Wilson, R.J.; Allan, R.; Mann, M.E. 2005. On the variability of ENSO over the past six centuries. Geophysical Research Letters. 32(3): L03711. doi:10.1029/2004gl022055

Das, A.; Battles, J.; van Mantgem, P.J.; Stephenson, N.L. 2008. Spatial elements of mortality risk in old-growth forests. Ecology. 89(6): 1744-1756. doi:10.1890/07-0524.1

Das, A.; Battles, J.; Stephenson, N.L.; van Mantgem, P.J. 2011. The contribution of competition to tree mortality in old-growth coniferous forests. Forest Ecology and Management. 261(7): 1203-1213. doi:10.1016/j.foreco.2010.12.035

Daubenmire, R.F. 1943. Soil temperature versus drought as a factor determining lower altitudinal limits of trees in the Rocky Mountains. Botanical Gazette. 105(1): 1-13. https://www.jstor.org/stable/2472085

Daubenmire, R.F. 1943. Vegetational zonation in the Rocky Mountains. The Botanical Review. 9(6): 325-393. doi:10.1007/BF02872481

Daubenmire, R.F. 1946. Radial growth of trees at different altitudes. Botanical Gazette. 107(4): 462-467. doi:10.1086/335371

Daubenmire, R. 1952. Forest vegetation of northern Idaho and adjacent Washington, and its bearing on concepts of vegetation classification. Ecological Monographs. 22(4): 301-330. doi:10.2307/1948472

Daubenmire, R. 1953a. Classification of the conifer forests of eastern Washington and northern Idaho. Northwest Science. 27(1): 17-24.

Daubenmire, R. 1953b. Nutrient content of leaf litter of trees in the northern Rocky Mountains. Ecology. 34(4): 786-793. doi:10.2307/1931341

Daubenmire, R. 1956. Climate as a determinant of vegetation distribution in eastern Washington and northern Idaho. Ecological Monographs. 26(2): 131-154. doi:10.2307/1943287

Daubenmire, R. 1957. Influence of temperature upon soil moisture constants and its possible ecologic significance. Ecology. 38(2): 320-324. doi:10.2307/1931692

Daubenmire, R. 1961. Vegetative indicators of rate of height growth in ponderosa pine. Forest Science. 7(1): 24-34. doi:10.1093/forestscience/7.1.24

Daubenmire, R. 1968. Soil moisture in relation to vegetation distribution in the mountains of northern Idaho. Ecology. 49(3): 431-438. doi:10.2307/1934109

Daubenmire, R. 1973. A comparison of approaches to the mapping of forest land for intensive management. Forestry Chronicle. 49(2): 87-91. doi:10.5558/tfc49087-2

Daubenmire, R. 1980. Mountain topography and vegetation patterns. Northwest Science. 54(2): 146-152. https://www.fs.fed.us/rm/pubs/rmrs_gtr292/1980_daubenmire.pdf

Daubenmire, R.; Daubenmire, J.B. 1968. Forest vegetation of eastern Washington and northern Idaho. Tech. Bull. 60. Pullman, WA: Washington State University, College of Agriculture and Home Economics, Agricultural Research Center. 104 p. https://www.fs.usda.gov/treesearch/pubs/50665

Daubenmire, R.; Prusso, D.C. 1963. Studies of the decomposition rates of tree litter. Ecology. 44(3): 589-592. doi:10.2307/1932544

Davies, K.W.; Svejcar, T.J.; Bates, J.D. 2009. Interaction of historical and nonhistorical disturbances maintains native plant communities. Ecological Applications. 19(6): 1536-1545. doi:10.1890/09-0111.1

Davis, L.S.; Johnson, K.N.; Bettinger, P.S.; Howard, T.E. 2001. Forest management: to sustain ecological, economic, and social values. 4th edition. New York: McGraw-Hill. 804 p. isbn:0-07-032694-0

Davis, K.T.; Higuera, P.E.; Sala, A. 2018. Anticipating fire-mediated impacts of climate change using a demographic framework. Functional Ecology. 32(7): 1729-1745. doi:10.1111/1365-2435.13132

Davis, K.T.; Dobrowski, S.Z.; Higuera, P.E.; Holden, Z.A.; Veblen, T.T.; Rother, M.T.; Parks, S.A.; Sala, A.; Maneta, M.P. 2019. Wildfires and climate change push low-elevation forests across a critical climate threshold for tree regeneration. Proceedings of the National Academy of Sciences. 116(13): 6193-6198. doi:10.1073/pnas.1815107116

Day, T.A.; DeLucia, E.H.; Smith, W.K. 1990. Effect of soil temperature on stem sap flow, shoot gas exchange and water potential of *Picea engelmannii* (Parry) during snowmelt. Oecologia. 84(4): 474-481. www.jstor.org/stable/4219453

Day, K.; Berg, J.; Brown, H.; Crow, T.; Morrison, J.; Nowacki, G.; Puckett, D.; Sallee, R.; Schenck, T.; Wood B. 2006. Ecosystem restoration: a framework for restoring and maintaining the national forests and grasslands. Washington, DC: USDA Forest Service. 24 p. https://www.fs.fed.us/restoration/documents/RestFramework_final_010606.pdf

DeBell, D. [1979.] Future potential for use of symbiotic nitrogen fixation in forest management. In: Gordon, J.C.; Wheeler, C.T.; Perry, D.A., eds. Symbiotic nitrogen fixation in the management of temperate forests; proceedings of a workshop held April 2-5, 1979. Corvallis, OR: Oregon State University, Forest Research Laboratory: 451-466.

DeBell, D.S.; Curtis, R.O.; Harrington, C.A.; Tappeiner, J.C. 1997. Shaping stand development through silvicultural practices. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press: 141-149. isbn:1-55963-399-9

DeByle, N.V.; Winokur, R.P., eds. 1985. Aspen: ecology and management in the western United States. Gen. Tech. Rep. RM-119. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 283 p. http://www.treesearch.fs.fed.us/pubs/24942

DeFries, R.S.; Foley, J.A.; Asner, G.P. 2004. Land-use choices: balancing human needs and ecosystem function. Frontiers in Ecology and the Environment. 2(5): 249-257. doi:10.1890/1540-9295(2004)002[0249:LCBHNA]2.0.CO;2

DeGraaf, R.M. 1991. Breeding bird assemblages in managed northern hardwood forests in New England. In: Rodiek, J.E.; Bolen, E.G., eds. Wildlife and habitats in managed landscapes. Washington, DC: Island Press: 153-171.

DeGraaf, R.M.; Healy, W.M. 1993. The myth of nature's constancy–preservation, protection and ecosystem management. In: Transactions of the 58th North American wildlife and natural resources conference. Washington, DC: Wildlife Management Institute: 17-28.

Degrassi, A.L.; Brantley, S.; Levine, C.R.; Mohan, J.; Record, S.; Tomback, D.F.; Ellison, A.M. 2019. Loss of foundation species revisited: conceptual framework with lessons learned from eastern hemlock and whitebark pine. Ecosphere. 10(11): e02917 (11 p).

doi:10.1002/ecs2.2917

de Groot, M.; Diaci, J.; Ogris, N. 2019. Forest management history is an important factor in bark beetle outbreaks: Lessons for the future. Forest Ecology and Management. 433: 467-474. doi:10.1016/j.foreco.2018.11.025

del Moral, R. 1972. Diversity patterns in forest vegetation of the Wenatchee Mountains, Washington. Bulletin of the Torrey Botanical Club. 99(2): 57-64. doi:10.2307/2484199

Dello, K.; Mote, P., eds. 2010. Oregon climate assessment report. Corvallis, OR: Oregon State University, College of Oceanic and Atmospheric Sciences. 417 p. http://occri.net/ocar

DeLong, S.C. 1998. Natural disturbance rate and patch size distribution of forests in northern British Columbia: implications for forest management. Northwest Science. 72(Special Issue): 35-48.

DeLong, S.C. 2007. Implementation of natural disturbance-based management in northern British Columbia. Forestry Chronicle. 83(3): 338-346. doi:10.5558/tfc83338-3

DeLong, S.C. 2011. Land units and benchmarks for developing natural disturbance-based forest management guidance for northeastern British Columbia. Tech. Rep. 059. Victoria, BC: British Columbia Ministry of Forests and Range, Forest Science Program. 42 p. http://www.for.gov.bc.ca/hfd/pubs/Docs/Tr/Tr059.htm

DeLong, S.C.; Kessler, W.B. 2000. Ecological characteristics of mature forest remnants left by wildfire. Forest Ecology and Management. 131(1-3): 93-106. doi:10.1016/S0378-1127(99)00203-0

DeLong, S.C.; Tanner, D. 1996. Managing the pattern of forest harvest: lessons from wildfire. Biodiversity and Conservation. 5(10): 1191-1205. doi:10.1007/BF00051571

DeLuca, T.H.; Aplet, G.H. 2008. Charcoal and carbon storage in forest soils of the Rocky Mountain west. Frontiers in Ecology and the Environment. 6(1): 18-24. doi:10.1890/070070

DeMeo, T. 2011. Moving towards a new social contract for managing federal lands. Northwest Science. 85(3): 502-503. doi:10.3955/046.085.0310

Denevan, W.M. 1992. The pristine myth: the landscape of the Americas in 1492. Annals of the Association of American Geographers. 82(3): 369-385. doi:10.1111/j.1467-8306.1992.tb01965.x

D'Eon, R.G.; Glenn, S.M. 2000. Perceptions of landscape patterns: Do the numbers count? Forestry Chronicle. 76(3): 475-480. doi:10.5558/tfc76475-3

D'Eon, R.G.; Glenn, S.M. 2005. The influence of forest harvesting on landscape spatial patterns and old-growth-forest fragmentation in southeast British Columbia. Landscape Ecology. 20(1): 19-33. doi:10.1007/s10980-004-0286-z

Despain, D.G.; Weaver, T.; Aspinall, R.J. 2001. A rule-based model for mapping potential exotic plant distribution. Western North American Naturalist. 61(4): 428-433. https://scholarsarchive.byu.edu/wnan/vol61/iss4/6

Desprez-Loustau, M.-L.; Marçais, B.; Nageleisen, L.-M.; Piou, D.; Vannini, A. 2006. Interactive effects of drought and pathogens in forest trees. Annals of Forest Science. 63(6): 597-612. doi:10.1051/forest:2006040

DeStefano, S.; Haight, R.G., eds. 2002. Forest wildlife-habitat relationships: population and community responses to forest management. Bethesda, MD: Society of American Foresters. 275 p. isbn:0-939970-84-8

Devine, W.; Aubry, C.; Bower, A.; Miller, J.; Ahr, N.M. 2012. Climate change and forest trees in the Pacific Northwest: A vulnerability assessment and recommended actions for national forests. Olympia, WA: USDA Forest Service, Pacific Northwest Region. 102 p (plus 8 appendixes). http://ecoshare.info/projects/ccft/

DeWalt, B.R. 1994. Using indigenous knowledge to improve agriculture and natural resource management. Human Organization. 53(2): 123-131. doi:10.17730/humo.53.2.ku60563817m03n73

Diaci, J.; Kerr, G.; O'hara, K. 2011. Twenty-first century forestry: integrating ecologically based, uneven-aged silviculture with increased demands on forests. Forestry. 84(5): 463-465. doi:10.1093/forestry/cpr053

Diamond, J. 1992. Must we shoot deer to save nature? Natural History. 101(8): 4 p.

Diamond, J. 1999. Guns, germs and steel: the fates of human societies. New York: W.W. Norton & Company. 480 p.

Dickinson, J.L.; Shirk, J.; Bonter, D.; Bonney, R.; Crain, R.L.; Martin, J.; Phillips, T.; Purcell, K. 2012. The current state of citizen science as a tool for ecological research and public engagement. Frontiers in Ecology and the Environment. 10(6): 291-297. doi:10.1890/110236

Dillon, G.K.; Holden, Z.A.; Morgan, P.; Crimmins, M.A.; Heyerdahl, E.K.; Luce, C.H. 2011. Both topography and climate affected forest and woodland burn severity in two regions of the western US, 1984 to 2006. Ecosphere. 2(12): art130 (33 p). doi:10.1890/es11-00271.1

Dinoor, A.; Eshed, N. 1984. The role and importance of pathogens in natural plant communities. Annual Review of Phytopathology. 22: 443-466. doi:10.1146/annurev.py.22.090184.002303

Dixon, G.E. 2015. Essential FVS: a user's guide to the Forest Vegetation Simulator. Fort Collins, CO: USDA Forest Service, Washington Office, Forest Management Service Center. 226 p. https://www.fs.fed.us/fmsc/ftp/fvs/docs/gtr/EssentialFVS.pdf

Dobor, L.; Hlásny, T.; Rammer, W.; Barka, I.; Trombik, J.; Pavlenda, P.; Šebeň, V.; Štěpánek, P.; Seidl, R. 2018. Post-disturbance recovery of forest carbon in a temperate forest landscape under climate change. Agricultural and Forest Meteorology. 263: 308-322. doi:10.1016/j.agrformet.2018.08.028

Dobson, A.; Crawley, M. 1994. Pathogens and the structure of plant communities. Trends in Ecology and Evolution. 9(10): 393-398. doi:10.1016/0169-5347(94)90062-0

Dodson, S.I.; Allen, T.F.H.; Carpenter, S.R.; Ives, A.R.; Jeanne, R.L.; Kitchell, J.F.; Langston, N.E.; Turner, M.G. 1998. Ecology. New York: Oxford University Press. 433 p. isbn:0-19-512079-5

Doliner, L.H.; Borden, J.H. 1984. Pesterms: a glossary of forest pest management terms. Pest Manage. Rep. Num. 3. Victoria, BC: British Columbia Ministry of Forests. 34 p.

Dolph, R.E., Jr. 1980. Budworm activity in Oregon and Washington, 1947-1979. Pub. R6-FIDM-033-1980. Portland, OR: USDA Forest Service, Pacific Northwest Region, State and Private Forestry, Forest Pest Management. 54 p.

Donato, D.C.; Fontaine, J.B.; Campbell, J.L.; Robinson, W.D.; Kauffman, J.B.; Law, B.E. 2006. Post-wildfire logging hinders regeneration and increases fire risk. Science. 311(5759): 352. doi:10.1126/science.1122855

Donato, D.C.; Fontaine, J.B.; Campbell, J.L.; Robinson, W.D.; Kauffman, J.B.; Law, B.E. 2009. Conifer regeneration in stand-replacement portions of a large mixed-severity wildfire in the Klamath-Siskiyou Mountains. Canadian Journal of Forest Research. 39(4): 823-838. doi:10.1139/X09-016

Donato, D.C.; Harvey, B.J.; Turner, M.G. 2016. Regeneration of montane forests 24 years after the 1988 Yellowstone fires: A fire-catalyzed shift in lower treelines? Ecosphere. 7(8): e01410 (16 p). doi:10.1002/ecs2.1410

Donner, B.L.; Running, S.W. 1986. Water stress response after thinning *Pinus contorta* stands in Montana. Forest Science. 32(3): 614-625. doi:10.1093/forestscience/32.3.614

Donovan, G.H.; Brown, T.C. 2005. An alternative incentive structure for wildfire management on national forest land. Forest Science. 51(5): 387-395.

doi:10.1093/forestscience/51.5.387

Donovan, G.H.; Brown, T.C. 2008. Estimating the avoided fuel-treatment costs of wildfire. Western Journal of Applied Forestry. 23(4): 197-201. doi:10.1093/wjaf/23.4.197

Dordel, J.; Feller, M.C.; Simard, S.W. 2008. Effects of mountain pine beetle (*Dendroctonus ponderosae* Hopkins) infestations on forest stand structure in the southern Canadian Rocky Mountains. Forest Ecology and Management. 255(10): 3563-3570. doi:10.1016/j.foreco.2008.02.018

Downing, W.M.; Krawchuk, M.A.; Meigs, G.W.; Haire, S.L.; Coop, J.D.; Walker, R.B.; Whitman, E.; Chong, G.; Miller, C. 2019. Influence of fire refugia spatial pattern on post-fire forest recovery in Oregon's Blue Mountains. Landscape Ecology. 34(4): 771-792. doi:10.1007/s10980-019-00802-1

Drever, M.C.; Martin, K. 2010. Response of woodpeckers to changes in forest health and harvest: implications for conservation of avian biodiversity. Forest Ecology and Management. 259(5): 958-966. doi:10.1016/j.foreco.2009.11.038

Drever, C.R.; Peterson, G.; Messier, C.; Bergeron, Y.; Flannigan, M. 2006. Can forest management based on natural disturbances maintain ecological resilience? Canadian Journal of Forest Research. 36(9): 2285-2299. doi:10.1139/x06-132

Drever, M.C.; Aitken, K.E.H.; Norris, A.R.; Martin, K. 2008. Woodpeckers as reliable indicators of bird richness, forest health and harvest. Biological Conservation. 141(3): 624-634. doi:10.1016/j.biocon.2007.12.004

Droske, C.A. 2012. Conifer seedling density and growth following post-fire salvage logging and grass seeding on the 2005 School Fire, Washington, USA. M.S. thesis. Moscow, ID: University of Idaho, Department of Forest Resources. 37 p.

Drury, W.H., Jr. 1998. Chance and change: ecology for conservationists. Berkeley, CA: University of California Press. 223 p. isbn:0-520-21155-3

Drury, W.H.; Nisbet, I.C.T. 1973. Succession. Journal of the Arnold Arboretum. 54(3): 331-368. https://www.biodiversitylibrary.org/itempdf/41242

Dudley, N. 1996. Authenticity as a means of measuring forest quality. Biodiversity Letters. 3(1): 6-9. doi:10.2307/2999703

Duff, T.J.; Cawson, J.G.; Penman, T.D. 2019. Determining burnability: Predicting completion rates and coverage of prescribed burns for fuel management. Forest Ecology and Management. 433: 431-440. doi:10.1016/j.foreco.2018.11.009

Dugan, A.J.; Birdsey, R.; Mascorro, V.S.; Magnan, M.; Smyth, C.E.; Olguin, M.; Kurz, W.A. 2018. A systems approach to assess climate change mitigation options in landscapes of the United States forest sector. Carbon Balance and Management. 13(1): 13. doi:10.1186/s13021-018-0100-x

Duguid, M.C.; Ashton, M.S. 2013. A meta-analysis of the effect of forest management for timber on understory plant species diversity in temperate forests. Forest Ecology and Management. 303: 81-90. doi:10.1016/j.foreco.2013.04.009

Dunster, J.; Dunster, K. 1996. Dictionary of natural resource management. Vancouver, BC: UBC Press. 363 p. isbn:0-7748-0503-X

Durbin, K. 1992. Regenerating ravaged forests. Portland, OR: Portland Oregonian. November 30; Section B: B1, B8.

Dwire, K.A.; Kauffman, J.B. 2003. Fire and riparian ecosystems in landscapes of the western USA. Forest Ecology and Management. 178(1-2): 61-74. doi:10.1016/S0378-1127(03)00053-7

East Oregonian. 1992. Forests in peril, a five-part series about forest health in the Blue Mountains. Pendleton, OR: East Oregonian. April 20-24.

Eaton, C.B.; Beal, J.A.; Furniss, R.L.; Speers, C.F. 1949. Airplane and helicopter spraying with DDT for spruce budworm control. Journal of Forestry. 47(10): 823-827. doi:10.1093/jof/47.10.823

Eberhardt, K.E.; Woodard, P.M. 1987. Distribution of residual vegetation associated with large fires in Alberta. Canadian Journal of Forest Research. 17(10): 1207-1212. doi:10.1139/87-186

Eckert, J.R. 1975. Roles of fire and allelopathy in ponderosa pine forests of the Colorado Front Range. Journal of Colorado-Wyoming Academy of Science. 7: 31.

Edmonds, R.L.; Agee, J.K.; Gara, R.I. 2005. Forest health and protection. Long Grove, IL: Waveland Press, Inc. 630 p. isbn:1-57766-396-9

Egan, D.; Howell, E.A. 2001. The historical ecology handbook: a restorationist's guide to reference ecosystems. Washington, DC: Island Press. 457 p. isbn:1-55963-746-3

Egan, J.M.; Jacobi, W.R.; Negron, J.F.; Smith, S.L.; Cluck, D.R. 2010. Forest thinning and subsequent bark beetle-caused mortality in northeastern California. Forest Ecology and Management. 260(10): 1832-1842. doi:10.1016/j.foreco.2010.08.030

Egerton, F.N. 1973. Changing concepts of the balance of nature. Quarterly Review of Biology. 48(2): 322-350. doi:10.2307/2820544

Ehrlich, P.R.; Daily, G.C. 1993. Science and the management of natural resources. Ecological Applications. 3(4): 558-560. doi:10.2307/1942078

Elkin, C.; Reineking, B.; Bigler, C.; Bugmann, H. 2012. Do small-grain processes matter for landscape scale questions? Sensitivity of a forest landscape model to the formulation of tree growth rate. Landscape Ecology. 27(5): 697-711. doi:10.1007/s10980-012-9718-3

Ellefson, P.V. 1974. Douglas-fir tussock moth infestation: a challenge to forestry professionals. Journal of Forestry. 72(6): 326-327, 379. doi:10.1093/jof/72.6.326

Ellison, A.M.; Bank, M.S.; Clinton, B.D.; Colburn, E.A.; Elliott, K.; Ford, C.R.; Foster, D.R.; Kloeppel, B.D.; Knoepp, J.D.; Lovett, G.M.; Mohan, J.; Orwig, D.A.; Rodenhouse, N.L.; Sobczak, W.V.; Stinson, K.A.; Stone, J.K.; Swan, C.M.; Thompson, J.; Von Holle, B.; Webster, J.R. 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment. 3(9): 479-486. doi:10.1890/1540-9295(2005)003[0479:LOFSCF]2.0.CO;2

Elzinga, C.L.; Salzer, D.W.; Willoughby, J.W. 1998. Measuring and monitoring plant populations. BLM Tech. Ref. 1730-1; BLM/RS/ST-98/005+1730. Denver, CO: U.S. Department of the Interior, Bureau of Land Management, National Applied Resource Sciences Center. 477 p. http://digitalcommons.unl.edu/cgi/viewcontent.cgi?article=1016&context=usblmpub

Emmingham, W.H.; Oester, P.T.; Fitzgerald, S.A.; Filip, G.M.; Edge, W.D. 2005. Ecology and management of eastern Oregon forests: a comprehensive manual for forest managers. Manual 12. Corvallis, OR: Oregon State University, Extension Service. 208 p. isbn:1-931979-09-X

Endress, B.A.; Wisdom, M.J.; Vavra, M.; Parks, C.G.; Dick, B.L.; Naylor, B.J.; Boyd, J.M. 2012. Effects of ungulate herbivory on aspen, cottonwood, and willow development under forest fuels treatment regimes. Forest Ecology and Management. 276: 33-40. doi:10.1016/j.foreco.2012.03.019

Eng, M. 1998. Spatial patterns in forested landscapes: implications for biology and forestry. In: Voller, J.; Harrison, S., eds. Conservation biology principles for forested landscapes. Vancouver, BC: UBC Press: 42-75. isbn:0-7748-0630-3

Ennos, A.R. 1997. Wind as an ecological factor. Trends in Ecology and Evolution. 12(3): 108-111. doi:10.1016/S0169-5347(96)10066-5

Enright, N.J.; Fontaine, J.B.; Lamont, B.B.; Miller, B.P.; Westcott, V.C. 2014. Resistance and resilience to changing climate and fire regime depend on plant functional traits. Journal of Ecology. 102(6): 1572-1581. doi:10.1111/1365-2745.12306

Enright, N.J.; Fontaine, J.B.; Bowman, D.M.; Bradstock, R.A.; Williams, R.J. 2015. Interval squeeze: altered fire regimes and demographic responses interact to threaten woody species persistence as climate changes. Frontiers in Ecology and the Environment. 13(5): 265-272. doi:10.1890/140231

Entry, J.A.; Cromack, K., Jr.; Kelsey, R.G.; Martin, N.E. 1991. Response of Douglas-fir to infection by *Armillaria ostoyae* after thinning or thinning plus fertilization. Phytopathology. 81(6): 682-689. doi:10.1094/Phyto-81-682

Entry, J.A.; Martin, N.E.; Kelsey, R.G.; Cromack, K., Jr. 1992. Chemical constituents in root bark of five species of western conifer saplings and infection by *Armillaria ostoyae*. Phytopathology. 82(4): 393-397. doi:10.1094/Phyto-82-393

Erelli, M.C.; Ayres, M.P.; Eaton, G.K. 1998. Altitudinal patterns in host suitability for forest insects. Oecologia. 117(1-2): 133-142. doi:10.1007/s004420050641

Erickson, J.R.; Toweill, D.E. 1994. Forest health and wildlife habitat management on the Boise National Forest, Idaho. Journal of Sustainable Forestry. 2(3-4): 389-409. doi:10.1300/J091v02n03_10

Estes, B.L.; Knapp, E.E.; Skinner, C.N.; Uzoh, F.C.C. 2012. Seasonal variation in surface fuel moisture between unthinned and thinned mixed conifer forest, northern California, USA. International Journal of Wildland Fire. 21(4): 428-435. doi:10.1071/WF11056

Evans, A.M.; Everett, R.G.; Stephens, S.L.; Youtz, J.A. 2011. Comprehensive fuels treatment practices guide for mixed conifer forests: California, central and southern Rockies, and the southwest. Santa Fe, NM: Forest Guild. 106 p. http://www.forestguild.org/mixed_conifer.html

Evenden, J.C.; Gibson, A.L. 1940. A destructive infestation in lodgepole pine stands by the mountain pine beetle. Journal of Forestry. 38(3): 271-275. doi:10.1093/jof/38.3.271

Everett, R.L., comp. 1994. Volume 4: restoration of stressed sites, and processes. Gen. Tech. Rep. PNW-GTR-330. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 123 p. https://www.fs.usda.gov/treesearch/pubs/6939

Everett, R.L.; Lehmkuhl, J.F. 1996. An emphasis-use approach to conserving biodiversity. Wildlife Society Bulletin. 24(2): 192-199. https://www.jstor.org/stable/3783107

Everett, R.L.; Lehmkuhl, J.F. 1999. Restoring biodiversity on public forest lands through disturbance and patch management irrespective of land-use allocation. In: Baydack, R.K.; Campa, H. III; Haufler, J.B., eds. Practical approaches to the conservation of biological diversity. Washington, DC: Island Press: 87-105. isbn:1-55963-544-4

Everett, R.L.; Hessburg, P.F.; Lillybrdge, T.R. 1994. Emphasis areas as an alternative to buffer zones and reserved areas in the conservation of biodiversity and ecosystem processes. Journal of Sustainable Forestry. 2(3/4): 283-292. doi:10.1300/J091v02n03_04

Everett, R.L.; Camp, A.E.; Schellhaas, R. 1996. Building a new forest with fire protection in mind. In: Proceedings of the Society of American Foresters National Convention. Bethesda, MD: Society of American Foresters: 253-258.

Everett, R.; Townsley, J.; Baumgartner, D. 2000. Chapter 11. Inherent disturbance regimes. Journal of Sustainable Forestry. 11(1-2): 265-288. doi:10.1300/J091v11n01_11

Everett, R.; Schellhaas, R.; Ohlson, P.; Spurbeck, D.; Keenum, D. 2003. Continuity in fire disturbance between riparian and adjacent sideslope Douglas-fir forests. Forest Ecology and Management. 175(1-3): 31-47. doi:10.1016/S0378-1127(02)00120-2

Everett, R.L.; Baumgartner, D.; Ohlson, P.; Schellhaas, R. 2008. Structural classes and age structure in 1860 and 1940 reconstructed fir-pine stands of eastern Washington. Western North American Naturalist. 68(3): 278-290. doi:10.3398/1527-0904(2008)68[278:SCAASI]2.0.CO;2

Everham, E.M., III; Brokaw, N.V.L. 1996. Forest damage and recovery from catastrophic wind. Botanical Review. 62(2): 113-185. doi:10.1007/BF02857920

Evers, L. 2002. Fire regimes of Oregon and Washington. Unpub. Rep. Portland, OR: USDA Forest Service, Pacific Northwest Region. 3 p.

Evers, C.R.; Ager, A.A.; Nielsen-Pincus, M.; Palaiologou, P.; Bunzel, K. 2019. Archetypes of community wildfire exposure from national forests of the western US. Landscape and Urban Planning. 182: 55-66. doi:10.1016/j.landurbplan.2018.10.004

Ewel, J.J. 1999. Natural systems as models for the design of sustainable systems of land use. Agroforestry Systems. 45(1): 1-21. doi:10.1023/a:1006219721151

Ewel, K.C. 2001. Natural resource management: the need for interdisciplinary collaboration. Ecosystems. 4(8): 716-722. doi:10.1007/s10021-001-0040-1

Eycott, A.; Watts, K.; Moseley, D.; Ray, D. 2007. Evaluating biodiversity in fragmented landscapes: the use of focal species. FC Infor. Note 89. Edinburgh, UK: Forestry Commission. 8 p.

Eyre, F.H., ed. 1980. Forest cover types of the United States and Canada. Washington, DC: Society of American Foresters. 148 p. isbn:978-0686306979

Fagan, B. 2002. The Little Ice Age: how climate made history, 1300-1850. New York: Basic Books. 246 p. isbn:0-465-02272-3

Fahey, T.J. 1983. Nutrient dynamics of aboveground detritus in lodgepole pine (*Pinus contorta* ssp. *latifolia*) ecosystems, southeastern Wyoming. Ecological Monographs. 53(1): 51-72. doi:10.2307/1942587

Fahey, T.J.; Knight, D.H. 1986. Lodgepole pine ecosystems. BioScience. 36(9): 610-617. doi:10.2307/1310196

Fahey, T.J; Woodbury, P.B; Battles, J.J; Goodale, C.L; Hamburg, S.; Ollinger, S.; Woodall, C.W. 2010. Forest carbon storage: ecology, management, and policy. Frontiers in Ecology and the Environment. 8(5): 245-252. doi:10.1890/080169

Fahnestock, G.R. 1976. Fires, fuels, and flora as factors in wilderness management; the Pasayten case. In: Proceedings of the annual Tall Timbers fire ecology conference no. 15. Tallahassee, FL: Tall Timbers Research Station: 33-69. https://talltimbers.org/wp-content/uploads/2018/09/33-Fahnestock1974_op.pdf

Falk, D. 1990. Discovering the future, creating the past: some reflections on restoration. Restoration and Management Notes. 8(2): 71-72. http://er.uwpress.org/content/8/2/71.full.pdf

Falk, D.A.; Palmer, M.A.; Zedler, J.B. 2006. Foundations of restoration ecology. Washington, DC: Island Press. 364 p. isbn:978-1-59726-017-6

Farris, C.A.; Neuenschwander, L.F.; Boudreau, S.L. 1998. Monitoring initial plant succession following fire in a subalpine spruce-fir forest. In: Pruden, T.L.; Brennan, L.A., eds. Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology Conference Proceedings No. 20. Tallahassee, FL: Tall Timbers Research Station: 298-305. https://talltimbers.org/wp-content/uploads/2018/09/298-Farrisetal1998_op.pdf

Fauss, D.L.; Pierce, W.R. 1969. Stand conditions and spruce budworm damage in a western Montana forest. Journal of Forestry. 67(5): 322-325. doi:10.1093/jof/67.5.322

Fayt, P.; Machmer, M.M.; Steeger, C. 2005. Regulation of spruce bark beetles by woodpeckers–a literature review. Forest Ecology and Management. 206(1-3): 1-14. doi:10.1016/j.foreco.2004.10.054

Felix, A.B.; Campa, H.; Millenbah, K.F.; Winterstein, S.R.; Moritz, W.E. 2004. Development of landscape-scale habitat-potential models for forest wildlife planning and management. Wildlife Society Bulletin. 32(3): 795-806. doi:10.2193/0091-7648(2004)032[0795:dolhmf]2.0.co;2

Feller, M.C. 1982. The ecological effects of slashburning with particular reference to British Columbia: a literature review. Land Management Report Number 13. Victoria, BC: Information Services Branch, British Columbia, Ministry of Forests. 60 p. https://www.for.gov.bc.ca/hfd/pubs/Docs/Mr/Lmr/Lmr013.pdf

Feller, M.C. 1988. Relationships between fuel properties and slashburning-induced nutrient losses. Forest Science. 34(4): 998-1015. doi:10.1093/forestscience/34.4.998

Feller, M.C. 1998. Influence of ecological conditions on Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) germinant survival and initial seedling growth in south-central British Columbia. Forest Ecology and Management. 107(1-3): 55-69. doi:10.1016/S0378-1127(97)00322-8

Feller, M.C. 2005. Maintaining plant diversity in mixed severity fire regime ecosystems. In: Taylor, L.; Zelnik, J.; Cadwallader, S.; Hughes, B., eds. Mixed severity fire regimes: ecology and management. AFE MISC03. Pullman, WA: Washington State University Extension: 21-32.

Fellin, D.G. 1976. Forest practices, silvicultural prescriptions, and the western spruce budworm. In: USDA Forest Service. Proceedings of a symposium on the spruce budworm. Misc. Pub. 1327. Washington, DC: USDA Forest Service, State and Private Forestry, Forest Insect and Disease Management: 117-121. Proceedings_of_a_Symposium_on_the_Spruce

Fellin, D.G. 1980. A review of some interactions between harvesting, residue management, fire, and forest insects and diseases. In: USDA Forest Service. Environmental consequences of timber harvesting in Rocky Mountain coniferous forests. Gen. Tech. Rep. INT-90. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 335-414. https://archive.org/download/CAT80751102/CAT80751102.pdf

Fellin, D.G. 1983. Chemical insecticide vs the western spruce budworm: after three decades, what's the score? Western Wildlands. 9(1): 8-12.

Fellin, D.G.; Dewey, J.E. 1986. Western spruce budworm. Forest Insect & Disease Leaflet 53. [Place of publication unknown]: USDA Forest Service. 10 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043445.pdf

Fellin, D.G.; Shearer, R.C.; Carlson, C. 1983. Western spruce budworm in the northern Rocky Mountains. Western Wildlands. 9(1): 2-7.

Fellin, D.G.; Schmidt, W.C.; Carlson, C.E. 1984. The western spruce budworm in the northern Rocky Mountains–ecological relations and silvicultural management strategies. In: Baumgartner, D.M.; Mitchell, R., comps. Silvicultural management strategies for pests of the interior Douglas-fir and grand fir forest types. Pullman, WA: Washington State University, Cooperative Extension: 81-94.

Ferguson, D.E. 1984. Needed: guidelines for defining acceptable advance regeneration. Res. Note INT-341. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 5 p. https://www.fs.usda.gov/treesearch/pubs/34888

Ferguson, D.E. 1988. Growth of regeneration defoliated by spruce budworm in Idaho. Res. Pap. INT-393. Ogden, UT: USDA Forest Service, Intermountain Research Station. 13 p. Growth_of_Regeneration_Defoliated_by_Spr

Ferguson, D.E. 1991. Allelopathic potential of western coneflower *(Rudbeckia occidentalis).* Canadian Journal of Botany. 69(12): 2806-2808. doi:10.1139/b91-351

Ferguson, D.E. 1994a. Advance regeneration in the inland West: considerations for individual tree and forest health. Journal of Sustainable Forestry. 2(3-4): 411-422. doi:10.1300/J091v02n03_11

Ferguson, D.E. 1994b. Natural regeneration following timber harvest in interior cedar-hemlock-white pine forests. In: Baumgartner, D.M.; Lotan, J.E.; Tonn, J.R., comps. Interior cedar-hemlock-white pine forests: ecology and management. Pullman, WA: Washington

State University, Department of Natural Resource Sciences: 239-247.

Ferguson, S.A. 1997. A climate-change scenario for the Columbia River basin. Res. Pap. PNW-RP-499. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 9 p. https://www.fs.usda.gov/treesearch/pubs/26876

Ferguson, S.A. 1999a. Climatology of the interior Columbia River basin. Gen. Tech. Rep. PNW-GTR-445. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 31 p. https://www.fs.usda.gov/treesearch/pubs/2994

Ferguson, D.E. 1999b. Effects of pocket gophers, bracken fern, and western coneflower on planted conifers in northern Idaho – an update and two more species. New Forests. 18: 199-217. doi:10.1023/A:1006504700542

Ferguson, S.A. 2001. Climatic variability in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 62-69. http://hdl.handle.net/2376/982

Ferguson, D.E.; Adams, D.L. 1994. Effects of pocket gophers, bracken fern and western coneflower on survival and growth of planted conifers. Northwest Science. 68(4): 241-249. http://hdl.handle.net/2376/1581

Ferguson, D.E.; Boyd, R.J. 1988. Bracken fern inhibition of conifer regeneration in northern Idaho. Res. Pap. INT-388. Ogden, UT: USDA Forest Service, Intermountain Research Station. 11 p. https://archive.org/download/brackenferninhib388ferg/brackenferninhib388ferg.pdf

Ferguson, D.E.; Byrne, J.C. 2000. Environmental characteristics of grand fir mosaic and adjacent habitat types. Res. Pap. RMRS-RP-24. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 20 p. https://www.fs.usda.gov/treesearch/pubs/29121

Ferguson, D.E.; Carlson, C.E. 1991. Natural regeneration of interior Douglas-fir in the northern Rocky Mountains. In: Baumgartner, D.M.; Lotan, J.E., comps. Interior Douglas-fir: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 239-246.

Ferguson, D.E.; Johnson, F.D. 1996. Classification of grand fir mosaic habitats. Gen. Tech. Rep. INT-GTR-337. Ogden, UT: USDA Forest Service, Intermountain Research Station. 16 p. https://archive.org/download/CAT10750562/CAT10750562.pdf

Ferguson, D.E.; Applegate, V.J.; Aune, P.S.; Carlson, C.E.; Geier-Hayes, K.; Graham, R.T.; Jacobsen, G.L.; Jain, T.B.; Powell, D.C.; Shepperd, W.D.; Sloan, J.P.; Youngblood, A. 1998. Communicating the role of silviculture and Forest Service silviculture research in the interior West. In: Murphy, D.; Loftus, N. Communicating the role of silviculture in managing the national forests: proceedings of the National Silviculture Workshop. Gen. Tech. Rep. NE-238. Radnor, PA: USDA Forest Service, Northeastern Forest Experiment Station: 104-109. http://www.treesearch.fs.fed.us/pubs/15099

Ferguson, B.A.; Dreisbach, T.A.; Parks, C.G.; Filip, G.M.; Schmitt, C.L. 2003. Coarse-scale population structure of pathogenic *Armillaria* species in a mixed-conifer forest in the Blue Mountains of northeast Oregon. Canadian Journal of Forest Research. 33(4): 612-623. doi:10.1139/x03-065

Ferguson, D.E.; Byrne, J.C.; Coffen, D.O. 2005. Reforestation trials and secondary succession with three levels of overstory shade in the grand fir mosaic ecosystem. Res. Pap. RMRS-RP-53. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 16 p. https://www.fs.usda.gov/treesearch/pubs/9020

Ferguson, D.E.; Johnson-Maynard, J.L.; McDaniel, P.A. 2007. The grand fir mosaic ecosystem–history and management impacts. In: Page-Dumroese, D.; Miller, R.; Mital, J.; McDaniel, P.; Miller, D., tech. eds. Volcanic-ash-derived forest soils of the inland northwest: properties and implications for management and restoration. Proceed. RMRS-P-44. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station: 175-184. http://www.treesearch.fs.fed.us/pubs/26223

Ferrell, G.T. 1978. Moisture stress threshold of susceptibility to fir engraver beetles in pole-size white firs. Forest Science. 24(1): 85-92. doi:10.1093/forestscience/24.1.85

Ferrell, G.T. 1986. Fir engraver. Forest Insect & Disease Leaflet 13. [Place of publication unknown]: USDA Forest Service. 8 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_042951.pdf

Ferrell, G.T.; Scharpf, R.F. 1982. Stem volume losses in grand firs topkilled by western spruce budworm in Idaho. Res. Pap. PSW-164. Berkeley, CA: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station. 10 p. https://www.fs.usda.gov/treesearch/pubs/28894

Ferrell, G.T.; Otrosina, W.J.; Demars, C.J., Jr. 1994. Predicting susceptibility of white fir during a drought-associated outbreak of the fir engraver, *Scolytus ventralis*, in California. Canadian Journal of Forest Research. 24(2): 302-305. doi:10.1139/x94-043

Ferry, G.W.; Clark, R.G.; Montgomery, R.E.; Mutch, R.W.; Leenhouts, W.P.; Zimmerman, G.T. 1995. Altered fire regimes within fire-adapted ecosystems. In: LaRoe, E.T.; Farris, G.S.; Puckett, C.E.; Doran, P.D.; Mac, M.J., eds. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems. Washington, DC: U.S. Department of the Interior, National Biological Service: 222-224. https://archive.org/download/ourlivingresourc00unit/ourlivingresourc00unit.pdf

Fettig, C.J.; Klepzig, K.D.; Billings, R.F.; Munson, A.S.; Nebeker, T.E.; Negron, J.F.; Nowak, J.T. 2007. The effectiveness of vegetation management practices for prevention and control of bark beetle infestations in coniferous forests of the western and southern United States. Forest Ecology and Management. 238(1-3): 24-53. doi:10.1016/j.foreco.2006.10.011

Fettig, C.; Borys, R.; Dabney, C. 2010. Effects of fire and fire surrogate treatments on bark beetle-caused tree mortality in the southern Cascades, California. Forest Science. 56(1): 60-73. doi:10.1093/forestscience/56.1.60

Fettig, C.J.; Mortenson, L.A.; Bulaon, B.M.; Foulk, P.B. 2019. Tree mortality following drought in the central and southern Sierra Nevada, California, U.S. Forest Ecology and Management. 432: 164-178. doi:10.1016/j.foreco.2018.09.006

Fialko, K.; Ex, S.; Wolk, B.H. 2020. Ecological niches of tree species drive variability in conifer regeneration abundance following fuels treatments. Forest Ecology and Management. 476: 118475 (13 p). doi:10.1016/j.foreco.2020.118475

Filip, G.M. 1990. Effects of tree harvesting on Armillaria root disease in an old-growth mixed-conifer stand in northeastern Oregon. Northwest Environmental Journal. 6(2): 412-413.

Filip, G.M. 1994. Forest health decline in central Oregon: a 13-year case study. Northwest Science. 68(4): 233-240. http://hdl.handle.net/2376/1557

Filip, G.M. 1995. Effects of silvicultural techniques on forest insects and disease pathogens in the Blue Mountains. Tech Notes BMNRI-TN-7. La Grande, OR: USDA Forest Service, Blue Mountains Natural Resources Institute. 4 p.

Filip, G.M. 2001. Managing tree wounding and stem decay in Oregon forests. EC 1519. Corvallis, OR: Oregon State University, Extension Service. 12 p. https://catalog.extension.oregonstate.edu/ec1519

Filip, G.M.; Goheen, D.J. 1984. Root diseases cause severe mortality in white and grand fir stands of the Pacific Northwest. Forest Science. 30(1): 138-142. doi:10.1093/forestscience/30.1.138

Filip, G.M.; Parks, C.A. 1987. Simultaneous infestation of dwarf mistletoe and western spruce budworm decreased growth of Douglas-fir in the Blue Mountains of Oregon. Forest Science. 33(3): 767-773. doi:10.1093/forestscience/33.3.767

Filip, G.M.; Schmitt, C.L. 1979. Susceptibility of native conifers to laminated root rot east of the Cascade Range in Oregon and Washington. Forest Science. 25(2): 261-265.

doi:10.1093/forestscience/25.2.261

Filip, G.M.; Schmitt, C.L. 1990. R_x for *Abies*: silvicultural options for diseased firs in Oregon and Washington. Gen. Tech. Rep. PNW-GTR-252. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 34 p. https://www.fs.usda.gov/treesearch/pubs/9100

Filip, G.M.; Yang-Erve, L. 1997. Effects of prescribed burning on the viability of *Armillaria ostoyae* in mixed-conifer forest soils in the Blue Mountains of Oregon. Northwest Science. 71(2): 137-144. http://hdl.handle.net/2376/1247

Filip, G.M.; Aho, P.E.; Wiitala, M.R. 1984. Strategies for reduction of decay in the interior Douglas-fir and grand fir forest types. In: Baumgartner, D.M.; Mitchell, R., comps. Silvicultural management strategies for pests of the interior Douglas-fir and grand fir forest types. Pullman, WA: Washington State University, Cooperative Extension: 73-80.

Filip, G.M.; Christiansen, E., Parks, C.A. 1989. Secondary resin production increases with vigor of *Abies grandis* inoculated with *Trichosporium symbioticum* in northeastern Oregon. Res. Note PNW-RN-489. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 10 p. https://www.fs.usda.gov/treesearch/pubs/21235

Filip, G.M.; Schwandt, J.W.; Hagle, S.K. 1990. Estimating decay in 40- to 90-year-old grand fir stands in the Clearwater region of northern Idaho. Res. Pap. PNW-RP-421. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 16 p. https://www.fs.usda.gov/treesearch/pubs/9256

Filip, G.M.; Schmitt, C.L.; Hosman, K.P. 1992a. Effects of harvesting season and stump size on incidence of annosus root disease of true fir. Western Journal of Applied Forestry. 7(2): 54-56. doi:10.1093/wjaf/7.2.54

Filip, G.M.; Wickman, B.E.; Mason, R.R.; Parks, C.A.; Hosman, K.P. 1992b. Thinning and nitrogen fertilization in a grand fir stand infested with western spruce budworm. Part III: tree wound dynamics. Forest Science. 38(2): 265-274. doi:10.1093/forestscience/38.2.265

Filip, G.M.; Colbert, J.J.; Shaw, C.G., III; Hessburg, P.F.; Hosman, K.P. 1993. Influence of dwarf mistletoe and western spruce budworm on growth and mortality of Douglas-fir in unmanaged stands. Forest Science. 39(3): 465-477. doi:10.1093/forestscience/39.3.465

Filip, G.M.; Colbert, J.J.; Parks, C.A.; Seidel, K.W. 1995a. Effects of thinning on growth and vigor of *Larix occidentalis* infected with *Arceuthobium laricis* in Oregon. In: Schmidt, W.C.; McDonald, K.J., comps. Ecology and management of Larix forests: a look ahead; proceedings of an international symposium. Gen. Tech. Rep. GTR-INT-319. Ogden, UT: USDA Forest Service, Intermountain Research Station: 307-309. https://archive.org/download/CAT10699418/CAT10699418.pdf

Filip, G.M.; Parks, C.A.; Wickman, B.E.; Mitchell, R.G. 1995b. Tree wound dynamics in thinned and unthinned stands of grand fir, ponderosa pine, and lodgepole pine in eastern Oregon. Northwest Science. 69(4): 276-283. http://hdl.handle.net/2376/1306

Filip, G.M.; Torgersen, T.R.; Parks, C.A.; Mason, R.R.; Wickman, B.E. 1996. Insect and disease factors in the Blue Mountains. In: Jaindl, R.G.; Quigley, T.M., eds. Search for a solution: sustaining the land, people, and economy of the Blue Mountains. Washington, DC: American Forests: 169-202. ASIN:B0030H52BC

Filip, G.M.; Fitzgerald, S.A.; Ganio, L.M. 1999. Precommercial thinning in a ponderosa pine stand affected by Armillaria root disease in central Oregon: 30 years of growth and mortality. Western Journal of Applied Forestry. 14(3): 144-148. doi:10.1093/wjaf/14.3.144

Filip, G.M.; Schmitt, C.L.; Parks, C.G. 2000. Mortality of mixed-conifer regeneration surrounding stumps infected by *Heterobasidion annosum* 15-19 years after harvesting in northeastern Oregon. Western Journal of Applied Forestry. 15(4): 189-194. doi:10.1093/wjaf/15.4.189

Filip, G.M.; Schmitt, C.L.; Chadwick, K.L. 2006. Incidence of *Heterobasidion annosum* in mixed-conifer natural regeneration surrounding large true fir stumps 20-25 years after harvesting in northeastern Oregon. Western Journal of Applied Forestry. 21(4): 178-184. doi:10.1093/wjaf/21.4.178

Filip, G.M.; Schmitt, C.L.; Scott, D.W.; Fitzgerald, S.A. 2007. Understanding and defining mortality in western conifer forests. Western Journal of Applied Forestry. 22(2): 105-115. doi:10.1093/wjaf/22.2.105

Filip, G.M.; Goheen, D.J.; Kimmey, J.W. 2009. Rust-red stringy rot caused by the Indian paint fungus. Forest Insect and Disease Leaflet 93; FS-R6-RO-FIDL#93/001-2009. Portland, OR: USDA Forest Service, Pacific Northwest Region. 11 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043028.pdf

Filip, G.M.; Maffei, H.M.; Chadwick, K.L.; Max, T.A. 2010. Armillaria root disease-caused tree mortality following silvicultural treatments (shelterwood or group selection) in an Oregon mixed-conifer forest: insights from a 10-year case study. Western Journal of Applied Forestry. 25(3): 136-143. doi:10.1093/wjaf/25.3.136

Filip, G.; Chadwick, K.; Zambino, P.; Omdal, D.; Ramsey-Kroll, A.; Schmitt, C.; Maffei, H.; Saavedra, A.; Rall, W.; Parks, C. 2011. Seven- to 14-year effects of artificially inoculating living conifers to promote stem decay and subsequent wildlife use in Oregon and Washington forests. Portland, OR: USDA Forest Service, Pacific Northwest Region, Forest Health Protection. 24 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5327687.pdf

Finegan, B. 1984. Forest succession. Nature. 312(5990): 109-114. doi:10.1038/312109a0

Finnegan, R.J. 1971. An appraisal of indigenous ants as limiting agents of forest pests in Quebec. Canadian Entomologist. 103(10): 1489-1493. doi:10.4039/Ent1031489-10

Finney, M.A. 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. Forest Science. 47(2): 219-228. doi:10.1093/forestscience/47.2.219

Finney, M.A. 2005. The challenge of quantitative risk analysis for wildland fire. Forest Ecology and Management. 211(1-2): 97-108. doi:10.1016/j.foreco.2005.02.010

Finney, M.A.; Seli, R.C.; McHugh, C.W.; Ager, A.A.; Bahro, B.; Agee, J.K. 2007. Simulation of long-term landscape-level fuel treatment effects on large wildfires. International Journal of Wildland Fire. 16(6): 712-727. doi:10.1071/WF06064

Fins, L.; Byler, J.; Ferguson, D.; Harvey, A.; Mahalovich, M.F.; McDonald, G.; Miller, D.; Schwandt, J.; Zack, A. 2001. Return of the giants: restoring white pine ecosystems by breeding and aggressive planting of blister rust-resistant white pines. Station Bull. 72. Moscow, ID: University of Idaho, College of Natural Resources, Idaho Forest, Wildlife and Range Experiment Station. 20 p. http://www.treesearch.fs.fed.us/pubs/35328

Fischer, A.P. 2018. Forest landscapes as social-ecological systems and implications for management. Landscape and Urban Planning. 177: 138-147. doi:10.1016/j.landurbplan.2018.05.001

Fischer, A.P. 2019a. Adapting and coping with climate change in temperate forests. Global Environmental Change. 54: 160-171. doi:10.1016/j.gloenvcha.2018.10.011

Fischer, A.P. 2019b. Characterizing behavioral adaptation to climate change in temperate forests. Landscape and Urban Planning. 188: 72-79. doi:10.1016/j.landurbplan.2018.09.024

Fischer, W.C.; Miller, M.; Johnston, C.M.; Smith, J.K.; Simmerman, D.G.; Brown, J.K. 1996. Fire Effects Information System: user's guide. Gen. Tech. Rep. INT-GTR-327. Ogden, UT: USDA Forest Service, Intermountain Research Station. 131 p. https://www.feis-crs.org/feis/

Fischer, A.P.; Spies, T.A.; Steelman, T.A.; Moseley, C.; Johnson, B.R.; Bailey, J.D.; Ager, A.A.; Bourgeron, P.; Charnley, S.; Collins, B.M.; Kline, J.D.; Leahy, J.E.; Littell, J.S.; Millington, J.D.; Nielsen-Pincus, M.; Olsen, C.S.; Paveglio, T.B.; Roos, Christopher I.; Steen-Adams, M.M.; Stevens, F.R.; Vukomanovic, J.; White, E.M.; Bowman, D.M. 2016.

Wildfire risk as a socioecological pathology. Frontiers in Ecology and the Environment. 14(5): 276-284. doi:10.1002/fee.1283

Fisher, G.M. 1935. Comparative germination of tree species on various kinds of surface-soil material in the western white pine type. Ecology. 16(4): 606-611. doi:10.2307/1932591

Fisher, R.F. 1980. Allelopathy: a potential cause of regeneration failure. Journal of Forestry. 78: 346-350. doi:10.1093/jof/78.6.346

Fisher, R.F.; Binkley, D. 2000. Ecology and management of forest soils. 3rd edition. New York, John Wiley & Sons, Inc. 489 p. isbn:0-471-19426-3

Fitzgerald, S.A. 1997. Applying uneven-aged management in forests heavily impacted by insects and disease in eastern Oregon. In: Emmingham, W.H., comp. Proceedings of the IUFRO interdisciplinary uneven-aged management symposium. [Corvallis, OR]: [Oregon State University]: 142.

Fitzgerald, S.A., ed. 2002. Fire in Oregon's forests: risks, effects, and treatment options. Portland, OR: Oregon Forest Resources Institute. 164 p.

Flamenco, H.N.; Gonzalez-Benecke, C.A.; Wightman, M.G. 2019. Long-term effects of vegetation management on biomass stock of four coniferous species in the Pacific Northwest United States. Forest Ecology and Management. 432: 276-285. doi:10.1016/j.foreco.2018.09.033

Flannigan, M.D.; Stocks, B.J.; Wotton, B.M. 2000. Climate change and forest fires. Science of the Total Environment. 262(3): 221-229. doi:10.1016/S0048-9697(00)00524-6

Flannigan, M.D.; Amiro, B.D.; Logan, K.A.; Stocks, B.J.; Wotton, B.M. 2005. Forest fires and climate change in the 21st century. Mitigation and Adaptation Strategies for Global Change. 11: 847-859. doi:10.1007/s11027-005-9020-7

Flannigan, M.D.; Krawchuk, M.A.; de Groot, W.J.; Wotton, B.M.; Gowman, L.M. 2009. Implications of changing climate for global wildland fire. International Journal of Wildland Fire. 18(5): 483-507. doi:10.1071/WF08187

Fleischman, F.; Struthers, C.; Arnold, G.; Dockry, M.; Scott, T. 2020. US Forest Service implementation of the National Environmental Policy Act: Fast, variable, rarely litigated, and declining. Journal of Forestry. 118(4): 403-418. doi:10.1093/jofore/fvaa016

Fleischner, T.L. 2005. Natural history and the deep roots of resource management. Natural Resources Journal. 45(1): 1-13. https://www.jstor.org/stable/24888961

Fleming, R.A.; Volney, W.J.A. 1995. Effects of climate change on insect defoliator population processes in Canada's boreal forest: some plausible scenarios. Water, Air, and Soil Pollution. 82(1): 445-454. doi:10.1007/BF01182854

Fleming, R.A.; Candau, J.-N.; McAlpine, R.S. 2002. Landscape-scale analysis of interactions between insect defoliation and forest fire in central Canada. Climatic Change. 55(1-2): 251-272. doi:10.1023/A:1020299422491

Flint, H.R. 1925. Fire resistance of northern Rocky Mountain conifers. Idaho Forester. 7: 7-10, 41-43.

Flint, C.; McFarlane, B.; Müller, M. 2009. Human dimensions of forest disturbance by insects: an international synthesis. Environmental Management. 43(6): 1174-1186. doi:10.1007/s00267-008-9193-4

Flower, A.; Murdock, T.Q.; Taylor, S.W.; Zwiers, F.W. 2013. Using an ensemble of downscaled climate model projections to assess impacts of climate change on the potential distribution of spruce and Douglas-fir forests in British Columbia. Environmental Science and Policy. 26: 63-74. doi:10.1016/j.envsci.2012.07.024

Flower, A.; Gavin, D.G.; Heyerdahl, E.K.; Parsons, R.A.; Cohn, G.M. 2014. Drought-triggered western spruce budworm outbreaks in the interior Pacific Northwest: A multi-century dendrochronological record. Forest Ecology and Management. 324: 16-27.

doi:10.1016/j.foreco.2014.03.042

Foiles, M.W. 1955. Thinning from below in a 60-year-old western white pine stand. Res. Note No. 19. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 6 p. https://www.fs.usda.gov/treesearch/pubs/48984

Foiles, M.W. 1959. Silvics of grand fir. Misc. Pub. No. 21. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 12 p. https://www.fs.usda.gov/treesearch/pubs/48994

Foiles, M.W.; Graham, R.T.; Olson, D.F., Jr. 1990. *Abies grandis* (Dougl. ex D. Don) Lindl.; grand fir. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America; volume 1, conifers. Agric. Handb. 654. Washington, DC: USDA Forest Service: 52-59. https://www.treesearch.fs.fed.us/pubs/1547 https://www.srs.fs.usda.gov/pubs/misc/ag_654/volume_1/abies/grandis.htm

Foit, F.F., Jr.; Mehringer, P.J., Jr.; Sheppard, J.C. 1993. Age, distribution, and stratigraphy of Glacier Peak tephra in eastern Washington and western Montana, United States. Canadian Journal of Earth Sciences. 30(3): 535-552. doi:10.1139/e93-042

Folke, C.; Carpenter, S.; Walker, B.; Scheffer, M.; Elmqvist, T.; Gunderson, L.; Holling, C.S. 2004. Regime shifts, resilience, and biodiversity in ecosystem management. Annual Review of Ecology, Evolution, and Systematics. 35(1): 557-581. doi:10.1146/annurev.ecolsys.35.021103.105711

Ford, C.R.; Laseter, S.H.; Swank, W.T.; Vose, J.M. 2011. Can forest management be used to sustain water-based ecosystem services in the face of climate change? Ecological Applications. 21(6): 2049-2067. doi:10.1890/10-2246.1

Forman, R.T.T.; Alexander, L.E. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics. 29: 207-231. doi:10.1146/annurev.ecolsys.29.1.207

Forman, R.T.T.; Godron, M. 1981. Patches and structural components for a landscape ecology. BioScience. 31(10): 733-740. doi:10.2307/1308780

Forman, R.T.T.; Godron, M. 1986. Landscape ecology. New York: John Wiley. 619 p. isbn:0-471-87037-4

Forrester, D.I. 2014. The spatial and temporal dynamics of species interactions in mixed-species forests: From pattern to process. Forest Ecology and Management. 312: 282-292. doi:10.1016/j.foreco.2013.10.003

Forrester, D.I. 2019. Linking forest growth with stand structure: Tree size inequality, tree growth or resource partitioning and the asymmetry of competition. Forest Ecology and Management. 447: 139-157. doi:10.1016/j.foreco.2019.05.053

Forsman, J.T.; Reunanen, P.; Jokimäki, J.; Mönkkönen, M. 2010. The effects of small-scale disturbance on forest birds: a meta-analysis. Canadian Journal of Forest Research. 40(9): 1833-1842. doi:10.1139/X10-126

Fortin, D.; Beyer, H.L.; Boyce, M.S.; Smith, D.W.; Duchesne, T.; Mao, J.S. 2005. Wolves influence elk movements: Behavior shapes a trophic cascade in Yellowstone National Park. Ecology. 86(5): 1320-1330. doi:10.1890/04-0953

Foster, H.D. 1906. Report on the silvics of the Wenaha Forest Reserve, Washington and Oregon. Unpublished typescript report obtained from the National Archives, College Park, MD; record group 95. 39 p. On file with: USDA Forest Service, Umatilla National Forest, Supervisor’s Office, Pendleton, OR. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015540.pdf

Foster, D.R.; Orwig, D.A.; McLachlan, J.S. 1996. Ecological and conservation insights from reconstructive studies of temperate old-growth forests. Trends in Ecology and Evolution. 11(10): 419-424. doi:10.1016/0169-5347(96)10047-1

Foster, D.R.; Aber, J.D.; Melillo, J.M.; Bowden, R.D.; Bazzaz, F.A. 1997. Forest response to disturbance and anthropogenic stress. BioScience. 47(7): 437-445. doi:10.2307/1313059

Foster, D.R.; Knight, D.H.; Franklin, J.F. 1998. Landscape patterns and legacies resulting from large, infrequent forest disturbances. Ecosystems. 1(6): 497-510. doi:10.1007/s100219900046

Foster, D.; Swanson, F.; Aber, J.; Burke, I.; Brokaw, N.; Tilman, D.; Knapp, A. 2003. The importance of land-use legacies to ecology and conservation. BioScience. 53(1): 77-88. doi:10.1641/0006-3568(2003)053[0077:TIOLUL]2.0.CO;2

Foster, C.N.; Barton, P.S.; Robinson, N.M.; MacGregor, C.I.; Lindenmayer, D.B. 2017a. Effects of a large wildfire on vegetation structure in a variable fire mosaic. Ecological Applications. 27(8): 2369-2381. doi:10.1002/eap.1614

Foster, A.C.; Walter, J.A.; Shugart, H.H.; Sibold, J.; Negron, J. 2017b. Spectral evidence of early-stage spruce beetle infestation in Engelmann spruce. Forest Ecology and Management. 384: 347-357. doi:10.1016/j.foreco.2016.11.004

Fox, J.F. 1989. Bias in estimating forest disturbance rates and tree lifetimes. Ecology. 70(5): 1267-1272. doi:10.2307/1938185

Fox, T.R. 2000. Sustained productivity in intensively managed forest plantations. Forest Ecology and Management. 138(1-3): 187-202. doi:10.1016/s0378-1127(00)00396-0

Franceschi, V.R.; Krokene, P.; Christiansen, E.; Krekling, T. 2005. Anatomical and chemical defenses of conifer bark against bark beetles and other pests. New Phytologist. 167(2): 353-376. doi:10.1111/j.1469-8137.2005.01436.x

Franklin, J.F. 1993. Preserving biodiversity: Species, ecosystems, or landscapes? Ecological Applications. 3(2): 202-205. doi:10.2307/1941820

Franklin, J.F.; Agee, J.K. 2003. Forging a science-based national forest fire policy. Issues in Science and Technology. 20(1): 59-66. https://issues.org/franklin/

Franklin, J.F.; Forman, R.T.T. 1987. Creating landscape patterns by forest cutting: ecological consequences and principles. Landscape Ecology. 1(1): 5-18. doi:10.1007/BF02275261

Franklin, J.F.; Johnson, K.N. 2012. A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry. 110(8): 429-439. doi:10.5849/jof.10-006

Franklin, J.F.; Van Pelt, R. 2004. Spatial aspects of structural complexity in old-growth forests. Journal of Forestry. 102(3): 22-28. doi:10.1093/jof/102.3.22

Franklin, J.F.; Cromack, K., Jr.; Denison, W.; McKee, A.; Maser, C.; Sedell, J.; Swanson, F.; Juday, G. 1981. Ecological characteristics of old-growth Douglas-fir forests. Gen. Tech. Rep. PNW-118. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 48 p. https://www.fs.usda.gov/treesearch/pubs/5546

Franklin, J.F.; Shugart, H.H.; Harmon, M.E. 1987. Tree death as an ecological process. BioScience. 37(8): 550-556. doi:10.2307/1310665

Franklin, J.F.; Spies, T.A.; Van Pelt, R.; Carey, A.B.; Thornburgh, D.A.; Berg, D.R.; Lindenmayer, D.B.; Harmon, M.E.; Keeton, W.S.; Shaw, D.C.; Bible, K.; Chen, J. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management. 155(1-3): 399-423. doi:10.1016/S0378-1127(01)00575-8

Franklin, J.F.; Mitchell, R.J.; Palik, B.J. 2007. Natural disturbance and stand development principles for ecological forestry. Gen. Tech. Rep. NRS-19. Newtown Square, PA: USDA Forest Service, Northern Research Station. 44 p. http://www.treesearch.fs.fed.us/pubs/13293

Fraser, A.I. 1962. The soil and roots as factors in tree stability. Forestry. 35(2): 117-127. doi:10.1093/forestry/35.2.117

Frederick, D.J.; Partridge, A.D. 1977. Indicators of decay in the grand fir/white fir complex in central Idaho. Northwest Science. 51(4): 282-292.

Frelich, L.E. 2003. Forest dynamics and disturbance regimes: studies from temperate evergreen–deciduous forests. Cambridge, UK: Cambridge University Press. 266 p. isbn:0-521-65082-8

Fremont, J.C. 1845. Report of the exploring expedition to the Rocky Mountains in the year 1842, and to Oregon and north California in the years 1843-'44. Washington, DC: Gales and Seaton, Printers. 707 p. doi:10.5962/bhl.title.46894 https://archive.org/download/mobot31753002784947/mobot31753002784947.pdf

Fried, J.S.; Gilless, J.K.; Riley, W.J.; Moody, T.J.; Simon de Blas, C.; Hayhoe, K.; Moritz, M.; Stephens, S.; Torn, M. 2008. Predicting the effect of climate change on wildfire behavior and initial attack success. Climatic Change. 87(1 Supplement): 251-264. doi:10.1007/s10584-007-9360-2

Fries, C.; Johansson, O.; Pettersson, B.; Simonsson, P. 1997. Silvicultural models to maintain and restore natural stand structures in Swedish boreal forests. Forest Ecology and Management. 94(1-3): 89-103. doi:10.1016/S0378-1127(97)00003-0

Fritts, S.R.; Moorman, C.E.; Hazel, D.W.; Jackson, B.D. 2014. Biomass harvesting guidelines affect downed woody debris retention. Biomass and Bioenergy. 70: 382-391. doi:10.1016/j.biombioe.2014.08.010

Froehlich, H.A.; Miles, D.W.R.; Robbins, R.W. 1985. Soil bulk density recovery on compacted skid trails in central Idaho. Soil Science Society of America Journal. 49(4): 1015-1017. doi:10.2136/sssaj1985.03615995004900040045x

Fry, D.L.; Stephens, S.L. 2006. Influence of humans and climate on the fire history of a ponderosa pine-mixed conifer forest in the southeastern Klamath Mountains, California. Forest Ecology and Management. 223(1-3): 428-438. doi:10.1016/j.foreco.2005.12.021

Fry, D.L.; Stephens, S.L.; Collins, B.M.; North, M.P.; Franco-Vizcaino, E.; Gill, S.J. 2014. Contrasting spatial patterns in active-fire and fire-suppressed Mediterranean climate old-growth mixed conifer forests. PLoS ONE. 9(2): e88985 (15 p). doi:10.1371/journal.pone.0088985

Fryer, J.L. 2011. *Alnus incana*. In: Fire Effects Information System [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us/database/feis/plants/tree/alninc/all.html

Fryxell, R. 1965. Mazama and Glacier Peak volcanic ash layers: relative ages. Science. 147(3663): 1288-1290. doi:10.1126/science.147.3663.1288

Fulé, P.Z. 2008. Does it make sense to restore wildland fire in changing climate? Restoration Ecology. 16(4): 526-531. doi:10.1111/j.1526-100X.2008.00489.x

Furniss, R.L. 1947. Memorandum regarding insect damage in the upper Umatilla River country. Unpub. Typescript Memoranda. Portland, OR: USDA Bureau of Entomology and Plant Quarantine, Forest Insect Laboratory. 5 p.

Furniss, M.M. 1962. Infestation patterns of Douglas-fir beetle in standing and windthrown trees in southern Idaho. Journal of Economic Entomology. 55(4): 486-491. doi:10.1093/jee/55.4.486

Furniss, M.M.; Kegley, S.J. 2014. Douglas-fir beetle. Forest Insect & Disease Leaflet 5; FS/R6/RO/FIDL#5-14/001. Portland, OR: USDA Forest Service, Pacific Northwest Region. 12 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5345470.pdf

Furniss, M.M.; McGregor, M.D.; Foiles, M.W.; Partridge, A.D. 1979. Chronology and characteristics of a Douglas-fir beetle outbreak in northern Idaho. Gen. Tech. Rep. INT-59. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 19 p. https://archive.org/download/IND79104938/IND79104938.pdf

Furniss, M.M.; Livingston, R.L.; McGregor, M.D. 1981. Development of a stand susceptibility classification for Douglas-fir beetle. In: Hedden, R.L.; Barras, S.J.; Coster, J.E., eds. Hazard-rating systems in forest pest management: symposium proceedings. Gen. Tech. Rep.

WO-27. Washington, DC: USDA Forest Service: 115-128. https://archive.org/download/CAT91960247/CAT91960247.pdf

Furniss, M.J.; Staab, B.P.; Hazelhurst, S.; Clifton, C.F.; Roby, K.B.; Ilhadrt, B.L.; Larry, E.B.; Todd, A.H.; Reid, L.M.; Hines, S.J.; Bennett, K.A.; Luce, C.H.; Edwards, P.J. 2010. Water, climate change, and forests: watershed stewardship for a changing climate. Gen. Tech. Rep. PNW-GTR-812. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 75 p. http://www.treesearch.fs.fed.us/pubs/35295

Gail, F.W. 1921. Factors controlling the distribution of Douglas fir in semi-arid regions of the northwest. Ecology. 2(4): 281-291. doi:10.2307/1928982

Gaines, W.; Haggard, M.; Begley, J.; Lehmkuhl, J.; Lyons, A. 2010. Short-term effects of thinning and burning restoration treatments on avian community composition, density, and nest survival in the eastern Cascades dry forests, Washington. Forest Science. 56(1): 88-99. doi:10.1093/forestscience/56.1.88

Gale, M.R.; Grigal, D.F. 1987. Vertical root distribution of northern tree species in relation to successional status. Canadian Journal of Forest Research. 17(8): 829-834. doi:10.1139/x87-131

Gallant, A.L.; Hansen, A.J.; Councilman, J.S.; Monte, D.K.; Betz, D.W. 2003. Vegetation dynamics under fire exclusion and logging in a Rocky Mountain watershed, 1856-1996. Ecological Applications. 13(2): 385-403. doi:10.1890/1051-0761(2003)013[0385:VDUFEA]2.0.CO;2

Gallet, C. 1994. Allelopathic potential in bilberry-spruce forests: influence of phenolic compounds on spruce seedlings. Journal of Chemical Ecology. 20(5): 1009-1024. doi:10.1007/BF02059738

Ganey, J.L.; Vojta, S.C. 2011. Tree mortality in drought-stressed mixed-conifer and ponderosa pine forests, Arizona, USA. Forest Ecology and Management. 261(1): 162-168. doi:10.1016/j.foreco.2010.09.048

Ganio, L.M.; Progar, R.A. 2017. Mortality predictions of fire-injured large Douglas-fir and ponderosa pine in Oregon and Washington, USA. Forest Ecology and Management. 390: 47-67. doi:10.1016/j.foreco.2017.01.008

Gara, R.I.; Geiszler, D.R.; Littke, W.R. 1984. Primary attraction of the mountain pine beetle to lodgepole pine in Oregon. Annals of the Entomological Society of America. 77(4): 333-334. doi:10.1093/aesa/77.4.333

Gara, R.I.; Littke, W.R.; Agee, J.K.; Geiszler, D.R.; Stuart, J.D.; Driver, C.H. 1985. Influence of fires, fungi and mountain pine beetles on development of a lodgepole pine forest in south-central Oregon. In: Baumgartner, D.M.; Krebill, R.G.; Arnott, J.T.; Weetman, G.F., comps. Lodgepole pine: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Cooperative Extension, Office of Conferences and Institutes: 153-162.

Gardiner, B.A., Quine, C.P. 2000. Management of forests to reduce the risk of abiotic damage – a review with particular reference to the effects of strong winds. Forest Ecology and Management. 135(1-3): 261-277. doi:10.1016/S0378-1127(00)00285-1

Gardner, W.J. 1905. Results of a Rocky Mountain forest fire studied fifty years after its occurrence. Proceedings of the Society of American Foresters. 1: 102-109. Proceedings_of_the_Society_of_American_Foresters

Garrison, M.T.; Moore, J.A. 2002. Tamarack #1 nutrient tracking study; Umatilla National Forest, Heppner Ranger District, completion report. Unpub. Rep. [Moscow, ID]: University of Idaho, Intermountain Forest Tree Nutrition Cooperative. 17 p.

Garrison, M.T.; Moore, J.A.; Shaw, T.M.; Mika, P.G. 2000. Foliar nutrient and tree growth response of mixed-conifer stands to three fertilization treatments in northeast Oregon and north central Washington. Forest Ecology and Management. 132(2-3): 183-198.

doi:10.1016/S0378-1127(99)00228-5

Garrison-Johnston, M.T. 2010. Northeast Oregon IFTNC region nutrition guidelines by rock type. Unpub. Rep. Moscow, ID: University of Idaho, Forest Resources Department, Intermountain Forest Tree Nutrition Cooperative. 60 p.

Garrison-Johnston, M.T.; Johnson, L.R. 2008. Washington state IFTNC regions nutrition guidelines by rock type. Unpub. Rep. Moscow, ID: University of Idaho, Forest Resources Department, Intermountain Forest Tree Nutrition Cooperative. 74 p.

Garrison-Johnston, M.T.; Moore, J.A.; Cook, S.P.; Niehoff, G.J. 2003. Douglas-fir beetle infestations are associated with certain rock and stand types in the inland northwestern United States. Environmental Entomology. 32(6): 1354-1363. doi:10.1603/0046-225X-32.6.1354

Garrison-Johnston, M.T.; Mika, P.G.; Miller, D.L.; Cannon, P.; Johnson, L.R. 2007. Ash cap influences on site productivity and fertilizer response in forests of the inland northwest. In: Page-Dumroese, D.S.; Miller, R.E.; Mital, J.; McDaniel, P.; Miller, D., tech. eds. Volcanic-ash-derived forest soils of the inland northwest: properties and implications for management and restoration. Proceed. RMRS-P-44. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station: 137-163. https://www.fs.usda.gov/treesearch/pubs/26202

Garrity, S.R.; Allen, C.D.; Brumby, S.P.; Gangodagamage, C.; McDowell, N.G.; Cai, D.M. 2013. Quantifying tree mortality in a mixed species woodland using multitemporal high spatial resolution satellite imagery. Remote Sensing of Environment. 129: 54-65. doi:10.1016/j.rse.2012.10.029

Garten, C.T., Jr. 2009. A disconnect between O horizon and mineral soil carbon – Implications for soil C sequestration. Acta Oecologica. 35(2): 218-226. doi:10.1016/j.actao.2008.10.004

Gärtner, S.; Reynolds, K.M.; Hessburg, P.F.; Hummel, S.; Twery, M. 2008. Decision support for evaluating landscape departure and prioritizing forest management activities in a changing environment. Forest Ecology and Management. 256(10): 1666-1676. doi:10.1016/j.foreco.2008.05.053

Garton, E.O. 1987. Habitat requirements of avian predators. In: Brookes, M.H.; Campbell, R.W.; Colbert, J.J.; Mitchell, R.G.; Stark, R.W., tech. coords. Western spruce budworm. Tech. Bull. 1694. Washington, DC: USDA Forest Service, Cooperative State Research Service: 82-85. https://digitalcommons.usu.edu/cgi/viewcontent.cgi?article=1318&context=barkbeetles

Gast, W.R., Jr.; Scott, D.W.; Schmitt, C.; Clemens, D.; Howes, S.; Johnson, C.G., Jr.; Mason, R.; Mohr, F.; Clapp, R.A. 1991. Blue Mountains forest health report: "new perspectives in forest health." Portland, OR: USDA Forest Service, Pacific Northwest Region, Malheur, Umatilla, and Wallowa-Whitman National Forests. Irregular pagination. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015666.pdf

Gavin, D.G.; Hu, F.S.; Lertzman, K.; Corbett, P. 2006. Weak climatic control of stand-scale fire history during the late Holocene. Ecology. 87(7): 1722-1732. doi:10.1890/0012-9658(2006)87[1722:WCCOSF]2.0.CO;2

Gavin, D.G.; Hallett, D.J.; Hu, F.S.; Lertzman, K.P.; Prichard, S.J.; Brown, K.J.; Lynch, J.A.; Bartlein, P.; Peterson, D.L. 2007. Forest fire and climate change in western North America: insights from sediment charcoal records. Frontiers in Ecology and the Environment. 5(9): 499-506. doi:10.1890/060161

Gedalof, Z. 2011. Climate and spatial patterns of wildfire in North America. In: McKenzie, D.; Miller, C.; Falk, D.A., eds. The landscape ecology of fire. New York: Springer: 89-115. isbn:978-94-007-0300-1

Gedalof, Z.; Peterson, D.L.; Mantua, N.J. 2005. Atmospheric, climatic, and ecological controls on extreme wildfire years in the northwestern United States. Ecological Applications. 15(1): 154-174. doi:10.1890/03-5116

Gedney, D.R. 1963. Toward complete use of eastern Oregon's forest resources. Res. Bull. PNW-3. Portland, OR: Pacific Northwest Forest and Range Experiment Station. 71 p. https://www.fs.usda.gov/treesearch/pubs/26699

Geier-Hayes, K. 1987. Occurrence of conifer seedlings and their microenvironments on disturbed sites in central Idaho. Res. Pap. INT-383. Ogden, UT: USDA Forest Service, Intermountain Research Station. 12 p. https://archive.org/download/occurrenceofconi383geie/occurrenceofconi383geie.pdf

Geils, B.W.; Hawksworth, F.G. 2002. Damage, effects, and importance of dwarf mistletoes. Chapter 5. In: Geils, B.W.; Tovar, J.C.; Moody, B., tech. coords. Mistletoes of North American conifers. Gen. Tech. Rep. RMRS-GTR-98. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station: 57-65. https://www.fs.usda.gov/treesearch/pubs/4812

Geils, B.W.; Tovar, J.C.; Moody, B., tech. coords. 2002. Mistletoes of North American conifers. Gen. Tech. Rep. RMRS-GTR-98. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station. 123 p. https://www.fs.usda.gov/treesearch/pubs/4812

Geils, B.W.; Hummer, K.E.; Hunt, R.S. 2010. White pines, Ribes, and blister rust: a review and synthesis. Forest Pathology. 40(3-4): 147-185. doi:10.1111/j.1439-0329.2010.00654.x

Geist, J.M.; Cochran, P.H. 1991. Influence of volcanic ash and pumice deposition on productivity of western interior forest soils. In: Harvey, A.E.; Neuenschwander, L.F., comps. Proceedings–management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: USDA Forest Service, Intermountain Research Station: 82-89. http://www.treesearch.fs.fed.us/pubs/33908

Geist, J.M.; Strickler, G.S. 1978. Physical and chemical properties of some Blue Mountain soils in northeast Oregon. Res. Pap. PNW-236. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 19 p. https://archive.org/download/CAT92273165/CAT92273165.pdf

Geist, J.M.; Hazard, J.W.; Seidel, K.W. 1989. Assessing physical conditions of some Pacific Northwest volcanic ash soils after forest harvest. Soil Science Society of America Journal. 53(3): 946-950. doi:10.2136/sssaj1989.03615995005300030052x

Geist, J.M.; Hazard, J.W.; Seidel, K.W. 2008. Juvenile tree growth on some volcanic ash soils disturbed by prior forest harvest. Res. Pap. PNW-RP-573 Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 22 p. http://www.treesearch.fs.fed.us/pubs/29522

Geiszler, D.R.; Gara, R.I.; Driver, C.H.; Gallucci, V.F.; Martin, R.E. 1980. Fire, fungi, and beetle influences on a lodgepole pine ecosystem of south-central Oregon. Oecologia. 46(2): 239-243. doi:10.1007/BF00540132

Gerlach, J.P.; Reich, P.B.; Puettmann, K.; Baker, T. 1997. Species, diversity, and density affect tree seedling mortality from Armillaria root rot. Canadian Journal of Forest Research. 27(9): 1509-1512. doi:10.1139/cjfr-27-9-1509

Germino, M.J.; Smith, W.K. 1999. Sky exposure, crown architecture, and low-temperature photoinhibition in conifer seedlings at alpine treeline. Plant, Cell & Environment. 22(4): 407-415. doi:10.1046/j.1365-3040.1999.00426.x

Getzin, S.; Wiegand, T.; Wiegand, K.; He, F. 2008. Heterogeneity influences spatial patterns and demographics in forest stands. Journal of Ecology. 96(4): 807-820. doi:10.1111/j.1365-2745.2008.01377.x

Gibbs, J.N.; Wainhouse, D. 1986. Spread of forest pests and pathogens in the northern hemisphere. Forestry. 59(2): 141-153. doi:10.1093/forestry/59.2.141

Gibson, K.; Kegley, S.; Bentz, B. 2009. Mountain pine beetle. Forest Insect & Disease Leaflet 2; FS-R6-RO-FIDL#2/002-2009. Portland, OR: USDA Forest Service, Pacific Northwest

Region. 12 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_042835.pdf

Gier, J.M.; Kindel, K.M.; Page-Dumroese, D.S.; Kuennen, L.J. 2018. Soil disturbance recovery on the Kootenai National Forest, Montana. Gen. Tech. Rep. RMRS-GTR-380. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 31 p. https://www.fs.usda.gov/treesearch/pubs/56603

Gilkey, H.P. 1912. Study of western larch. Unpublished typescript report obtained from the National Archives, College Park, MD; record group 95. 18 p. On file with: USDA Forest Service, Umatilla National Forest, Supervisor's Office, Pendleton, OR. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5415179.pdf

Gleick, P.H. 1998. Water in crisis: Paths to sustainable water use. Ecological Applications. 8(3): 571-579. doi:10.1890/1051-0761(1998)008[0571:Wicpts]2.0.Co;2

Gluck, M.J.; Rempel, R.S. 1996. Structural characteristics of post-wildfire and clearcut landscapes. Environmental Monitoring and Assessment. 39(1-3): 435-450. doi:10.1007/BF00396160

Goeking, S.A.; Tarboton, D.G. 2020. Forests and water yield: A synthesis of disturbance effects on streamflow and snowpack in western coniferous forests. Journal of Forestry. 118(2): 172-192. doi:10.1093/jofore/fvz069

Goforth, B.R.; Minnich, R.A. 2008. Densification, stand-replacement wildfire, and extirpation of mixed conifer forest in Cuyamaca Rancho State Park, southern California. Forest Ecology and Management. 256(1-2): 36-45. doi:10.1016/j.foreco.2008.03.032

Goheen, E.M.; Willhite, E.A. 2006. Field guide to the common diseases and insect pests of Oregon and Washington conifers. Portland, OR: USDA Forest Service, Pacific Northwest Region. 325 p. isbn:0-16-076244-8

González, G.; Seastedt, T.R.; Donato, Z. 2003. Earthworms, arthropods and plant litter decomposition in aspen (*Populus tremuloides*) and lodgepole pine (*Pinus contorta*) forests in Colorado, USA. Pedobiologia. 47(5-6): 863-869. doi:10.1078/0031-4056-00272

Goodwin, M.J.; North, M.P.; Zald, H.S.J.; Hurteau, M.D. 2018. The 15-year post-treatment response of a mixed-conifer understory plant community to thinning and burning treatments. Forest Ecology and Management. 429: 617-624. doi:10.1016/j.foreco.2018.07.058

Gordon, J.C.; Lyons, J. 1997. The emerging role of science and scientists in ecosystem management. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press: 447-453. isbn:1-55963-399-9

Gordon, J.C.; Wheeler, C.T.; Perry, D.A., eds. [1979.] Symbiotic nitrogen fixation in the management of temperate forests; proceedings of a workshop held April 2-5, 1979. Corvallis, OR: Oregon State University, Forest Research Laboratory. 501 p.

Gosz, J.R.; Asher, J.; Holder, B.; Knight, R.; Naiman, R.; Raines, G.; Stine, P.; Wigley, T.B. 1999. An ecosystem approach for understanding landscape diversity. Chapter 3. In: Johnson, N.C.; Malk, A.J.; Szaro, R.C.; Sexton, W.T., eds. Ecological stewardship: a common reference for ecosystem management; volume II. Oxford, UK: Elsevier Science Ltd.: 157-194.

Graham, S.A. 1941. Climax forests of the upper peninsula of Michigan. Ecology. 22(4): 355-362. doi:10.2307/1930708

Graham, R.T. 1994. Silviculture, fire and ecosystem management. Journal of Sustainable Forestry. 2(3-4): 339-351. doi:10.1300/J091v02n03_08

Graham, R.T.; Jain, T.B. 2004. Past, present, and future role of silviculture in forest management. In: Shepperd, W.D.; Eskew, L.G., comps. Silviculture in special places: proceedings of the National Silviculture Workshop. Proceedings RMRS-P-34. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station: 1-14. http://www.fs.fed.us/rm/pubs/rmrs_p034/rmrs_p034_001_014.pdf

Graham, R.T.; Jain, T.B. 2005a. Application of free selection in mixed forests of the inland northwestern United States. Forest Ecology and Management. 209(1-2): 131-145. doi:10.1016/j.foreco.2005.01.019

Graham, R.T.; Jain, T.B. 2005b. Silvicultural tools applicable in forests burned by a mixed severity fire regime. In: Taylor, L.; Zelnik, J.; Cadwallader, S.; Hughes, B., eds. Mixed severity fire regimes: ecology and management. AFE MISC03. Pullman, WA: Washington State University Extension: 45-58.

Graham, D.A.; Mounts, J.; Almas, D. 1975. 1974 cooperative Douglas-fir tussock moth control project; Oregon, Washington, Idaho. Unnum. Rep. Portland, OR: USDA Forest Service, Pacific Northwest Region. 74 p. 1974_Cooperative_Douglas_fir_Tussock_Mot

Graham, R.T.; Minore, D.; Harvey, A.E.; Jurgensen, M.F.; Page-Dumroese, D.S. 1991. Soil management as an integral part of silvicultural systems. In: Harvey, A.E.; Neuenschwander, L.F., comps. Proceedings–management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: USDA Forest Service, Intermountain Research Station: 59-64. http://www.treesearch.fs.fed.us/pubs/33908

Graham R.T.; Harvey, A.E.; Jurgensen, M.F.; Jain, T.B.; Tonn, J.R.; Page-Dumroese, D.S. 1994a. Managing coarse woody debris in forests of the Rocky Mountains. Res. Pap. INT-RP-477. Ogden, UT: USDA Forest Service, Intermountain Research Station. 12 p. http://www.treesearch.fs.fed.us/pubs/24829

Graham, R.T.; Tonn, J.R.; Jain, T.B.; Adams, D.L. 1994b. The role of silviculture in ecosystem management: a practice in transition. In: Baumgartner, D.M.; Lotan, J.E.; Tonn, J.R., comps. Interior cedar-hemlock-white pine forests: ecology and management. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 269-275.

Graham, R.T.; Harvey, A.E.; Jain, T.B.; Tonn, J.R. 1999. The effects of thinning and similar stand treatments on fire behavior in western forests. Gen. Tech. Rep. PNW-GTR-463. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 27 p. http://www.treesearch.fs.fed.us/pubs/2979

Graham, R.T.; Jain, T.B.; Tonn, J.R. 1999. Uneven-aged silviculture in cedar-hemlock-grand fir ecosystems of the northern Rocky Mountains. In: Emmingham, W.H., comp. Proceedings of the IUFRO interdisciplinary uneven-aged management symposium. [Corvallis, OR]: [Oregon State University]: 70-87.

Graham, R.T.; McCaffrey, S.; Jain, T.B., tech. eds. 2004. Science basis for changing forest structure to modify wildfire behavior and severity. Gen. Tech. Rep. RMRS-GTR-120. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 43 p. http://www.treesearch.fs.fed.us/pubs/6279

Graham, R.T.; Jain, T.B.; Cannon, P. 2005. Stand establishment and tending in the inland northwest. In: Harrington, C.A.; Schoenholtz, S.H., tech. eds. Productivity of western forests: a forest products focus. Gen. Tech. Rep. PNW-GTR-642. Portland, OR: USDA Forest Service, Pacific Northwest Research Station: 47-78. http://www.treesearch.fs.fed.us/pubs/20635

Graham, R.T.; Jain, T.B.; Sandquist, J. 2007. Free selection: a silvicultural option. In: Powers, R., tech. ed. Restoring fire-adapted ecosystems: proceedings of the 2005 national silviculture workshop. Gen. Tech. Rep. PSW-GTR-203. Albany, CA: USDA Forest Service, Pacific Southwest Research Station: 121-156. http://www.treesearch.fs.fed.us/pubs/25897

Graham, R.T.; Jain, T.B.; Kingery, J.L. 2010. Ameliorating conflicts among deer, elk, cattle and/or other ungulates and other forest uses: a synthesis. Forestry. 83(3): 245-255. doi:10.1093/forestry/cpq003

Granhall, U.; Lindberg, T. 1980. Nitrogen input through biological nitrogen fixation. Ecological Bulletins. 32: 333-340. https://www.jstor.org/stable/20112822

Grant, G.E.; Tague, C.L.; Allen, C.D. 2013. Watering the forest for the trees: an emerging priority for managing water in forest landscapes. Frontiers in Ecology and the Environment. 11(6): 314-321. doi:10.1890/120209

Gray, R.W. 2005. Identifying and restoring forest structure in mixed severity fire regimes. In: Taylor, L.; Zelnik, J.; Cadwallader, S.; Hughes, B., eds. Mixed severity fire regimes: ecology and management. AFE MISC03. Pullman, WA: Washington State University Extension: 59-67.

Gray, G.; Clark, L. 1992. The changing science of forest health. American Forests. 98(11/12): 13-16.

Gray, A.N.; Zald, H.S.J.; Kern, R.A.; North, M. 2005. Stand conditions associated with tree regeneration in Sierran mixed-conifer forests. Forest Science. 51(3): 198-210. doi:10.1093/forestscience/51.3.198

Grayson, L.M.; Progar, R.A.; Hood, S.M. 2017. Predicting post-fire tree mortality for 14 conifers in the Pacific Northwest, USA: Model evaluation, development, and thresholds. Forest Ecology and Management. 399: 213-226. doi:10.1016/j.foreco.2017.05.038

Green, P.; Jensen, M. 1991. Plant succession within managed grand fir forests of northern Idaho. In: Harvey, A.E.; Neuenschwander, L.F., comps. Proceedings–management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: USDA Forest Service, Intermountain Research Station: 232-236. http://www.treesearch.fs.fed.us/pubs/33908

Greene, D.F.; Johnson, E.A. 2000. Tree recruitment from burn edges. Canadian Journal of Forest Research. 30(8): 1264-1274. doi:10.1139/x00-040

Greene, D.F.; Zasada, J.C.; Sirois, L.; Kneeshaw, D.; Morin, H.; Charron, I.; Simard, M.-J. 1999. A review of the regeneration dynamics of North American boreal forest tree species. Canadian Journal of Forest Research. 29(6): 824-839. doi:10.1139/x98-112

Greenlee, D.; Greenlee, J. 2002. Changes in fire hazard as a result of the Cerro Grande fire. Fire Management Today. 62(1): 15-21. http://www.fs.fed.us/sites/default/files/legacy_files/fire-management-today/62-1.pdf

Greiner, S.M.; Grimm, K.E.; Waltz, A.E.M. 2020. Managing for resilience? Examining management implications of resilience in southwestern national forests. Journal of Forestry. 118(4): 433-443. doi:10.1093/jofore/fvaa006

Griess, V.C.; Acevedo, R.; Härtl, F.; Staupendahl, K.; Knoke, T. 2012. Does mixing tree species enhance stand resistance against natural hazards? A case study for spruce. Forest Ecology and Management. 267: 284-296. doi:10.1016/j.foreco.2011.11.035

Griffith, R.S. 1992. *Pinus monticola*. In: Fire Effects Information System [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us/database/feis/plants/tree/pinmot/all.html

Grigal, D.F. 2000. Effects of extensive forest management on soil productivity. Forest Ecology and Management. 138(1-3): 167-185. doi:10.1016/S0378-1127(00)00395-9

Grigal, D.F.; Vance, E.D. 2000. Influence of soil organic matter on forest productivity. New Zealand Journal of Forestry Science. 30(1/2): 169-205. http://www.scionresearch.com/general/science-publications/science-publications/nzjfs/previous-volumes/nzjfs-volume-30

Grimm, N.B.; Chapin, F.S., III; Bierwagen, B.; Gonzalez, P.; Groffman, P.M.; Luo, Y.; Melton, F.; Nadelhoffer, K.; Pairis, A.; Raymond, P.A.; Schimel, J.; Williamson, C.E. 2013. The impacts of climate change on ecosystem structure and function. Frontiers in Ecology and the Environment. 11(9): 474-482. doi:10.1890/120282

Gruell, G.E. 1983. Fire and vegetative trends in the Northern Rockies: interpretations from 1871-1982 photographs. Gen. Tech. Rep. INT-158. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 117 p. http://www.treesearch.fs.fed.us/pubs/32994

Gruell, G.E. 1985. Indian fires in the interior west: a widespread influence. In: Lotan, J.E.; Kilgore, B.M.; Fischer, W.C.; Mutch, R.W., tech. coords. Proceedings–symposium and workshop on wilderness fire. Gen. Tech. Rep. INT-182. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 68-74. https://archive.org/download/CAT31118982/CAT31118982.pdf

Gruell, G.E. 2001. Fire in Sierra Nevada forests: a photographic interpretation of ecological change since 1849. Missoula, MT: Mountain Press Publishing Company. 238 p. isbn:0-87842-446-6

Grumbine, R.E. 1994. What is ecosystem management? Conservation Biology. 8(1): 27-38. doi:10.1046/j.1523-1739.1994.08010027.x

Gucker, C. 2012. Betula occidentalis. In: Fire Effects Information System [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us /database/feis/plants/tree/betocc/all.html

Guldin, J.M. 1996. Ecological basis of uneven-aged silviculture. In: Baker, J.B.; Cain, M.D.; Guldin, J.M.; Murphy, P.A.; Shelton, M.G., eds. Uneven-aged silviculture for the loblolly and shortleaf pine forest cover types. Gen. Tech. Rep. SO-118. Asheville, NC: USDA Forest Service, Southern Research Station: 2-6. http://www.treesearch.fs.fed.us/pubs/685

Guldin, J.M.; Cawrse, D.; Graham, R.; Hemstrom, M.; Joyce, L.; Kessler, S.; McNair, R.; Peterson, G.; Shaw, C.G.; Stine, P.; Twery, M.; Walter, J. 2003a. Science consistency reviews: a primer for application. FS-771. Washington, DC: USDA Forest Service. 9 p. https://www.fs.usda.gov/treesearch/pubs/20749

Guldin, J.M.; Cawrse, D.; Graham, R.; Hemstrom, M.; Joyce, L.; Kessler, S.; McNair, R.; Peterson, G.; Shaw, C.G.; Stine, P.; Twery, M.; Walter, J. 2003b. The science consistency review: a tool to evaluate the use of scientific information in land management decision-making. FS-772. Washington, DC: USDA Forest Service, Washington Office. 32 p. https://www.fs.usda.gov/treesearch/pubs/20750

Gunderson, L.H. 2000. Ecological resilience – in theory and application. Annual Review of Ecology and Systematics. 31(1): 425-439. doi:10.1146/annurev.ecolsys.31.1.42

Gunderson, L.H.; Holling, C.S., eds. 2002. Panarchy: understanding transformations in human and natural systems. Washington, DC: Island Press. 507 p. isbn:978-1-55963-857-9

Gunderson, L.H.; Allen, C.R.; Holling, C.S., eds. 2010. Foundations of ecological resilience. Washington, DC: Island Press. 466 p. isbn:978-1-59726-511-9

Gurevitch, J.; Scheiner, S.M.; Fox, G.A. 2006a. Disturbance and succession. Chapter 12. In: Gurevitch, J.; Scheiner, S.M.; Fox, G.A. The ecology of plants. 2nd edition. Sunderland, MA: Sinauer Associates: 283-305. isbn:978-0-87893-294-8

Gurevitch, J.; Scheiner, S.M.; Fox, G.A. 2006b. The ecology of plants. 2nd edition. Sunderland, MA: Sinauer Associates, Inc. 574 p. isbn:978-0-87893-294-8

Gustafsson, L.; Kouki, J.; Sverdrup-Thygeson, A. 2010. Tree retention as a conservation measure in clear-cut forests of northern Europe: a review of ecological consequences. Scandinavian Journal of Forest Research. 25(4): 295-308. doi:10.1080/02827581.2010.497495

Gustafsson, L.; Baker, S.C.; Bauhus, J.; Beese, W.J.; Brodie, A.; Kouki, J.; Lindenmayer, D.B.; Lõhmus, A.; Pastur, G.M.; Messier, C.; Neyland, M.; Palik, B.; Sverdrup-Thygeson, A.; Volney, W.J.A.; Wayne, A.; Franklin, J.F. 2012. Retention forestry to maintain multifunctional forests: A world perspective. BioScience. 62(7): 633-645. doi:10.1525/bio.2012.62.7.6

Gutsell, S.L.; Johnson, E.A. 2007. Wildfire and tree population processes. In: Johnson, E.A.; Miyanishi, K., eds. Plant disturbance ecology: the process and the response. Amsterdam: Academic Press: 441-485. isbn:978-0-12-088778-1

Guyette, R.P.; Stambaugh, M.C.; Dey, D.C.; Muzika, R.-M. 2012. Predicting fire frequency with chemistry and climate. Ecosystems. 15(2): 322-335. doi:10.1007/s10021-011-9512-0

Haack, R.A.; Byler, J.W. 1993. Insects and pathogens: regulators of forest ecosystems. Journal of Forestry. 91(9): 32-37. doi:10.1093/jof/91.9.32

Habeck, J.R. 1976. Forests, fuels and fire in the Selway-Bitterroot Wilderness, Idaho. In: Proceedings of the annual Tall Timbers fire ecology conference number 14. Tallahassee, FL: Tall Timbers Research Station: 305-353. https://talltimbers.org/wp-content/uploads/2018/09/305-Habeck1974_op.pdf

Habeck, J.R. 1987. Present-day vegetation in the northern Rocky Mountains. Annals of the Missouri Botanical Garden. 74(4): 804-840. doi:10.2307/2399451

Habeck, J.R. 1990. Old-growth ponderosa pine–western larch forests in western Montana: ecology and management. Northwest Environmental Journal. 6(2): 271-292.

Habeck, J.R.; Mutch, R.W. 1973. Fire-dependent forests in the northern Rocky Mountains. Quaternary Research. 3(3): 408-424. doi:10.1016/0033-5894(73)90006-9

Hadfield, J.S. 1984. Root disease problems and opportunities in the interior Douglas-fir and grand fir forest types. In: Baumgartner, D.M.; Mitchell, R., comps. Silvicultural management strategies for pests of the interior Douglas-fir and grand fir forest types. Pullman, WA: Washington State University, Cooperative Extension: 59-66.

Hadfield, J.S.; Russell, K.W. 1978. Dwarf mistletoe management in the Pacific Northwest. In: Scharpf, R.F.; Parmeter, Jr., J.R., tech. coords. Proceedings of the symposium on dwarf mistletoe control through forest management. Berkeley, CA: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station: 73-81. https://www.fs.usda.gov/treesearch/pubs/24083

Hadfield, J.S.; Mathiasen, R.L.; Hawksworth, F.G. 2000. Douglas-fir dwarf mistletoe. Forest Insect & Disease Leaflet 54; R6-NR-FID-PR-003-00. [Place of publication unknown]: USDA Forest Service. 9 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043663.pdf

Hadley, K.S.; Veblen, T.T. 1993. Stand response to western spruce budworm and Douglas-fir beetle outbreaks, Colorado Front Range. Canadian Journal of Forest Research. 23(3): 479-491. doi:10.1139/x93-066

Haeussler, S.; Kneeshaw, D. 2003. Comparing forest management to natural processes. Chapter 9. In: Burton, P.J.; Messier, C.; Smith, D.W.; Adamowicz, W.L., eds. Towards sustainable management of the boreal forest. Ottawa, ON: NRC Research Press: 307-368. isbn:0-660-18762-0

Haeussler, S.; Bedford, L.; Boateng, J.O.; MacKinnon, A. 1999. Plant community responses to mechanical site preparation in northern interior British Columbia. Canadian Journal of Forest Research. 29(7): 1084-1100. doi:10.1139/x99-057

Hagle, S.K.; McDonald, G.I.; Norby, E.A. 1989. White pine blister rust in northern Idaho and western Montana: alternatives for integrated management. Gen. Tech. Rep. INT-261. Ogden, UT: USDA Forest Service, Intermountain Research Station. 35 p. https://www.fs.usda.gov/treesearch/pubs/24638

Hagle, S.K., S. Kegley and S.B. Williams. 1995. Assessing pathogen and insect succession functions in forest ecosystems. In: Eskew, L.G., comp. Forest health through silviculture: proceedings of the 1995 National Silviculture Workshop. Gen. Tech. Rep. RM-GTR-267. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO: 117-127. https://www.fs.usda.gov/treesearch/pubs/22037

Hagle, S.K.; Gibson, K.E.; Tunnock, S. 2003. Field guide to diseases and insect pests of northern and central Rocky Mountain conifers. R1-03-08. Missoula, MT: USDA Forest Service, State and Private Forestry, Northern and Intermountain Regions. 197 p. Complete-field-guide

Hagmann, R.K.; Franklin, J.F.; Johnson, K.N. 2013. Historical structure and composition of ponderosa pine and mixed-conifer forests in south-central Oregon. Forest Ecology and Management. 304: 492-504. doi:10.1016/j.foreco.2013.04.005

Hagmann, R.K.; Franklin, J.F.; Johnson, K.N. 2014. Historical conditions in mixed-conifer forests on the eastern slopes of the northern Oregon Cascade Range, USA. Forest Ecology and Management. 330: 158-170. doi:10.1016/j.foreco.2014.06.044

Hagmann, R.K.; Merschel, A.G.; Reilly, M.J. 2019. Historical patterns of fire severity and forest structure and composition in a landscape structured by frequent large fires: Pumice Plateau ecoregion, Oregon, USA. Landscape Ecology. 34(3): 551-568. doi:10.1007/s10980-019-00791-1

Haight, R.G.; Monserud, R.A.; Chew, J.D. 1992. Optimal harvesting with stand density targets: managing Rocky Mountain conifer stands for multiple forest outputs. Forest Science. 38(3): 554-574. doi:10.1093/forestscience/38.3.554

Haines, F. 1938. The northward spread of horses among the plains Indians. American Anthropologist. 40(3): 429-437. doi:10.2307/662040

Haines-Young, R.; Chopping, M. 1996. Quantifying landscape structure: a review of landscape indices and their application to forested landscapes. Progress in Physical Geography. 20(4): 418-445. doi:10.1177/030913339602000403

Haire, S.L.; McGarigal, K. 2009. Changes in fire severity across gradients of climate, fire size, and topography: a landscape ecological perspective. Fire Ecology. 5(2): 86-103. doi:10.4996/fireecology.0502086

Hall, F.C. 1973. Plant communities of the Blue Mountains in eastern Oregon and southeastern Washington. R6 Area Guide 3-1. Portland, OR: USDA Forest Service, Pacific Northwest Region. 62 p. https://archive.org/download/CAT91954348/CAT91954348.pdf

Hall, F.C. 1976. Fire and vegetation in the Blue Mountains – implications for land managers. In: Komarek, E.V., Sr., chair. Proceedings of the annual Tall Timbers fire ecology conference No. 15. Tallahassee, FL: Tall Timbers Research Station: 155-170. https://talltimbers.org/wp-content/uploads/2018/09/155-Hall1974_op.pdf

Hall, F.C. 1980. Fire history – Blue Mountains, Oregon. In: Stokes, M.A.; Dieterich, J.H., tech. coords. Proceedings of the fire history workshop. Gen. Tech. Rep. RM-81. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station: 75-81. https://www.fs.usda.gov/treesearch/pubs/41408

Hall, F.C. 1982. Ecology of grand fir. In: Oliver, C.D.; Kenady, R.M., eds. Proceedings of the biology and management of true fir in the Pacific Northwest symposium. Contribution No. 45. Seattle, WA: University of Washington, College of Forest Resources, Institute of Forest Resources: 43-52.

Hall, F.C. 1984. Vegetation diversity in managed versus unmanaged ecosystems and effects on various resources. In: Cooley, J.L.; Cooley, J.H., eds. Natural diversity in forest ecosystems. Athens, GA: University of Georgia, Institute of Ecology: 251-262.

Hall, F.C. 1991. Ecology of fire in the Blue Mountains of eastern Oregon. Unpub. Rep. Portland, OR, USDA Forest Service, Pacific Northwest Region. 27 p.

Hall, T.E.; Bigler-Cole, H. 2001. Sociocultural factors and forest health management. Northwest Science. 75(Special Issue): 208-233. http://hdl.handle.net/2376/998

Hall, F.C.; Bryant, L.; Clausnitzer, R.; Geier-Hayes, K.; Keane, R.; Kertis, J.; Shlisky, A.; Steele, R. 1995. Definitions and codes for seral status and structure of vegetation. Gen. Tech. Rep. PNW-GTR-363. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 39 p. http://www.treesearch.fs.fed.us/pubs/5619

Halofsky, J.E.; Peterson, D.L. 2017. Climate change vulnerability and adaptation in the Blue Mountains region. Gen. Tech. Rep. PNW-GTR-939. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 331 p. https://www.treesearch.fs.fed.us/pubs/53937

Halofsky, J.E.; Donato, D.C.; Hibbs, D.E.; Campbell, J.L.; Cannon, M.D.; Fontaine, J.B.; Thompson, J.R.; Anthony, R.G.; Bormann, B.T.; Kayes, L.J.; Law, B.E.; Peterson, D.L.; Spies, T.A. 2011. Mixed-severity fire regimes: lessons and hypotheses from the Klamath-Siskiyou Ecoregion. Ecosphere. 2(4): art40. 19 p. doi:10.1890/es10-00184.1

Halpern, C.B. 1988. Early successional pathways and the resistance and resilience of forest communities. Ecology. 69(6): 1703-1715. doi:10.2307/1941148

Halpern, C.B. 1989. Early successional patterns of forest species: interactions of life history traits and disturbance. Ecology. 70(3): 704-720. doi:10.2307/1940221

Halpern, C.B.; Spies, T.A. 1995. Plant species diversity in natural and managed forests of the Pacific Northwest. Ecological Applications. 5(4): 913-934. doi:10.2307/2269343

Hamann, A.; Wang, T. 2006. Potential effects of climate change on ecosystem and tree species distribution in British Columbia. Ecology. 87(11): 2773-2786. doi:10.1890/0012-9658(2006)87[2773:PEOCCO]2.0.CO;2

Hamilton, L.C.; Hartter, J.; Stevens, F.; Congalton, R.G.; Ducey, M.J.; Campbell, M.; Maynard, D.; Staunton, M. 2012. Forest views: Northeast Oregon survey looks at community and environment. Issue Brief No. 47. Durham, NH: University of New Hampshire, Carsey Institute. 12 p. www.carseyinstitute.unh.edu

Hamilton, L.C.; Hartter, J.; Safford, T.G.; Stevens, F.R. 2014. Rural environmental concern: Effects of position, partisanship, and place. Rural Sociology. 79(2): 257-281. doi:10.1111/ruso.12023

Hamilton, L.C.; Hartter, J.; Lemcke-Stampone, M.; Moore, D.W.; Safford, T.G. 2015a. Tracking public beliefs about anthropogenic climate change. PLoS ONE. 10(9): e0138208. doi:10.1371/journal.pone.0138208

Hamilton, L.C.; Hartter, J.; Safford, T.G. 2015b. Validity of county-level estimates of climate change beliefs. Nature Climate Change. 5(8): 704-704. doi:10.1038/nclimate2720

Hamilton, L.C.; Hartter, J.; Keim, B.D.; Boag, A.E.; Palace, M.W.; Stevens, F.R.; Ducey, M.J. 2016. Wildfire, climate, and perceptions in northeast Oregon. Regional Environmental Change. 16(6): 1819-1832. doi:10.1007/s10113-015-0914-y

Hamilton, M.; Fischer, A.P.; Ager, A. 2019. A social-ecological network approach for understanding wildfire risk governance. Global Environmental Change. 54: 113-123. doi:10.1016/j.gloenvcha.2018.11.007

Han, H.-S.; Page-Dumroese, D.; Han, S.-K.; Tirocke, J. 2006. Effects of slash, machine passes, and soil moisture on penetration resistance in a cut-to-length harvesting. International Journal of Forest Engineering. 17(2): 11-24. doi:10.1080/14942119.2006.10702532

Han, S.-K.; Han, H.-S.; Page-Dumroese, D.S.; Johnson, L.R. 2009. Soil compaction associated with cut-to-length and whole-tree harvesting of a coniferous forest. Canadian Journal of Forest Research. 39(5): 976-989. doi:10.1139/X09-027

Hanberry, B.B.; Justice, D.C.; Powell, D.C. 2020. Discovering Douglas-Fir woodlands in the historical forests of Umatilla National Forest, eastern Oregon and Washington. Forests. 11(10): 1122. doi:10.3390/f11101122

Hanewinkel, M.; Breidenbach, J.; Neeff, T.; Kublin, E. 2008. Seventy-seven years of natural disturbances in a mountain forest area—the influence of storm, snow, and insect damage analysed with a long-term time series. Canadian Journal of Forest Research. 38(8): 2249-2261. doi:10.1139/X08-070

Hanewinkel, M.; Hummel, S.; Albrecht, A. 2011. Assessing natural hazards in forestry for risk management: a review. European Journal of Forest Research. 130(3): 329-351. doi:10.1007/s10342-010-0392-1

Haney, A.; Power, R.L. 1996. Adaptive management for sound ecosystem management. Environmental Management. 20(6): 879-886. doi:10.1007/BF01205968

Hankin, L.E.; Higuera, P.E.; Davis, K.T.; Dobrowski, S.Z. 2018. Accuracy of node and bud-scar counts for aging two dominant conifers in western North America. Forest Ecology and Management. 427: 365-371. doi:10.1016/j.foreco.2018.06.001

Hankin, L.E.; Higuera, P.E.; Davis, K.T.; Dobrowski, S.Z. 2019. Impacts of growing-season climate on tree growth and post-fire regeneration in ponderosa pine and Douglas-fir forests. Ecosphere. 10(4): e02679 (20 p). doi:10.1002/ecs2.2679

Hann, W.J.; Wisdom, M.J.; Rowland, M.M. 2003. Disturbance departure and fragmentation of natural systems in the interior Columbia basin. Res. Pap. PNW-RP-545. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 19 p. https://www.fs.usda.gov/treesearch/pubs/5283

Hannah, L.; Costello, C.; Guo, C.; Ries, L.; Kolstad, C.; Panitz, D.; Snider, N. 2011. The impact of climate change on California timberlands. Climatic Change. 109(Supplement 1): 429-443. doi:10.1007/s10584-011-0307-2

Hansen, H.P. 1943. A pollen study of a subalpine bog in the Blue Mountains of northeastern Oregon. Ecology. 24(1): 70-78. doi:10.2307/1929861

Hansen, E.M.; Goheen, E.M. 2000. *Phellinus weirii* and other native root pathogens as determinants of forest structure and process in western North America. Annual Review of Phytopathology. 38: 515-539. doi:10.1146/annurev.phyto.38.1.515

Hansen, L.J.; Hoffman, J.R. 2011. Climate savvy: adapting conservation and resource management to a changing world. Washington, DC: Island Press. 245 p. isbn:978-1-59726-686-4

Hansen, A.J.; Spies, T.A.; Swanson, F.J.; Ohmann, J.L. 1991. Conserving biodiversity in managed forests: lessons from natural forests. BioScience. 41(6): 382-392. doi:10.2307/1311745

Hanson, P.J.; Weltzin, J.F. 2000. Drought disturbance from climate change: response of United States forests. Science of the Total Environment. 262(3): 205-220. doi:10.1016/S0048-9697(00)00523-4

Hard, J.S. 1985. Spruce beetles attack slowly growing spruce. Forest Science. 31(4): 839-850. doi:10.1093/forestscience/31.4.839

Hardy, C.C.; Keane, R.E.; Harrington, M.G. 1999. Restoration of northwest interior forests. In: Transactions of the 64th North American wildlife and natural resources conference. Washington, DC: Wildlife Management Institute: 117-138.

Hardy, C.C.; Keane, R.E.; Stewart, C.A. 2000. Ecosystem-based management in the lodgepole pine zone. In: Smith, H.Y., ed. The Bitterroot ecosystem management research project: What we have learned: symposium proceedings. Proc. RMRS-P-17. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station: 31-35. https://www.fs.usda.gov/treesearch/pubs/21657

Hargrove, W.W.; Gardner, R.H.; Turner, M.G.; Romme, W.H.; Despain, D.G. 2000. Simulating fire patterns in heterogeneous landscapes. Ecological Modelling. 135(2-3): 243-263. doi:10.1016/S0304-3800(00)00368-9

Harlow, W.M.; Harrar, E.S.; Hardin, J.W.; White, F.M. 1996. Textbook of dendrology. 8th edition. New York: McGraw-Hill. 534 p. isbn:0-07-026572-0

Harmon, M.E. 2001. Moving towards a new paradigm for woody detritus management. Ecological Bulletins. 49: 269-278. https://www.jstor.org/stable/20113283

Harmon, M.E.; Franklin, J.F.; Swanson, F.J.; Sollins, P.; Gregory, S.V.; Lattin, J.D.; Anderson, N.H.; Cline, S.P.; Aumen, N.G.; Sedell, J.R.; Lienkaemper, G.W.; Cromack, K., Jr.; Cummins, K.W. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research. 15: 113-302. doi:10.1016/S0065-2504(08)60121-X

Harris, L.D. 1984. The fragmented forest: island biogeography theory and the preservation of biotic diversity. Chicago, IL: University of Chicago Press. 211 p. isbn:0-226-31764-1

Harris, M.K.; Frederiksen, R.A. 1984. Concepts and methods regarding host plant resistance to arthropods and pathogens. Annual Review of Phytopathology. 22(1): 247-272. doi:10.1146/annurev.py.22.090184.001335

Harris, C.C.; McLaughlin, W.J.; Brown, G. 1998. Rural communities in the interior Columbia basin: How resilient are they? Journal of Forestry. 96(3): 11-15. doi:10.1093/jof/96.3.11

Harris, J.A.; Hobbs, R.J.; Higgs, E.; Aronson, J. 2006. Ecological restoration and global climate change. Restoration Ecology. 14(2): 170-176. doi:10.1111/j.1526-100X.2006.00136.x

Hart, S.J.; Veblen, T.T. 2015. Detection of spruce beetle-induced tree mortality using high- and medium-resolution remotely sensed imagery. Remote Sensing of Environment. 168: 134-145. doi:10.1016/j.rse.2015.06.015

Hartsough, B.R.; Drews, E.S.; McNeel, J.F.; Durston, T.A.; Stokes, B.J. 1997. Comparison of mechanized systems for thinning ponderosa pine and mixed conifer stands. Forest Products Journal. 47(11/12): 59-68. Hartsough et al 1997

Hartsough, B.R.; Zhang, X.; Fight, R.D. 2001. Harvesting cost model for small trees in natural stands in the interior Northwest. Forest Products Journal. 51(4): 54-61.

Hartter, J.; Stevens, F.R.; Hamilton, L.C.; Oester, P.T.; Congalton, R.G.; Ducey, M.J.; Crowley, M.A. 2014. Forest management and wildfire risk in inland Northwest: Survey of landowners' concerns about public land management in northeastern Oregon. National Issue Brief #70. Durham, NH: University of New Hampshire, Carsey Institute. 8 p. https://scholars.unh.edu/carsey/211/

Hartter, J.; Stevens, F.R.; Hamilton, L.C.; Congalton, R.G.; Ducey, M.J.; Oester, P.T. 2015. Modelling associations between public understanding, engagement and forest conditions in the Inland Northwest, USA. PLoS ONE. 10(2): e0117975 (25 p). doi:10.1371/journal.pone.0117975

Hartter, J.; Hamilton, L.C.; Ducey, M.J.; Boag, A.E.; Christoffersen, N.D.; Belair, E.P.; Oester, P.T.; Palace, M.W.; Stevens, F.R. 2017. Drier conditions, more wildfire, and heightened concerns about forest management in eastern Oregon. National Issue Brief #127. Durham, NH: University of New Hampshire, Carsey School of Public Policy. 6 p. https://carsey.unh.edu/publication/forest-management-oregon

Harvey, R.D.; Jr. 1979. Rate of increase of blue stained volume in mountain pine beetle killed lodgepole pine in northeastern Oregon, U.S.A. Canadian Journal of Forest Research. 9(3): 323-326. doi:10.1139/x79-055

Harvey, A.E. 1994. Integrated roles for insects, diseases and decomposers in fire dominated forests of the inland western United States: past, present and future forest health. Journal of Sustainable Forestry. 2(1-2): 211-220. doi:10.1300/J091v02n01_10

Harvey, A.E.; Neuenschwander, L.F., comps. 1991. Proceedings–management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: USDA Forest Service, Intermountain Research Station. 254 p. http://www.treesearch.fs.fed.us/pubs/33908

Harvey, A.E.; Larsen, M.J.; Jurgensen, M.F. 1976. Distribution of ectomycorrhizae in a mature Douglas-fir/larch forest soil in western Montana. Forest Science. 22(4): 393-398. doi:10.1093/forestscience/22.4.393

Harvey, A.E.; Jurgensen, M.F.; Larsen, M.J. 1978. Seasonal distribution of ectomycorrhizae in a mature Douglas-fir/larch forest soil in western Montana. Forest Science. 24(2): 203-208. doi:10.1093/forestscience/24.2.203

Harvey, A.E.; Larsen, M.J.; Jurgensen, M.F. 1979. Comparative distribution of ectomycorrhizae in soils of three western Montana forest habitat types. Forest Science. 25(2): 350-358. doi:10.1093/forestscience/25.2.350

Harvey, A.E.; Jurgensen, M.F.; Larsen, M.J. 1980a. Clearcut harvesting and ectomycorrhizae: survival of activity on residual roots and influence on a bordering forest stand in western Montana. Canadian Journal of Forest Research. 10(3): 300-303. doi:10.1139/x80-051

Harvey, A.E.; Larsen, M.J.; Jurgensen, M.F. 1980b. Ecology of ectomycorrhizae in northern Rocky Mountain forests. In: USDA Forest Service. Environmental consequences of timber harvesting in Rocky Mountain coniferous forests. Gen. Tech. Rep. INT-90. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 189-208. https://archive.org/download/CAT80751102/CAT80751102.pdf

Harvey, A.E.; Jurgensen, M.F.; Larsen, M.J. 1981. Organic reserves: importance to ectomycorrhizae in forest soils in western Montana. Forest Science. 27(3): 442-445. doi:10.1093/forestscience/27.3.442

Harvey, A.E.; Jurgensen, M.F.; Larsen, M.J.; Schlieter, J.A. 1986. Distribution of active ectomycorrhizal short roots in forest soils of the inland northwest: effects of site and disturbance. Res. Pap. INT-374. Ogden, UT: USDA Forest Service, Intermountain Research Station. 8 p. https://www.biodiversitylibrary.org/itempdf/136954

Harvey, A.E.; Jurgensen, M.F.; Larsen, M.J.; Graham, R.T. 1987a. Decaying organic materials and soil quality in the inland northwest: a management opportunity. Gen. Tech. Rep. INT-225. Ogden, UT: USDA Forest Service, Intermountain Research Station. 15 p. https://www.fs.usda.gov/treesearch/pubs/39575

Harvey, A.E.; Jurgensen, M.F.; Larsen, M.J. 1987b. Residues, beneficial microbes, diseases, and soil management in cool, east slope, Rocky Mountain lodgepole pine ecosystems. In: Barger, R.L., comp. Management of small-stem stands of lodgepole pine–workshop proceedings. Gen. Tech. Rep. INT-237. Ogden, UT: USDA Forest Service, Intermountain Research Station: 137-149. https://archive.org/download/CAT88893233/CAT88893233.pdf

Harvey, A.E.; Meurisse, R.T.; Geist, J.M.; Jurgensen, M.F.; McDonald, G.I.; Graham, R.T.; Stark, N. 1989. Managing productivity processes in the inland northwest–mixed conifers and pines. In: Perry, D.A.; Meurisse, R.; Thomas, B.; Miller, R.; Boyle, J.; Means, J.; Perry, C.R.; Powers, R.F., eds. Maintaining the long-term productivity of Pacific Northwest forest ecosystems. Portland, OR: Timber Press: 164-184. isbn:0-88192-144-0

Harvey, A.E.; Geist, J.M.; McDonald, G.L; Jurgensen, M.F.; Cochran, P.H.; Zabowski, D.; Meurisse, R.T. 1994a. Biotic and abiotic processes in eastside ecosystems: the effects of management on soil properties, processes, and productivity. Gen. Tech. Rep. PNW-GTR-323. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 71 p. https://www.fs.usda.gov/treesearch/pubs/6286

Harvey, A.E.; McDonald, G.I.; Jurgensen, M.F.; Larsen, M.J. 1994b. Microbes: driving forces for long-term ecological processes in the inland northwest's cedar-hemlock-white pine forests. In: Baumgartner, D.M.; Lotan, J.E.; Tonn, J.R., comps. Interior cedar-hemlock-white pine forests: ecology and management. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 157-163.

Harvey, A.E.; Hessburg, P.F.; Byler, J.W.; McDonald, G.I.; Weatherby, J.C.; Wickman, B.E. 1995. Health declines in western interior forests: symptoms and solutions. In: Everett, R.L.; Baumgartner, D.M., comps. Ecosystem management in western interior forests. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 163-170.

Harvey, A.E.; Graham, R.T.; McDonald, G.I. 1999. Tree species composition change – soil organism interaction: potential effects on nutrient cycling and conservation in interior forests. In: Meurisse, R.T.; Ypsilantis, W.G.; Seybold, C., tech. eds. Proceedings: Pacific

Northwest forest and rangeland soil organism symposium. Gen. Tech. Rep. PNW-GTR-461. Portland, OR: USDA Forest Service, Pacific Northwest Research Station: 137-145. http://www.treesearch.fs.fed.us/pubs/7603

Harvey, A.E.; Graham, R.T.; McDonald, G.I.; Larsen, M.J. 2000. Fire/decay: managing codependent forest processes across the landscape. In: Crossing the millenium: integrating spatial technologies and ecological principles for a new age in fire management: proceedings, the Joint Fire Science Conference and Workshop. Boise, ID: University of Idaho: 179-188.

Harvey, B.D.; Nguyen-Xuan, T.; Bergeron, Y.; Gauthier, S.; Leduc, A. 2003. Forest management planning based on natural disturbance and forest dynamics. Chapter 11. In: Burton, P.J.; Messier, C.; Smith, D.W.; Adamowicz, W.L., eds. Towards sustainable management of the boreal forest. Ottawa, ON: NRC Research Press: 395-432. isbn:0-660-18762-0

Harvey, A.E.; Byler, J.W.; McDonald, G.I.; Neuenschwander, L.F.; Tonn, J.R. 2008. Death of an ecosystem: perspectives on western white pine ecosystems of North America at the end of the twentieth century. Gen. Tech. Rep. RMRS-GTR-208. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 10 p. http://www.treesearch.fs.fed.us/pubs/29617

Harvey, B.J.; Donato, D.C.; Romme, W.H.; Turner, M.G. 2014. Fire severity and tree regeneration following bark beetle outbreaks: the role of outbreak stage and burning conditions. Ecological Applications. 24(7): 1608-1625. doi:10.1890/13-1851.1

Harvey, B.J.; Donato, D.C.; Turner, M.G. 2016. High and dry: post-fire tree seedling establishment in subalpine forests decreases with post-fire drought and large stand-replacing burn patches. Global Ecology and Biogeography. 25(6): 655-669. doi:10.1111/geb.12443

Harvey, B.J.; Andrus, R.A.; Anderson, S.C. 2019. Incorporating biophysical gradients and uncertainty into burn severity maps in a temperate fire-prone forested region. Ecosphere. 10(2): e02600 (20 p). doi:10.1002/ecs2.2600

Hassan, R.; Scholes, R.; Ash, N. 2005. Ecosystems and human well-being: current state and trends, volume 1 (findings of the condition and trends working group of the Millennium Ecosystem Assessment). Washington, DC: Island Press. 917 p. isbn:1-55963-228-3

Haufler, J.B. 1994. An ecological framework for planning for forest health. Journal of Sustainable Forestry. 2(3-4): 307-316. doi:10.1300/J091v02n03_06

Haufler, J.B.; Mehl, C.A.; Roloff, G.J. 1996. Using a coarse-filter approach with species assessment for ecosystem management. Wildlife Society Bulletin. 24(2): 200-208. doi:10.2307/3783108

Haughian, S.R.; Burton, P.J.; Taylor, S.W.; Curry, C. 2012. Expected effects of climate change on forest disturbance regimes in British Columbia. BC Journal of Ecosystems and Management. 13(1): 1-24. http://jem.forrex.org/index.php/jem/article/view/152

Haugo, R.D.; Hall, S.A.; Gray, E.M.; Gonzalez, P.; Bakker, J.D. 2010. Influences of climate, fire, grazing, and logging on woody species composition along an elevation gradient in the eastern Cascades, Washington. Forest Ecology and Management. 260(12): 2204-2213. doi:10.1016/j.foreco.2010.09.021

Haukioja, E. 2005. Plant defenses and population fluctuations of forest defoliators: mechanism-based scenarios. Annales Zoologici Fennici. 42(4): 313-325. https://www.jstor.org/stable/23735879

Hawbaker, T.J.; Radeloff, V.C.; Stewart, S.I.; Hammer, R.B.; Keuler, N.S.; Clayton, M.K. 2013. Human and biophysical influences on fire occurrence in the United States. Ecological Applications. 23(3): 565-582. doi:10.1890/12-1816.1

Hawksworth, F.G.; Dooling, O.J. 1984. Lodgepole pine dwarf mistletoe. Forest Insect & Disease Leaflet 18. [Place of publication unknown]: USDA Forest Service. 11 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043427.pdf

Hawksworth, F.G.; Wiens, D. 1996. Dwarf mistletoes: biology, pathology, and systematics. Agricul. Handb. 709. Washington, DC: USDA Forest Service. 410 p. http://www.treesearch.fs.fed.us/pubs/4699

Hayes, J.P. 1980. Old growth forest wildlife habitat on the Umatilla National Forest. Unpublished report. Pendleton, OR: USDA Forest Service, Umatilla National Forest. 41 p.

Hayes, J.L.; Daterman, G.E. 2001. Bark beetles (Scolytidae) in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 21-30. http://hdl.handle.net/2376/977

Hayes, J.L.; Ragenovich, I. 2001. Non-native invasive forest insects of eastern Oregon and Washington. Northwest Science. 75(Special Issue): 77-84. http://hdl.handle.net/2376/984

Haynes, R.W.; Quigley, T.M. 2001. Broad-scale consequences of land management: Columbia basin example. Forest Ecology and Management. 153(1-3): 179-188. doi:10.1016/S0378-1127(01)00460-1

Haynes, R.W.; Graham, R.T.; Quigley, T.M., tech. eds. 1996. A framework for ecosystem management in the interior Columbia Basin including portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-374. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 68 p. http://www.treesearch.fs.fed.us/pubs/3060

Haynes, R.; Fight, R.; Lowell, E.; Stevens, J.; Barbour, J. 2001a. Economic aspects of thinning and harvest for forest health improvement in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 199-207. http://hdl.handle.net/2376/997

Haynes, R.W.; Quigley, T.M.; Clifford, J.L.; Gravenmier, R.A. 2001b. Science and ecosystem management in the interior Columbia basin. Forest Ecology and Management. 153(1-3): 3-14. doi:10.1016/S0378-1127(01)00450-9

Healey, S.P.; Raymond, C.L.; Lockman, I.B.; Hernandez, A.J.; Garrard, C.; Huang, C. 2016. Root disease can rival fire and harvest in reducing forest carbon storage. Ecosphere. 7(11): e01569 (16 p). doi:10.1002/ecs2.1569

Heath, R.; Alfaro, R.I. 1990. Growth response in a Douglas-fir/lodgepole pine stand after thinning of lodgepole pine by the mountain pine beetle: a case study. Journal of the Entomological Society of British Columbia. 87: 16-21. https://journal.entsocbc.ca/index.php/journal/article/download/657/666

Heavilin, J.; Powell, J.; Logan, J.A. 2007. Dynamics of mountain pine beetle outbreaks. In: Johnson, E.A.; Miyanishi, K., eds. Plant disturbance ecology: the process and the response. Amsterdam: Academic Press: 527-553. isbn:978-0-12-088778-1

Hedden, R.L.; Barras, S.J.; Coster, J.E., eds. 1981. Hazard-rating systems in forest pest management: symposium proceedings. Gen. Tech. Rep. WO-27. Washington, DC: USDA Forest Service. 169 p. https://archive.org/download/CAT91960247/CAT91960247.pdf

Hedrick, D.W.; Young, J.A.; McArthur, J.A.B.; Keniston, R.F. 1968. Effects of forest and grazing practices on mixed coniferous forests of northeastern Oregon. Tech. Bull. 103. Corvallis, OR: Oregon State University, Agricultural Experiment Station. 24 p. https://ir.library.oregonstate.edu/downloads/wp988m05s

Hedwall, S.J.; Mathiasen, R.L. 2006. Wildlife use of Douglas-fir dwarf mistletoe witches' brooms in the southwest. Western North American Naturalist. 66(4): 450-455. doi:10.3398/1527-0904(2006)66[450:WUODDM]2.0.CO;2

Heil, L.J.; Burkle, L.A. 2019. The effects of post-wildfire salvage logging on plant reproductive success and pollination in *Symphoricarpos albus*, a fire-tolerant shrub. Forest Ecology and Management. 432: 157-163. doi:10.1016/j.foreco.2018.09.013

Heinonen, T.; Pukkala, T.; Ikonen, V.-P.; Peltola, H.; Venäläinen, A.; Dupont, S. 2009. Integrating the risk of wind damage into forest planning. Forest Ecology and Management. 258(7): 1567-1577. doi:10.1016/j.foreco.2009.07.006

Heinselman, M.L. 1981. Fire intensity and frequency as factors in the distribution and structure of northern ecosystems. In: Mooney, H.A.; Bonnicksen, T.M.; Christensen, N.L.; Lotan, J.E.; Reiners, W.A., tech. coords. Fire regimes and ecosystem properties; proceedings of the conference. Gen. Tech. Rep. WO-26. [Washington, DC]: USDA Forest Service: 7-57. https://archive.org/download/CAT83781017/CAT83781017.pdf

Heinselman, M.L.; Wright, H.E., Jr. 1973. The ecological role of fire in natural conifer forests of western and northern North America. Quaternary Research. 3(3): 317-318. doi:10.1016/0033-5894(73)90001-X

Heintzleman, B.F. 1913. North slope sub-type of the Blue Mountains. Annual Silvical Rep. Sumpter, OR: USDA Forest Service, Whitman National Forest. Unpublished report obtained from National Archives, College Park, Maryland; record group 95. 10 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015535.pdf

Heller, N.E.; Zavaleta, E.S. 2009. Biodiversity management in the face of climate change: A review of 22 years of recommendations. Biological Conservation. 142(1): 14-32. doi:10.1016/j.biocon.2008.10.006

Helms, J.A., ed. 1998. The dictionary of forestry. Bethesda, MD: Society of American Foresters. 210 p. isbn:0-939970-73-2

Hemstrom, M.A. 2001. Vegetative patterns, disturbances, and forest health in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 91-109. http://hdl.handle.net/2376/986

Hemstrom, M.A.; Korol, J.J.; Hann, W.J. 2001. Trends in terrestrial plant communities and landscape health indicate the effects of alternative management strategies in the interior Columbia River basin. Forest Ecology and Management. 153(1-3): 105-125. doi:10.1016/S0378-1127(01)00456-X

Hendrickson, O.Q.; Burgess, D. 1989. Nitrogen-fixing plants in a cut-over lodgepole pine stand of southern British Columbia. Canadian Journal of Forest Research. 19(7): 936-939. doi:10.1139/x89-143

Hendrickson, O.Q.; Chatarpaul, L.; Robinson, J.B. 1985. Effects of two methods of timber harvesting on microbial processes in forest soil. Soil Science Society of America Journal. 49(3): 739-746. doi:10.2136/sssaj1985.03615995004900030041x

Hennebelle, A.; Grondin, P.; Aleman, J.C.; Ali, A.A.; Bergeron, Y.; Borcard, D.; Blarquez, O. 2018. Using paleoecology to improve reference conditions for ecosystem-based management in western spruce-moss subdomain of Québec. Forest Ecology and Management. 430: 157-165. doi:10.1016/j.foreco.2018.08.007

Hennon, P.E. 1995. Are heart rot fungi major factors of disturbance in gap-dynamic forests? Northwest Science. 69(4): 284-292. http://hdl.handle.net/2376/1307

Hermann, R.K.; Chilcote, W.W. 1965. Effect of seedbeds on germination and survival of Douglas-fir. Res. Paper 4. Corvallis, OR: Oregon State University, Forest Research Laboratory. 30 p. https://ir.library.oregonstate.edu/concern/technical_reports/qb98mg772

Hermann, R.K.; Lavender, D.P. 1990. *Pseudotsuga menziesii* (Mirb.) Franco; Douglas-fir. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America; volume 1, conifers. Agric. Handb. 654. Washington, DC: USDA Forest Service: 527-540. https://www.treesearch.fs.fed.us/pubs/1547

Hertert, H.D.; Miller, D.L.; Partridge, A.D. 1975. Interaction of bark beetles (Coleoptera:Scolytidae) and root-rot pathogens in grand fir in northern Idaho. Canadian Entomologist 107(8): 899-904. http://pubservices.nrc-cnrc.ca/rp-ps/absres.jsp?jcode=ent&ftl=Ent107899-8&lang=eng

Hess, J.J.; Malilay, J.N.; Parkinson, A.J. 2008. Climate change: The importance of place. American Journal of Preventive Medicine. 35(5): 468-478. doi:10.1016/j.amepre.2008.08.024

Hessburg, P.F.; Agee, J.K. 2003. An environmental narrative of inland northwest United States forests, 1800-2000. Forest Ecology and Management. 178(1-2): 23-59. doi:10.1016/S0378-1127(03)00052-5

Hessburg, P.F.; Everett, R.L. 1991. Forest pathogens and insects as agents of disturbance and recovery in fire-restricted forests of eastern Oregon and Washington. In: Proceedings of the 1991 Annual Meeting of the American Phytopathological Society, St. Louis, MO, August 19, 1991 ("Effects of Pathogens on Ecological Processes in Natural Forests" session). 22 p.

Hessburg, P.F.; Everett, R.L. 1994. Managing landscape patterns and insect and disease disturbances on national forests of the interior Northwest. In: Proceedings of the Society of American Foresters National Convention. Bethesda, MD: Society of American Foresters: 123-128.

Hessburg, P.F.; Smith, B.G. 1999. Management implications of recent changes in spatial patterns of interior northwest forests. In: Transactions of the 64th North American Wildlife and Natural Resources Conference. Washington, DC: Wildlife Management Institute: 55-78. http://www.fs.fed.us/pnw/pubs/journals/pnw_1999_hessburg001.pdf

Hessburg, P.F.; Mitchell, R.G.; Filip, G.M. 1994. Historical and current roles of insects and pathogens in eastern Oregon and Washington forested landscapes. Gen. Tech. Rep. PNW-GTR-327. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 72 p. http://www.treesearch.fs.fed.us/pubs/6390

Hessburg, P.F.; Smith, B.G.; Kreiter, S.D.; Miller, C.A.; Salter, R.B.; McNicholl, C.H.; Hann, W.J. 1999a. Historical and current forest and range landscapes in the interior Columbia River basin and portions of the Klamath and Great basins. Part 1: linking vegetation patterns and landscape vulnerability to potential insect and pathogen disturbances. Gen. Tech. Rep. PNW-GTR-458. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 357 p. http://www.treesearch.fs.fed.us/pubs/29638

Hessburg, P.F.; Smith, B.G; Salter, R.B. 1999b. Detecting change in forest spatial patterns from reference conditions. Ecological Applications. 9(4): 1232-1252. doi:10.1890/1051-0761(1999)009[1232:DCIFSP]2.0.CO;2

Hessburg, P.F.; Smith, B.G.; Salter, R.B. 1999c. Using estimates of natural variation to detect ecologically important change in forest spatial patterns: a case study, Cascade Range, eastern Washington. Res. Pap. PNW-RP-514. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 65 p. https://www.fs.usda.gov/treesearch/pubs/2910

Hessburg, P.F.; Smith, B.G.; Salter, R.B.; Ottmar, R.D.; Alvarado, E. 2000. Recent changes (1930s-1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management. 136(1-3): 53-83. doi:10.1016/S0378-1127(99)00263-7

Hessburg, P.F.; Reynolds, K.M.; Salter, R.B.; Richmond, M.B. 2004. Using a decision support system to estimate departures of present forest landscape patterns from historical reference conditions. In: Perera, A.H.; Buse, L.J.; Weber, M.G., eds. Emulating natural forest landscape disturbances: concepts and applications. New York: Columbia University Press: 158-175. isbn:0-231-12916-5

Hessburg, P.F.; Salter, R.B.; James, K.M. 2005. Evidence for mixed severity fires in pre-management era dry forests of the inland northwest, USA. In: Taylor, L.; Zelnik, J.; Cadwallader, S.; Hughes, B., eds. Mixed severity fire regimes: ecology and management. AFE MISC03. Pullman, WA: Washington State University Extension: 89-104.

Hessburg, P.; Salter, R.; James, K. 2007. Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology. 22(Supplement 1): 5-24. doi:10.1007/s10980-007-9098-2

Hessburg, P.F.; Povak, N.A.; Salter, R.B. 2008. Thinning and prescribed fire effects on dwarf mistletoe severity in an eastern Cascade Range dry forest, Washington. Forest Ecology and Management. 255(7): 2907-2915. doi:10.1016/j.foreco.2008.01.066

Hessburg, P.; Reynolds, K.; Salter, R.; Dickinson, J.; Gaines, W.; Harrod, R. 2013. Landscape evaluation for restoration planning on the Okanogan-Wenatchee National Forest, USA. Sustainability. 5(3): 805-840. doi:10.3390/su5030805

Hessburg, P.F.; Miller, C.L.; Parks, S.A.; Povak, N.A.; Taylor, A.H.; Higuera, P.E.; Prichard, S.J.; North, M.P.; Collins, B.M.; Hurteau, M.D.; Larson, A.J.; Allen, C.D.; Stephens, S.L.; Rivera-Huerta, H.; Stevens-Rumann, C.S.; Daniels, L.D.; Gedalof, Z.e.; Gray, R.W.; Kane, V.R.; Churchill, D.J.; Hagmann, R.K.; Spies, T.A.; Cansler, C.A.; Belote, R.T.; Veblen, T.T.; Battaglia, M.A.; Hoffman, C.; Skinner, C.N.; Safford, H.D.; Salter, R.B. 2019. Climate, environment, and disturbance history govern resilience of western North American forests. Frontiers in Ecology and Evolution. 7: 239 (27 p). doi:10.3389/fevo.2019.00239

Hessburg, P.F.; Charnley, S.; Wendel, K.L.; White, E.M.; Singleton, P.H.; Peterson, D.W.; Halofsky, J.E.; Gray, A.N.; Spies, T.A.; Flitcroft, R.L.; White, R. 2020. The 1994 Eastside Screens large-tree harvest limit: review of science relevant to forest planning 25 years later. Gen. Tech. Rep. PNW-GTR-990. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 114 p. https://www.fs.usda.gov/treesearch/pubs/60635

Hessl, A. 2002. Aspen, elk, and fire: the effects of human institutions on ecosystem processes. BioScience. 52(11): 1011-1022. doi:10.1641/0006-3568(2002)052[1011:AEAFTE]2.0.CO;2

Hessl, A.E.; McKenzie, D.; Schellhaas, R. 2004. Drought and Pacific Decadal Oscillation linked to fire occurrence in the inland Pacific Northwest. Ecological Applications. 14(2): 425-442. doi:10.1890/03-5019

Heyerdahl, E.K. 1997. Spatial and temporal variation in historical fire regimes of the Blue Mountains, Oregon and Washington: the influence of climate. Ph.D. dissertation. Seattle, WA: University of Washington, College of Forest Resources. 224 p. http://hdl.handle.net/1773/5575

Heyerdahl, E.K.; Agee, J.K. 1996. Historical fire regimes of four sites in the Blue Mountains, Oregon and Washington. Final Rep. Seattle, WA: University of Washington, College of Forest Resources. 173 p.

Heyerdahl, E.K.; Brubaker, L.B.; Agee, J.K. 2001. Spatial controls of historical fire regimes: a multiscale example from the interior west, USA. Ecology. 82(3): 660-678. doi:10.1890/0012-9658(2001)082[0660:SCOHFR]2.0.CO;2

Heyerdahl, E.K.; Brubaker, L.B.; Agee, J.K. 2002. Annual and decadal climate forcing of historical fire regimes in the interior Pacific Northwest, USA. Holocene. 12(5): 597-604. doi:10.1191/0959683602hl570rp

Heyerdahl, E.K.; McKenzie, D.; Daniels, L.D.; Hessl, A.E.; Littell, J.S.; Mantua, N.J. 2008. Climate drivers of regionally synchronous fires in the inland Northwest (1651-1900). International Journal of Wildland Fire. 17(1): 40-49. doi:10.1071/WF07024

Heyerdahl, E.K.; Loehman, R.A.; Falk, D.A. 2018. A multi-century history of fire regimes along a transect of mixed-conifer forests in central Oregon, U.S.A. Canadian Journal of Forest Research. 49(1): 76-86. doi:10.1139/cjfr-2018-0193

Hicke, J.A.; Jenkins, J.C. 2008. Mapping lodgepole pine stand structure susceptibility to mountain pine beetle attack across the western United States. Forest Ecology and Management. 255(5-6): 1536-1547. doi:10.1016/j.foreco.2007.11.027

Hicke, J.A.; Logan, J.A.; Powell, J.; Ojima, D.S. 2006. Changing temperatures influence suitability for modeled mountain pine beetle (*Dendroctonus ponderosae*) outbreaks in the western United States. Journal of Geophysical Research. 111(G2): G02019 (12 p). doi:10.1029/2005jg000101

Hicke, J.A.; Johnson, M.C.; Hayes, J.L.; Preisler, H.K. 2012. Effects of bark beetle-caused tree mortality on wildfire. Forest Ecology and Management. 271: 81-90.

doi:10.1016/j.foreco.2012.02.005

Hicke, J.A.; Xu, B.; Meddens, A.J.H.; Egan, J.M. 2020. Characterizing recent bark beetle-caused tree mortality in the western United States from aerial surveys. Forest Ecology and Management. 475: 118402 (13 p). doi:10.1016/j.foreco.2020.118402

Higgs, E.S. 1997. What is good ecological restoration? Conservation Biology. 11(2): 338-348. doi:10.1046/j.1523-1739.1997.95311.x

Highley, T.L.; Kirk, T.K. 1979. Mechanisms of wood decay and the unique features of heartrots. Phytopathology. 69(10): 1151-1157. doi:10.1094/Phyto-69-1151

Hilborn, R.; Walters, C.J.; Ludwig, D. 1995. Sustainable exploitation of renewable resources. Annual Review of Ecology and Systematics. 26(1): 45-67. doi:10.1146/annurev.es.26.110195.000401

Hilimire, K.; Nesmith, J.C.B.; Caprio, A.C.; Milne, R. 2013. Attributes of windthrown trees in a Sierra Nevada mixed-conifer forest. Western Journal of Applied Forestry. 28(2): 85-88. doi:10.5849/wjaf.12-027

Hill, E.M.; Ex, S. 2020. Microsite conditions in a low-elevation Engelmann spruce forest favor ponderosa pine establishment during drought conditions. Forest Ecology and Management. 463: 118037. doi:10.1016/j.foreco.2020.118037

Hill, E.; Ex, S.; Aldridge, C.; Prolic, C. 2018. Overstory density, short growing seasons, and moisture limit Engelmann spruce establishment over time in high-elevation managed stands. Canadian Journal of Forest Research. 49(1): 64-75. doi:10.1139/cjfr-2018-0268

Hinderliter, J.M.; Carson, R.J.; Jordan, B.T. 2006. The Big Sink: A large landslide in Oregon's Blue Mountains. Eastern Oregon Science Journal. 19: 52-60. https://drive.google.com/file/d/0B-C4DHYzXM1LNVNXTnhONDVQMHM/view

Hindmarch, T.D.; Reid, M.L. 2001. Thinning of mature lodgepole pine stands increases scolytid bark beetle abundance and diversity. Canadian Journal of Forest Research. 31(9): 1502-1512. doi:10.1139/x01-083

Hinds, T.E. 1985. Diseases. In: DeByle, N.V.; Winokur, R.P., eds. Aspen: ecology and management in the western United States. Gen. Tech. Rep. RM-119. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station: 87-106. https://www.fs.usda.gov/treesearch/pubs/27785

Hirsch, K.; Kafka, V.; Tymstra, C.; McAlpine, R.; Hawkes, B.; Stegehuis, H.; Quintilio, S.; Gauthier, S.; Peck, K. 2001. Fire-smart forest management: a pragmatic approach to sustainable forest management in fire-dominated ecosystems. Forestry Chronicle. 77(2): 357-363. doi:10.5558/tfc77357-2

Hoff, R.J. 1992. Susceptibility of lodgepole pine to western gall rust within the middle Columbia River system. Res. Note INT-401. Ogden, UT: USDA Forest Service, Intermountain Research Station. 8 p. Susceptibility_of_Lodgepole_Pine_to_West

Hoffman, G.R. 1960. The small mammal components of six climax plant associations in eastern Washington and northern Idaho. Ecology. 41(3): 571-572. doi:10.2307/1933338

Hoffman, R.E.; Twery, M.J.; Alban, L.M.; Nyland, R.D. 1999. Forests and people. Pub. NE-INF-138-99. Burlington, VT: USDA Forest Service, Northeastern Research Station, Aiken Forestry Sciences Laboratory. 24 p. https://www.fs.usda.gov/treesearch/pubs/4032

Hoffman, C.; Mathiasen, R.; Sieg, C.H. 2007. Dwarf mistletoe effects on fuel loadings in ponderosa pine forests in northern Arizona. Canadian Journal of Forest Research. 37(3): 662-670. doi:10.1139/X06-259

Hoffman, C.; Morgan, P.; Mell, W.; Parsons, R.; Strand, E.K.; Cook S. 2012. Numerical simulation of crown fire hazard immediately after bark beetle-caused mortality in lodgepole pine forests. Forest Science. 58(2): 178-188. doi:10.5849/forsci.10-137

Hoffman, C.M.; Morgan, P.; Mell, W.; Parsons, R.; Strand, E.; Cook, S. 2013. Surface fire intensity influences simulated crown fire behavior in lodgepole pine forests with recent mountain pine beetle-caused tree mortality. Forest Science. 59(4): 390-399. doi:10.5849/forsci.11-114

Hofmann, J.V. 1917. Natural reproduction from seed stored in the forest floor. Journal of Agricultural Research. 11(1): 1-26. Hofmann 1917 Natural Reproduction

Holdenrieder, O.; Pautasso, M.; Weisberg, P.J.; Lonsdale, D. 2004. Tree diseases and landscape processes: the challenge of landscape pathology. Trends in Ecology and Evolution. 19(8): 446-452. doi:10.1016/j.tree.2004.06.003

Holl, K.D.; Aide, T.M. 2011. When and where to actively restore ecosystems? Forest Ecology and Management. 261(10): 1558-1563. doi:10.1016/j.foreco.2010.07.004

Hollenbeck, J.P.; Ripple, W.J. 2007. Aspen and conifer heterogeneity effects on bird diversity in the northern Yellowstone ecosystem. Western North American Naturalist. 67(1): 92-101. doi:10.3398/1527-0904(2007)67[92:AACHEO]2.0.CO;2

Holling, C.S. 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics. 4: 1-23. doi:10.1146/annurev.es.04.110173.000245

Holling, C.S. 1981. Forest insects, forest fires, and resilience. In: Mooney, H.A.; Bonnicksen, T.M.; Christensen, N.L.; Lotan, J.E.; Reiners, W.A., tech. coords. Fire regimes and ecosystem properties; proceedings of the conference. Gen. Tech. Rep. WO-26. [Washington, DC]: USDA Forest Service: 445-464. https://archive.org/download/CAT83781017/CAT83781017.pdf

Holling, C.S. 1992. Cross-scale morphology, geometry, and dynamics of ecosystems. Ecological Monographs. 62(4): 447-502. doi:10.2307/2937313

Holling, C.S. 1996. Surprise for science, resilience for ecosystems, and incentives for people. Ecological Applications. 6(3): 733-735. doi:10.2307/2269475

Holling, C.S. 2001. Understanding the complexity of economic, ecological, and social systems. Ecosystems. 4(5): 390-405. doi:10.1007/s10021-001-0101-5

Holling, C.S.; Meffe, G.K. 1996. Command and control and the pathology of natural resource management. Conservation Biology. 10(2): 328-337. doi:10.1046/j.1523-1739.1996.10020328.x

Holmgren, M.; Scheffer, M.; Huston, M.A. 1997. The interplay of facilitation and competition in plant communities. Ecology. 78(7): 1966-1975. doi:10.1890/0012-9658(1997)078[1966:TIOFAC]2.0.CO;2

Holsten, E.H.; Thier, R.W.; Munson, A.S.; Gibson, K.E. 1999. The spruce beetle. Forest Insect & Disease Leaflet 127. [Place of publication unknown]: USDA Forest Service. 12 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043099.pdf

Homann, P.S., Harmon, M., Remillard, S., Smithwick, E.A.H. 2005. What the soil reveals: potential total ecosystem C stores of the Pacific Northwest region, USA. Forest Ecology and Management. 220(1-3): 270-283. doi:10.1016/j.foreco.2005.08.035

Hood, S.; Bentz, B.; Gibson, K.; Ryan, K.; DeNitto, G. 2007. Assessing post-fire Douglas-fir mortality and Douglas-fir beetle attacks in the northern Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-199. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 31 p. http://www.treesearch.fs.fed.us/pubs/28511

Hooper, P.R. 1982. The Columbia River basalts. Science. 215(4539): 1463-1468. doi:10.1126/science.215.4539.1463

Hoover, C.M.; Bush, R.; Palmer, M.; Treasure, E. 2020. Using Forest Inventory and Analysis data to support national forest management: Regional case studies. Journal of Forestry. 118(3): 313-323. doi:10.1093/jofore/fvz073

Hopkins, T.; Larson, A.J.; Belote, R.T. 2014. Contrasting effects of wildfire and ecological restoration in old-growth western larch forests. Forest Science. 60(5): 1005-1013. doi:10.5849/forsci.13-088

Horák, J.; Brestovanská, T.; Mladenović, S.; Kout, J.; Bogusch, P.; Halda, J.P.; Zasadil, P. 2019. Green desert?: Biodiversity patterns in forest plantations. Forest Ecology and Management. 433: 343-348. doi:10.1016/j.foreco.2018.11.019

Horn, H.S.; Nathan, R.A.N.; Kaplan, S.R. 2001. Long-distance dispersal of tree seeds by wind. Ecological Research. 16(5): 877-885. doi:10.1046/j.1440-1703.2001.00456.x

Horning, N.; Robinson, J.A.; Sterling, E.J.; Turner, W.; Spector, S. 2010. Remote sensing for ecology and conservation: a handbook of techniques. Oxford, UK: Oxford University Press. 467 p. isbn:978-0-19-921995-7

Hossain, K.L.; Comeau, P.G. 2019. Characterizing light across a strip shelterwood in a mixed conifer forest. Forest Ecology and Management. 445: 134-145. doi:10.1016/j.foreco.2019.05.015

Houtman, R.M.; Montgomery, C.A.; Gagnon, A.R.; Calkin, D.E.; Dietterich, T.G.; McGregor, S.; Crowley, M. 2013. Allowing a wildfire to burn: estimating the effect on future fire suppression costs. International Journal of Wildland Fire. 22(7): 871-882. doi:10.1071/WF12157

Howard, J.L. 1996. *Populus tremuloides*. In: Fire Effects Information System [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us/database/feis/plants/tree/poptre/all.html

Howard, J.L.; Aleksoff, K.C. 2000. *Abies grandis*. In: Fire Effects Information System [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us /database/feis/plants/tree/abigra/all.html

Howe, A.A.; Landhäusser, S.M.; Burney, O.T.; Long, J.N.; Violett, R.D.; Mock, K.E. 2020. Exploring seedling-based aspen (*Populus tremuloides*) restoration near range limits in the Intermountain West, USA. Forest Ecology and Management. 476: 118470 (10 p). doi:10.1016/j.foreco.2020.118470

Huberman, M.A. 1935. The role of western white pine in forest succession in northern Idaho. Ecology. 16(2): 137-151. doi:10.2307/1932421

Hubert, E.E. 1918. Fungi as contributory causes of windfall in the Northwest. Journal of Forestry. 16(6): 696-714. doi:10.1093/jof/16.6.696

Hubert, E.E. 1955. Decay – a problem in the future management of grand fir. Journal of Forestry. 53(6): 409-411. doi:10.1093/jof/53.6.409

Hudec, J.L.; Peterson, D.L. 2012. Fuel variability following wildfire in forests with mixed severity fire regimes, Cascade Range, USA. Forest Ecology and Management. 277: 11-24. doi:10.1016/j.foreco.2012.04.008

Huff, M.H.; Ottmar, R.D.; Alvarado, E.; Everett, R.L.; Vihnanek, R.E.; Lehmkuhl, J.F.; Hessburg, P.F. 1995. Historical and current forest landscapes in eastern Oregon and Washington; part II: linking vegetation characteristics to potential fire behavior and related smoke production. Gen. Tech. Rep. PNW-GTR-355. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 43 p. http://www.treesearch.fs.fed.us/pubs/3063

Huff, D.D.; Hargrove, B.; Tharp, M.L.; Graham, R. 2000. Managing forests for water yield: the importance of scale. Journal of Forestry. 98(12): 15-19. doi:10.1093/jof/98.12.15

Huisinga, K.D.; Laughlin, D.C.; Fulé, P.Z.; Springer, J.D.; McGlone, C.M. 2005. Effects of an intense prescribed fire on understory vegetation in a mixed conifer forest. Journal of the Torrey Botanical Society. 132(4): 590-601. http://www.jstor.org/stable/20063804

Hull, R.B.; Gobster, P.H. 2000. Restoring forest ecosystems: the human dimension. Journal of Forestry. 98(8): 32-36. doi:10.1093/jof/98.8.32

Hull, I.T.; Shipley, L.A.; Berry, S.L.; Loggers, C.; Johnson, T.R. 2020. Effects of fuel reduction timber harvests on forage resources for deer in northeastern Washington. Forest Ecology and Management. 458: 117757 (13 p). doi:10.1016/j.foreco.2019.117757

Hummel, S.; Agee, J.K. 2003. Western spruce budworm defoliation effects on forest structure and potential fire behavior. Northwest Science. 77(2): 159-169. http://hdl.handle.net/2376/807

Humphrey, R.R. 1943. A history of range use and its relation to soil and water losses on the Walla Walla River watershed, Washington and Oregon. Northwest Science. 17: 82-87.

Hungerford, R.D.; Harrington, M.G.; Frandsen, W.H.; Ryan, K.C.; Niehoff, G.J. 1991. Influence of fire on factors that affect site productivity. In: Harvey, A.E.; Neuenschwander, L.F., comps. Proceedings–management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: USDA Forest Service, Intermountain Research Station: 32-50. http://www.treesearch.fs.fed.us/pubs/33908

Hunn, E.S. 1981. On the relative contribution of men and women to subsistence among hunter-gatherers of the Columbia plateau: a comparison with *Ethnographic Atlas* summaries. Journal of Ethnobiology. 1(1): 124-134. http://ethnobiology.org/sites/default/files/pdfs/JoE/1-1/Hunn1981.pdf

Hunt, R.S.; Geils, B.W.; Hummer, K.E. 2010. White pines, Ribes, and blister rust: integration and action. Forest Pathology. 40(3-4): 402-417. doi:10.1111/j.1439-0329.2010.00665.x

Hunter, M.L. 1990. Wildlife, forests, and forestry: principles of managing forests for biological diversity. Englewood Cliffs, NJ: Prentice Hall. 370 p. isbn:0-13-959479-5

Hunter, M.L. 1991. Coping with ignorance: the coarse-filter strategy for maintaining biodiversity. In: Kohm, K., ed. Balancing on the brink of extinction: the Endangered Species Act and lessons for the future. Washington, DC: Island Press: 266-281.

Hunter, M.L. 1993. Natural fire regimes as spatial models for managing boreal forests. Biological Conservation. 65(2): 115-120. doi:10.1016/0006-3207(93)90440-C

Hunter, M., Jr. 1996. Benchmarks for managing ecosystems: are human activities natural? Conservation Biology. 10(3): 695-697. doi:10.1046/j.1523-1739.1996.10030695.x

Hunter, M.L., ed. 1999. Maintaining biodiversity in forest ecosystems. Cambridge, UK: Cambridge University Press. 550 p. isbn:0-521-63768-6

Hunter, M.E.; Robles, M.D. 2020. Tamm review: The effects of prescribed fire on wildfire regimes and impacts: A framework for comparison. Forest Ecology and Management. 475: 118435 (13 p). doi:10.1016/j.foreco.2020.118435

Hunter, M.L., Jr.; White, A.S. 1997. Ecological thresholds and the definition of old-growth forest stands. Natural Areas Journal. 17(4): 292-296. https://www.jstor.org/stable/43911699

Hunter, M.L., Jr.; Jacobson, G.L., Jr.; Webb, T., III. 1988. Paleoecology and the coarse-filter approach to maintaining biological diversity. Conservation Biology. 2(4): 375-385. doi:10.1111/j.1523-1739.1988.tb00202.x

Huntington, J.L.; Hegewisch, K.C.; Daudert, B.; Morton, C.G.; Abatzoglou, J.T.; McEvoy, D.J.; Erickson, T. 2017. Climate engine: Cloud computing and visualization of climate and remote sensing data for advanced natural resource monitoring and process understanding. Bulletin of the American Meteorological Society. 98(11): 2397-2410. doi:10.1175/bams-d-15-00324.1

Huntly, N.; Inouye, R. 1988. Pocket gophers in ecosystems: Patterns and mechanisms. BioScience. 38(11): 786-793. doi:10.2307/1310788

Huntzinger, M. 2003. Effects of fire management practices on butterfly diversity in the forested western United States. Biological Conservation. 113(1): 1-12. doi:10.1016/S0006-3207(02)00356-7

Hurteau, M.; North, M. 2008. Mixed-conifer understory response to climate change, nitrogen, and fire. Global Change Biology. 14(7): 1543-1552. doi:10.1111/j.1365-2486.2008.01584.x

Hurteau, M.; North, M. 2009. Fuel treatment effects on tree-based forest carbon storage and emissions under modeled wildfire scenarios. Frontiers in Ecology and the Environment. 7(8): 409-414. doi:10.1890/080049

Hurteau, M.D.; North, M. 2010. Carbon recovery rates following different wildfire risk mitigation treatments. Forest Ecology and Management. 260(5): 930-937. doi:10.1016/j.foreco.2010.06.015

Huston, M. 1979. A general hypothesis of species diversity. American Naturalist. 113(1): 81-101. https://www.jstor.org/stable/2459944

Hutto, R.L. 2008. The ecological importance of severe wildfires: some like it hot. Ecological Applications. 18(8): 1827-1834. doi:10.1890/08-0895.1

Hutto, R.L.; Belote, R.T. 2013. Distinguishing four types of monitoring based on the questions they address. Forest Ecology and Management. 289: 183-189. doi:10.1016/j.foreco.2012.10.005

Hyde, K.; Dickinson, M.B.; Bohrer, G.; Calkin, D.; Evers, L.; Gilbertson-Day, J.; Nicolet, T.; Ryan, K.; Tague, C. 2013. Research and development supporting risk-based wildfire effects prediction for fuels and fire management: status and needs. International Journal of Wildland Fire. 22(1): 37-50. doi:10.1071/WF11143

Innes, J.C.; North, M.P.; Williamson, N. 2006. Effect of thinning and prescribed fire restoration treatments on woody debris and snag dynamics in a Sierran old-growth, mixed-conifer forest. Canadian Journal of Forest Research. 36(12): 3183-3193. doi:10.1139/x06-184

Irwin, L.L. 1994. A process for improving wildlife habitat models for assessing forest ecosystem health. Journal of Sustainable Forestry. 2(3-4): 293-306. doi:10.1300/J091v02n03_05

Irwin, L.L.; Cook, J.G.; Riggs, R.A.; Skovlin, J.M. 1994. Effects of long-term grazing by big game and livestock in the Blue Mountains forest ecosystems. Gen. Tech. Rep. PNW-GTR-325. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 49 p. https://www.fs.usda.gov/treesearch/pubs/6330

Irwin, L.L.; Riggs, R.A.; Verschuyl, J.P. 2018. Reconciling wildlife conservation to forest restoration in moist mixed-conifer forests of the inland northwest: A synthesis. Forest Ecology and Management. 424: 288-311. doi:10.1016/j.foreco.2018.05.007

Isaac, L.A. 1938. Factors affecting establishment of Douglas fir seedlings. Circular No. 486. Washington, DC: U.S. Department of Agriculture. 46 p. Factors_Affecting_Establishment_of_Dougl

Ishii, H.T.; Tanabe, S-i.; Hiura, T. 2004. Exploring the relationships among canopy structure, stand productivity, and biodiversity of temperate forest ecosystems. Forest Science. 50(3): 342-355. doi:10.1093/forestscience/50.3.342

Iverson, D.C.; Alston, R.M. 1993. Ecosystem-based forestry requires a broader economic focus. Journal of Sustainable Forestry. 1(2): 97-106. doi:10.1300/J091v01n02_05

Jack, S.B.; Long, J.N. 1996. Linkages between silviculture and ecology: an analysis of density management diagrams. Forest Ecology and Management. 86(1-3): 205-220. doi:10.1016/S0378-1127(96)03770-X

Jackson, S.T.; Hobbs, R.J. 2009. Ecological restoration in the light of ecological history. Science. 325(5940): 567-569. doi:10.1126/science.1172977

Jactel, H.; Brockerhoff, E.G. 2007. Tree diversity reduces herbivory by forest insects. Ecology Letters. 10(9): 835-848. doi:10.1111/j.1461-0248.2007.01073.x

Jactel, H.; Nicoll, B., C.; Branco, M.; Gonzalez-Olabarria, J., Ramon; Grodzki, W.; Långström, B.; Moreira, F.; Netherer, S.; Orazio, C.; Piou, D.; Santos, H.; Schelhaas, M., Jan; Tojic, K.; Vodde, F. 2009. The influences of forest stand management on biotic and abiotic risks of damage. Annals of Forest Science. 66(7): 701 (18 p). doi:10.1051/forest/2009054

Jaenicke, A.J. 1948. Miscellaneous memoranda regarding a proposal to control budworm in the vicinity of the Umatilla National Forest. Unpub. Typescript Memoranda. Portland, OR: USDA Forest Service. 4 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5412767.pdf

Jain, T.B.; Graham, R.T. 2002. Restoring moist forests of the inland northwestern United States. In: Proceedings of the IUFRO conference on restoration of boreal and temperate forests: documenting forest restoration knowledge and practices in boreal and temperate ecosystems. Denmark: Danish Center for Forest, Landscape and Planning: 157-158.

Jain, T.B.; Graham, R.T. 2005. Restoring dry and moist forests of the inland northwestern U.S. Chapter 30. In: Stanturf, J.A.; Madsen, P., eds. Restoration of boreal and temperate forests. Boca Raton, FL: CRC Press: 463-480. isbn:1-56670-635-1

Jain, T.B.; Graham, R.T. 2007. The relation between tree burn severity and forest structure in the Rocky Mountains. In: Powers, R.F., ed. Restoring fire-adapted ecosystems: proceedings of the 2005 National Silviculture Workshop. Albany, CA: USDA Forest Service, Pacific Southwest Research Station: 213-250. http://www.treesearch.fs.fed.us/pubs/25904

Jain, T.B.; Graham, R.T.; Morgan, P. 2002. Western white pine development in relation to biophysical characteristics across different spatial scales in the Coeur d'Alene River basin in northern Idaho, U.S.A. Canadian Journal of Forest Research. 32(7): 1109-1125. doi:10.1139/x02-032

Jain, T.B.; Graham, R.T.; Morgan, P. 2004. Western white pine growth relative to forest openings. Canadian Journal of Forest Research. 34(11): 2187-2198. doi:10.1139/x04-094

Jain, T.B.; Graham, R.T.; Adams, D. 2008a. Carbon concentrations and carbon pool distributions in dry, moist, and cold mid-aged forests of the Rocky Mountains. In: Deal, R.L., ed. Integrated restoration of forested ecosystems to achieve multi-resource benefits: proceedings of the 2007 national silviculture workshop. Gen. Tech. Rep. PNW-GTR-733. Portland, OR: USDA Forest Service, Pacific Northwest Research Station: 39-59. https://www.fs.usda.gov/treesearch/pubs/29399

Jain, T.B.; Graham, R.T.; Sandquist, J.; Butler, M.; Brockus, K.; Frigard, D.; Cobb, D.; Sup-Han, H.; Halbrook, J.; Denner, R.; Evans, J.S. 2008b. Restoration of northern Rocky Mountain moist forests: integrating fuel treatments from the site to the landscape. In: Deal, R.L., ed. Integrated restoration of forested ecosystems to achieve multi-resource benefits: proceedings of the 2007 national silviculture workshop. Gen. Tech. Rep. PNW-GTR-733. Portland, OR: USDA Forest Service, Pacific Northwest Research Station: 147-172. https://www.fs.usda.gov/treesearch/pubs/32782

Jaindl, R.G.; Quigley, T.M., eds. 1996. Search for a solution: sustaining the land, people, and economy of the Blue Mountains. Washington, DC: American Forests. 316 p. ASIN:B0030H52BC

James, J.N.; Kates, N.; Kuhn, C.D.; Littlefield, C.E.; Miller, C.W.; Bakker, J.D.; Butman, D.E.; Haugo, R.D. 2018. The effects of forest restoration on ecosystem carbon in western North America: A systematic review. Forest Ecology and Management. 429: 625-641. doi:10.1016/j.foreco.2018.07.029

Jameson, D.A. 1968. Species interactions of growth inhibitors in native plants of northern Arizona. Res. Note RM-11. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 2 p. Species_Interactions_of_Growth_Inhibitors

Janousek, W.M.; Hicke, J.A.; Meddens, A.J.H.; Dreitz, V.J. 2019. The effects of mountain pine beetle outbreaks on avian communities in lodgepole pine forests across the greater Rocky Mountain region. Forest Ecology and Management. 444: 374-381. doi:10.1016/j.foreco.2019.04.047

Jaworski, D.; Kline, J.D.; Miller, C.; Ng, K.; Retzlaff, M.; Eichman, H.; Smith, D. 2018. Evaluating ecosystem services as management outcomes in national forest and grassland planning assessments. Gen. Tech. Rep. PNW-GTR-968. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 38 p. https://www.fs.usda.gov/treesearch/pubs/57455

Jeansson, E. 1997. Twenty-five advantages of natural regeneration. In: Emmingham, W.H., comp. Proceedings of the IUFRO interdisciplinary uneven-aged management symposium. [Corvallis, OR]: [Oregon State University]: 288-306.

Jenkins, M.J.; Dicus, C.A.; Hebertson, E.G. 1998. Postfire succession and disturbance interactions on an intermountain subalpine spruce-fir forest. In: Pruden, T.L.; Brennan, L.A., eds. Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology Conference Proceedings No. 20. Tallahassee, FL: Tall Timbers Research Station: 219-229. https://talltimbers.org/wp-content/uploads/2018/09/219-Jenkinsetal1998_op.pdf

Jenkins, M.J.; Hebertson, E.; Page, W.; Jorgensen, C.A. 2008. Bark beetles, fuels, fires and implications for forest management in the Intermountain West. Forest Ecology and Management. 254(1): 16-34. doi:10.1016/j.foreco.2007.09.045

Jenkins, M.J.; Page, W.G.; Hebertson, E.G.; Alexander, M.E. 2012. Fuels and fire behavior dynamics in bark beetle-attacked forests in western North America and implications for fire management. Forest Ecology and Management. 275: 23-34. doi:10.1016/j.foreco.2012.02.036

Jennings, M.; Loucks, O.; Peet, R.; Faber-Langendoen, D.; Glenn-Lewin, D.; Grossman, D.; Damman, A.; Barbour, M.; Pfister, R.; Walker, M.; Talbot, S.; Walker, J.; Hartshorn, G.; Waggoner, G.; Abrams, M.; Hill, A.; Roberts, D.; Tart, D.; Rejmanek, M. 2003. Guidelines for describing associations and alliances of the U.S. national vegetation classification; version 3 (November 25, 2003). Washington, DC: Ecological Society of America. 152 p.

Jensen, M.; Goodman, I.; Brewer, K.; Frost, T.; Ford, G.; Nesser, J. 1997. Chapter 2: biophysical environments of the basin. In: Quigley, T.M.; Arbelbide, S.J., tech. eds. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins: volume 1. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: USDA Forest Service, Pacific Northwest Research Station: 99-320. http://www.treesearch.fs.fed.us/pubs/24921

Jimenez, J.M. 2005. Bracken fern communities of the grand fir mosaic: biomass allocation and influence on selected soil properties. M.S. thesis. Moscow, ID: University of Idaho. 209 p.

Johnsey, R. 1984. Bark beetles of the interior Douglas-fir–grand fir forest type. In: Baumgartner, D.M.; Mitchell, R., comps. Silvicultural management strategies for pests of the interior Douglas-fir and grand fir forest types. Pullman, WA: Washington State University, Cooperative Extension: 103-107.

Johnson, B.L. 1999. The role of adaptive management as an operational approach for resource management agencies. Conservation Ecology. 3(2): article 8. http://www.ecologyandsociety.org/vol3/iss2/art8/

Johnson, C.G. 1993 (August 9). Ecosystem screens; file designation 2060 memorandum to Wallowa-Whitman, Umatilla, and Malheur Forest Supervisors. Baker City, OR: USDA Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest. 4 p (and 6 exhibits). On file with: USDA Forest Service, Umatilla National Forest, Supervisor's Office, Pendleton, OR.

Johnson, C.G., Jr. 1994. Forest health in the Blue Mountains: a plant ecologist's perspective on ecosystem processes and biological diversity. Gen. Tech. Rep. PNW-GTR-339. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 24 p. http://www.treesearch.fs.fed.us/pubs/5104

Johnson, C.G., Jr. 1998. Vegetation response after wildfires in national forests of northeastern Oregon. R6-NR-ECOL-TP-06-98. Portland, OR: USDA Forest Service, Pacific Northwest Region. 155 p. http://ecoshare.info/uploads/publications/VegetationResponseWFNFNEOrg.pdf

Johnson, C.G., Jr. 2004. Alpine and subalpine vegetation of the Wallowa, Seven Devils and Blue Mountains. Tech. Publ. R6-NR-ECOL-TP-03-04. Portland, OR: USDA Forest Service, Pacific Northwest Region. 612 p. http://ecoshare.info/uploads/publications/AlpineVegCompleteBook.pdf

Johnson, N.E.; Belluschi, P.G. 1969. Host-finding behavior of the Douglas-fir beetle. Journal of Forestry. 67(5): 290-295. doi:10.1093/jof/67.5.290

Johnson, C.G., Jr.; Clausnitzer, R.R. 1992. Plant associations of the Blue and Ochoco Mountains. Tech. Pub. R6-ERW-TP-036-92. Portland, OR: USDA Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest. 164 p. Plant-Associations-of-the-blue-and-Ochoco-Mountains

Johnson, D.R.; Chance, D.H. 1974. Presettlement overharvest of upper Columbia River beaver populations. Canadian Journal of Zoology. 52(12): 1519-1521. doi:10.1139/z74-195

Johnson, C.G., Jr.; Simon, S.A. 1987. Plant associations of the Wallowa-Snake province. Tech. Publ. R6-ECOL-TP-225b-86. Baker City, OR: USDA Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest. 272 p. http://ecoshare.info/2011/11/03/plant-associations-of-the-wallowa-snake-province/

Johnson, C.G., Jr.; Swanson, D.K. 2005. Bunchgrass plant communities of the Blue and Ochoco Mountains: a guide for managers. Gen. Tech. Rep. PNW-GTR-641. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 119 p. http://www.treesearch.fs.fed.us/pubs/20801

Johnson, E.A.; Fryer, G.I. 1989. Population dynamics in lodgepole pine-Engelmann spruce forests. Ecology. 70(5): 1335-1345. doi:10.2307/1938193

Johnson, E.A.; Fryer, G.I. 1996. Why Engelmann spruce does not have a persistent seed bank. Canadian Journal of Forest Research. 26(5): 872-878. doi:10.1139/x26-095

Johnson, E.A.; Gutsell, S.L. 1994. Fire frequency models, methods and interpretations. Advances in Ecological Research. 25: 239-287. doi:10.1016/S0065-2504(08)60216-0

Johnson, E.A.; Larsen, C.P.S. 1991. Climatically induced change in fire frequency in the southern Canadian Rockies. Ecology. 72(1): 194-201. doi:10.2307/1938914

Johnson, E.A.; Miyanishi, K. 2007a. Disturbance and succession. In: Johnson, E.A.; Miyanishi, K., eds. Plant disturbance ecology: the process and the response. Amsterdam: Academic Press: 1-14. isbn:978-0-12-088778-1

Johnson, E.A.; Miyanishi, K., eds. 2007b. Plant disturbance ecology: the process and the response. Amsterdam: Academic Press. 698 p. isbn:978-0-12-088778-1

Johnson, D.W.; Turner, J. 2019. Nutrient cycling in forests: A historical look and newer developments. Forest Ecology and Management. 444: 344-373. doi:10.1016/j.foreco.2019.04.052

Johnson, C.G., Jr.; Clausnitzer, R.R.; Mehringer, P.J.; Oliver, C.D. 1994. Biotic and abiotic processes of eastside ecosystems: the effects of management on plant and community ecology, and on stand and landscape vegetation dynamics. Gen. Tech. Rep. PNW-GTR-322. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 66 p. http://www.treesearch.fs.fed.us/pubs/6252

Johnson, D.W.; Beatty, J.S.; Hinds, T.E. 1995a. Cankers on western quaking aspen. Forest Insect and Disease Leaflet 152. [Place of publication unknown]: USDA Forest Service. 8 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043102.pdf

Johnson, K.N.; Agee, J.; Beschta, R.; Beuter, J.; Gregory, S.; Kellogg, L.; McComb, W.; Sedell, J.; Schowalter, T.; Tesch, S. 1995b. Forest health and timber harvest on national forests in the Blue Mountains of Oregon: a report to Governor Kitzhaber. Corvallis, OR: Oregon State University, College of Forestry. 51 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015662.pdf

Johnson, E.A.; Miyanishi, K.; Weir, J.M.H. 1995c. Old-growth, disturbance, and ecosystem management. Canadian Journal of Botany. 73(6): 918-926. doi:10.1139/b95-100

Johnson, E.A.; Miyanishi, K.; Weir, J.M.H. 1998. Wildfires in the western Canadian boreal forest: Landscape patterns and ecosystem management. Journal of Vegetation Science. 9(4): 603-610. doi:10.2307/3237276

Johnson, E.A.; Miyanishi, K.; Bridge, S.R.J. 2001. Wildfire regime in the boreal forest and the idea of suppression and fuel buildup. Conservation Biology. 15(6): 1554-1557. doi:10.1046/j.1523-1739.2001.01005.x

Johnson, E.A.; Morin, H.; Miyanishi, K.; Gagnon, R.; Greene, D.F. 2003. A process approach to understanding disturbance and forest dynamics for sustainable forestry. In: Burton, P.J.; Messier, C.; Smith, D.W.; Adamowicz, W.L., eds. Towards sustainable management of the boreal forest. Ottawa, ON: NRC Research Press: 261-306. isbn:0-660-18762-0

Johnson, D.; Murphy, J.D.; Walker, R.F.; Glass, D.W.; Miller, W.W. 2007. Wildfire effects on forest carbon and nutrient budgets. Ecological Engineering. 31(3): 183-192. doi:10.1016/j.ecoleng.2007.03.003

Johnson-Maynard, J.L.; McDaniel, P.A.; Ferguson, D.E.; Falen, A.L. 1998. Changes in soil solution chemistry of andisols following invasion by bracken fern. Soil Science. 163(10): 814-821. https://www.fs.usda.gov/treesearch/pubs/35330

Jolly, W.M.; Parsons, R.; Varner, J.M.; Butler, B.W.; Ryan, K.C.; Gucker, C.L. 2012. Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? Comment. Ecology 93(4): 941-946. doi:10.1890/11-1035.1

Jones, M.E.; Paine, T.D.; Fenn, M.E.; Poth, M.A. 2004. Influence of ozone and nitrogen deposition on bark beetle activity under drought conditions. Forest Ecology and Management. 200(1-3): 67-76. doi:10.1016/j.foreco.2004.06.003

Jones, B.E.; Rickman, T.H.; Vazquez, A.; Sado, Y.; Tate, K.W. 2005. Removal of encroaching conifers to regenerate degraded aspen stands in the Sierra Nevada. Restoration Ecology. 13(2): 373-379. doi:10.1111/j.1526-100X.2005.00046.x

Josephy, A.M., Jr. 1971. The Nez Perce Indians and the opening of the northwest. London, UK: Yale University. 667 p. isbn:0-300-01494-5

Joyce, L.A.; Blate, G.M.; Littell, J.S.; McNulty, S.G.; Millar, C.I.; Moser, S.C.; Neilson, R.P.; O'Halloran, K.; Peterson, D.L. 2008. National forests. In: Julius, S.H.; West, J.M., eds. Preliminary review of adaptation options for climate-sensitive ecosystems and resources. Washington, DC: U.S. Environmental Protection Agency: 3-1 to 3-127.

Joyce, L.; Blate, G.; McNulty, S.; Millar, C.; Moser, S.; Neilson, R.; Peterson, D. 2009. Managing for multiple resources under climate change: national forests. Environmental Management. 44(6): 1022-1032. doi:10.1007/s00267-009-9324-6

Jurgensen, M.F.; Arno, S.F.; Harvey, A.E.; Larsen, M.J.; Pfister, R.D. [1979.] Symbiotic and nonsymbiotic nitrogen fixation in northern Rocky Mountain ecosystems. In: Gordon, J.C.; Wheeler, C.T.; Perry, D.A., eds. Symbiotic nitrogen fixation in the management of temperate forests; proceedings of a workshop held April 2-5, 1979. Corvallis, OR: Oregon State University, Forest Research Laboratory: 294-308.

Jurgensen, M.F.; Larsen, M.J.; Harvey, A.E. 1980. Microbial processes associated with nitrogen cycling in northern Rocky Mountain forest soils. In: USDA Forest Service. Environmental consequences of timber harvesting in Rocky Mountain coniferous forests. Gen. Tech. Rep. INT-90. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 175-188.

https://archive.org/download/CAT80751102/CAT80751102.pdf

Jurgensen, M.F.; Larsen, M.J.; Graham, R.T.; Harvey, A.E. 1987. Nitrogen fixation in woody residue of northern Rocky Mountain conifer forests. Canadian Journal of Forest Research. 17(10): 1283-1288. doi:10.1139/x87-198

Jurgensen, M.; Larsen, M.; Wolosiewicz, M.; Harvey, A. 1989. A comparison of dinitrogen fixation rates in wood litter decayed by white-rot and brown-rot fungi. Plant and Soil. 115(1): 117-122. doi:10.1007/bf02220701

Jurgensen, M.F.; Tonn, J.R.; Graham, R.T.; Harvey, A.E.; Geier-Hayes, K. 1991. Nitrogen fixation in forest soils of the inland northwest. In: Harvey, A.E.; Neuenschwander, L.F., comps. Proceedings–management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: USDA Forest Service, Intermountain Research Station: 101-109. http://www.treesearch.fs.fed.us/pubs/33908

Jurgensen, M.F.; Graham, R.T.; Larsen, M.J.; Harvey, A.E. 1992. Clear-cutting, woody residue removal, and nonsymbiotic nitrogen fixation in forest soils of the inland Pacific Northwest. Canadian Journal of Forest Research. 22(8): 1172-1178. doi:10.1139/x92-155

Jurgensen, M.F.; Graham, R.T.; Harvey, A.E.; Page-Dumroese, D.S.; Tonn, J.R. 1995. Woody residue and soil organic matter in western larch ecosystems. In: Schmidt, W.C.; McDonald, K.J., comps. Ecology and management of Larix forests: a look ahead; proceedings of an international symposium. Gen. Tech. Rep. GTR-INT-319. Ogden, UT: USDA Forest Service, Intermountain Research Station: 370-374. https://archive.org/download/CAT10699418/CAT10699418.pdf

Jurgensen, M.F.; Harvey, A.E.; Graham, R.T.; Page-Dumroese, D.S.; Tonn, J.R.; Larsen, M.J.; Jain, T.B. 1997. Impacts of timber harvesting on soil organic matter, nitrogen, productivity, and health of inland northwest forests. Forest Science. 43(2): 234-251. doi:10.1093/forestscience/43.2.234

Kalabokidis, K.D.; Gatzojannis, S.; Galatsidas, S. 2002. Introducing wildfire into forest management planning: towards a conceptual approach. Forest Ecology and Management. 158(1-3): 41-50. doi:10.1016/S0378-1127(00)00715-5

Kane, J.M.; Kolb, T.E.; McMillin, J.D. 2014. Stand-scale tree mortality factors differ by site and species following drought in southwestern mixed conifer forests. Forest Ecology and Management. 330: 171-182. doi:10.1016/j.foreco.2014.06.042

Kane, V.R.; Lutz, J.A.; Alina Cansler, C.; Povak, N.A.; Churchill, D.J.; Smith, D.F.; Kane, J.T.; North, M.P. 2015. Water balance and topography predict fire and forest structure patterns. Forest Ecology and Management. 338: 1-13. doi:10.1016/j.foreco.2014.10.038

Kane, J.M.; Varner, J.M.; Metz, M.R.; van Mantgem, P.J. 2017. Characterizing interactions between fire and other disturbances and their impacts on tree mortality in western U.S. Forests. Forest Ecology and Management. 405: 188-199. doi:10.1016/j.foreco.2017.09.037

Kane, V.R.; Bartl-Geller, B.N.; North, M.P.; Kane, J.T.; Lydersen, J.M.; Jeronimo, S.M.A.; Collins, B.M.; Monika Moskal, L. 2019. First-entry wildfires can create opening and tree clump patterns characteristic of resilient forests. Forest Ecology and Management. 454: 117659. doi:10.1016/j.foreco.2019.117659

Kangas, A.; Korhonen, K.T.; Packalen, T.; Vauhkonen, J. 2018. Sources and types of uncertainties in the information on forest-related ecosystem services. Forest Ecology and Management. 427: 7-16. doi:10.1016/j.foreco.2018.05.056

Kareiva, P.; Watts, S.; McDonald, R.; Boucher, T. 2007. Domesticated nature: shaping landscapes and ecosystems for human welfare. Science. 316(5833): 1866-1869. doi:10.1126/science.1140170

Karl, T.R.; Melillo, J.M.; Peterson, T.C. 2009. Global climate change impacts in the United States. New York: Cambridge University Press. 188 p. isbn:978-0-521-14407-0

Kashian, D.M.; Romme, W.H.; Tinker, D.B.; Turner, M.G.; Ryan, M.G. 2006. Carbon storage on landscapes with stand-replacing fires. BioScience 56(7):598-606. doi:10.1641/0006-3568(2006)56[598:CSOLWS]2.0.CO;2

Kashian, D.M.; Romme, W.H.; Tinker, D.B.; Turner, M.G.; Ryan, M.G. 2013. Postfire changes in forest carbon storage over a 300-year chronosequence of *Pinus contorta*-dominated forests. Ecological Monographs. 83(1): 49-66. doi:10.1890/11-1454.1

Kauffman, J.B. 1990. Ecological relationships of vegetation and fire in Pacific Northwest forests. In: Walstad, J.D.; Radosevich, S.R.; Sandberg, D.V., eds. Natural and prescribed fire in Pacific Northwest forests. Corvallis, OR: Oregon State University Press: 39-52. isbn:0-87071-359-0

Kaufmann, M.R. 1996. To live fast or not: growth, vigor and longevity of old-growth ponderosa pine and lodgepole pine trees. Tree Physiology. 16(1-2): 139-144. doi:10.1093/treephys/16.1-2.139

Kauffman, J.B. 2004. Death rides the forest: perceptions of fire, land use, and ecological restoration of western forests. Conservation Biology. 18(4): 878-882. doi:10.1111/j.1523-1739.2004.545_1.x

Kauffman, J.B.; Martin, R.E. 1989. Fire behavior, fuel consumption, and forest-floor changes following prescribed understory fires in Sierra Nevada mixed conifer forests. Canadian Journal of Forest Research. 19(4): 455-462. doi:10.1139/x89-071

Kaufmann, M.R.; Graham, R.T.; Boyce, D.A., Jr.; Moir, W.H.; Perry, L.; Reynolds, R.T.; Bassett, R.L.; Mehlhop, P.; Edminster, C.B.; Block, W.M.; Corn, P.S. 1994. An ecological basis for ecosystem management. Gen. Tech. Rep. RM-246. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 22 p. http://www.treesearch.fs.fed.us/pubs/7612

Kay, C.E. 1994. Aboriginal overkill: the role of native Americans in structuring western ecosystems. Human Nature. 5(4): 359-398. doi:10.1007/BF02734166

Kay, C.E.; Simmons, R.T., eds. 2002. Wilderness and political ecology: aboriginal influences and the original state of nature. Salt Lake City, UT: University of Utah Press. 342 p. isbn:0-87480-719-0

Keane, R.E.; Morgan, P.; Menakis, J.P. 1994. Landscape assessment of the decline of whitebark pine *(Pinus albicaulis)* in the Bob Marshall Wilderness complex, Montana, USA. Northwest Science. 68(3): 213-229. http://hdl.handle.net/2376/1552

Keane, R.E.; Morgan, P.; Running, S.W. 1996a. FIRE-BGC – a mechanistic ecological process model for simulating fire succession on coniferous forest landscapes of the northern Rocky Mountains. Res. Pap. INT-RP-484. Ogden, UT: USDA Forest Service, Intermountain Research Station. 122 p. https://archive.org/download/firebgcamechanis484kean/firebgcamechanis484kean.pdf

Keane, R.E.; Ryan, K.C.; Running, S.W. 1996b. Simulating effects of fire on northern Rocky Mountain landscapes with the ecological process model FIRE-BGC. Tree Physiology. 16(3): 319-331. doi:10.1093/treephys/16.3.319

Keane, R.E.; Ryan, K.C.; Veblen, T.T.; Allen, C.D.; Logan, J.; Hawkes, B. 2002. Cascading effects of fire exclusion in Rocky Mountain ecosystems: a literature review. Gen. Tech. Rep. RMRS-GTR-91. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 24 p. https://www.fs.usda.gov/treesearch/pubs/5132

Keane, R.E.; Parsons, R.E.; Rollins, M.G. 2004. Predicting fire regimes at multiple scales. In: Perera, A.H.; Buse, L.J.; Weber, M.G., eds. Emulating natural forest landscape disturbances: concepts and applications. New York: Columbia University Press: 55-68. isbn:0-231-12916-5

Keane, R.E.; Agee, J.K.; Fulé, P.; Keeley, J.E.; Key, C.; Kitchen, S.G.; Miller, R.; Schulte, L.A. 2008. Ecological effects of large fires on US landscapes: benefit or catastrophe? International Journal of Wildland Fire. 17(6): 696-712. doi:10.1071/WF07145

Keane, R.E.; Hessburg, P.F.; Landres, P.B.; Swanson, F.J. 2009. The use of historical range and variability (HRV) in landscape management. Forest Ecology and Management. 258(7): 1025-1037. doi:10.1016/j.foreco.2009.05.035

Keane, R.E.; Holsinger, L.M.; Mahalovich, M.F.; Tomback, D.F. 2017. Restoring whitebark pine ecosystems in the face of climate change. Gen. Tech. Rep. RMRS-GTR-361. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 123 p. https://www.fs.usda.gov/treesearch/pubs/54577

Keane, R.E.; Sikkink, P.G.; Jain, T.B. 2018. Physical and chemical characteristics of surface fuels in masticated mixed-conifer stands of the U.S. Rocky Mountains. Gen. Tech. Rep. RMRS-GTR-370. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 56 p. https://www.fs.usda.gov/treesearch/pubs/55649

Keane, R.E.; Holsinger, L.M.; Loehman, R. 2020. Bioclimatic modeling of potential vegetation types as an alternative to species distribution models for projecting plant species shifts under changing climates. Forest Ecology and Management. 477: 118498. doi:10.1016/j.foreco.2020.118498

Keeley, J.E. 1981. Reproductive cycles and fire regimes. In: Mooney, H.A.; Bonnicksen, T.M.; Christensen, N.L.; Lotan, J.E.; Reiners, W.A., tech. coords. Fire regimes and ecosystem properties; proceedings of the conference. Gen. Tech. Rep. WO-26. [Washington, DC]: USDA Forest Service: 231-277. https://archive.org/download/CAT83781017/CAT83781017.pdf

Keeley, J.E. 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. International Journal of Wildland Fire. 18(1): 116-126. doi:10.1071/WF07049

Keeley, J.E.; Aplet, G.H.; Christensen, N.L.; Conard, S.G.; Johnson, E.A.; Omi, P.N.; Peterson, D.L.; Swetnam, T.W. 2009. Ecological foundations for fire management in North American forest and shrubland ecosystems. Gen. Tech. Rep. PNW-GTR-779. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 92 p. https://www.fs.usda.gov/treesearch/pubs/32483

Keen, F.P. 1950. The influence of insects on ponderosa pine silviculture. Journal of Forestry. 48(3): 186-188. doi:10.1093/jof/48.3.186

Keenan, R.J.; Kimmins, J.P. 1993. The ecological effects of clear-cutting. Environmental Reviews. 1(2): 121-144. doi:10.1139/a93-010

Keisker, D.G. 2000. Types of wildlife trees and coarse woody debris required by wildlife of north-central British Columbia. Work. Pap. 50. Victoria, BC: British Columbia Ministry of Forests, Research Branch. 92 p. http://www.for.gov.bc.ca/hfd/pubs/docs/wp/wp50.htm

Kelly, L.T.; Brotons, L. 2017. Using fire to promote biodiversity. Science. 355(6331): 1264-1265. doi:10.1126/science.aam7672

Kelly, J.; Harwell, M. 1990. Indicators of ecosystem recovery. Environmental Management. 14(5): 527-545. doi:10.1007/bf02394708

Kelly, A.; Powell, D.C.; Riggs, R.A. 2005. Predicting potential natural vegetation in an interior northwest landscape using classification tree modeling and a GIS. Western Journal of Applied Forestry. 20(2): 117-127. doi:10.1093/wjaf/20.2.117

Kelsey, R.G. 2001. Chemical indicators of stress in trees: their ecological significance and implication for forestry in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 70-76. http://hdl.handle.net/2376/983

Kelsey, R.G.; Harrington, M.G. 1979. A search for phytotoxins influencing germination and early growth of ponderosa pine. Res. Pap. INT-216. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 28 p. https://archive.org/download/searchforphytoto216kels/searchforphytoto216kels.pdf

Kelty, M.J. 2006. The role of species mixtures in plantation forestry. Forest Ecology and

Management. 233(2-3): 195-204. doi:10.1016/j.foreco.2006.05.011

Kemp, K.B.; Higuera, P.E.; Morgan, P.; Abatzoglou, J.T. 2019. Climate will increasingly determine post-fire tree regeneration success in low-elevation forests, Northern Rockies, USA. Ecosphere. 10(1): e02568 (17 p). doi:10.1002/ecs2.2568

Kennedy, J.J.; Thomas, J.W. 1995. Managing natural resources as social value. In: Knight, R.L.; Bates, S.F., eds. A new century for natural resources management. Washington, DC: Island Press: 311-321. isbn:1-55963-262-3

Kennedy, M.C.; Johnson, M.C.; Fallon, K.; Mayer, D. 2019. How big is enough? Vegetation structure impacts effective fuel treatment width and forest resiliency. Ecosphere. 10(2): e02573 (11 p). doi:10.1002/ecs2.2573

Kenworthy, T. 1992. 'Unraveling' of ecosystem looms in Oregon forests. Washington, DC: Washington Post. May 15; Section A: A01.

Kercher, J.R.; Axelrod, M.C. 1984. A process model of fire ecology and succession in a mixed-conifer forest. Ecology. 65(6): 1725-1742. doi:10.2307/1937768

Kessler, W.B.; Salwasser, H., Cartwright, C.W., Jr.; Caplan, J.A. 1992. New perspectives for sustainable natural resources management. Ecological Applications. 2(3): 221-225. doi:10.2307/1941856

Key, C.H. 2006. Ecological and sampling constraints on defining landscape fire severity. Fire Ecology. 2(2): 34-59. doi:10.4996/fireecology.0202034

Keyes, C.R. 1996. Stand structures and silvicultural strategies to prevent crown fires in northern Rocky Mountain forests. M.S. thesis. Missoula, MT: University of Montana. 116 p. http://scholarworks.umt.edu/cgi/viewcontent.cgi?article=4675&context=etd

Keyes, C.R.; O'Hara, K.L. 2002. Quantifying stand targets for silvicultural prevention of crown fires. Western Journal of Applied Forestry. 17(2): 101-109. doi:10.1093/wjaf/17.2.101

Keyser, A.R.; Westerling, A.L. 2019. Predicting increasing high severity area burned for three forested regions in the western United States using extreme value theory. Forest Ecology and Management. 432: 694-706. doi:10.1016/j.foreco.2018.09.027

Keyser, T.L.; Lentile, L.B.; Smith, F.W.; Shepperd, W.D. 2008. Changes in forest structure after a large, mixed-severity wildfire in ponderosa pine forests of the Black Hills, South Dakota, USA. Forest Science. 54(3): 328-338. doi:10.1093/forestscience/54.3.328

Khanduri, V.P.; Sharma, C.M.; Singh, S.P. 2008. The effects of climate change on plant phenology. Environmentalist. 28(2): 143-147. doi:10.1007/s10669-007-9153-1

Kibbee, D.L.; Berryman, A.A.; Amman, G.D.; Stark, R.W., tech. eds. 1978. Theory and practice of mountain pine beetle management in lodgepole pine forest. Moscow, ID: University of Idaho, Forest, Wildlife and Range Experiment Station. 224 p. Theory_and_Practice_of_Mountain_Pine_Beetle

Kilgore, B.M. 1981. Fire in ecosystem distribution and structure: western forests and scrublands. In: Mooney, H.A.; Bonnicksen, T.M.; Christensen, N.L.; Lotan, J.E.; Reiners, W.A., tech. coords. Fire regimes and ecosystem properties; proceedings of the conference. Gen. Tech. Rep. WO-26. [Washington, DC]: USDA Forest Service: 58-89. https://archive.org/download/CAT83781017/CAT83781017.pdf

Kim, Y.-H.; Bettinger, P.; Finney, M. 2009. Spatial optimization of the pattern of fuel management activities and subsequent effects on simulated wildfires. European Journal of Operational Research. 197(1): 253-265. doi:10.1016/j.ejor.2008.05.025

Kimmerer, R.W.; Lake, F.K. 2001. The role of indigenous burning in land management. Journal of Forestry. 99(11): 36-41. doi:10.1093/jof/99.11.36

Kimmins, J.P. 1996a. Importance of soil and role of ecosystem disturbance for sustained productivity of cool temperate and boreal forests. Soil Science Society of America Journal.

60(6): 1643-1654. doi:10.2136/sssaj1996.03615995006000060007x

Kimmins, J.P. 1996b. The health and integrity of forest ecosystems: are they threatened by forestry? Ecosystem Health. 2(1): 5-18.

Kimmins, J.P. 1997. Forest ecology; a foundation for sustainable management. 2nd edition. Upper Saddle River, NJ: Prentice Hall. 596 p. isbn:0-02-364071-5

Kimmins, J.P. 2004. Emulating natural forest disturbance: What does this mean? In: Perera, A.H.; Buse, L.J.; Weber, M.G., eds. Emulating natural forest landscape disturbances: concepts and applications. New York: Columbia University Press: 8-28. isbn:0-231-12916-5

Kimmins, J.P. 2008. From science to stewardship: harnessing forest ecology in the service of society. Forest Ecology and Management. 256(10): 1625-1635. doi:10.1016/j.foreco.2008.02.057

Kimsey, M.J.; Shaw, T.M.; Coleman, M.D. 2019. Site sensitive maximum stand density index models for mixed conifer stands across the Inland Northwest, USA. Forest Ecology and Management. 433: 396-404. doi:10.1016/j.foreco.2018.11.013

Kingery, J.L.; Mosley, J.C.; Bordwell, K.C. 1996. Dietary overlap among cattle and cervids in northern Idaho forests. Journal of Range Management. 49(1): 8-15. https://journals.uair.arizona.edu/index.php/jrm/article/download/9078/8690

Kipfmueller, K.F.; Baker, W.L. 1998a. A comparison of three techniques to date stand-replacing fires in lodgepole pine forests. Forest Ecology and Management. 104(1-3): 171-177. doi:10.1016/S0378-1127(97)00245-4

Kipfmueller, K.F.; Baker, W.L. 1998b. Fires and dwarf mistletoe in a Rocky Mountain lodgepole pine ecosystem. Forest Ecology and Management. 108(1-2): 77-84. doi:10.1016/S0378-1127(98)00217-5

Kipfmueller, K.F.; Kupfer, J.A. 2005. Complexity of successional pathways in subalpine forests of the Selway-Bitterroot Wilderness area. Annals of the Association of American Geographers. 95(3): 495-510. doi:10.1111/j.1467-8306.2005.00471.x

Kishchuk, B.E.; Thiffault, E.; Lorente, M.; Quideau, S.; Keddy, T.; Sidders, D. 2015. Decadal soil and stand response to fire, harvest, and salvage-logging disturbances in the western boreal mixedwood forest of Alberta, Canada. Canadian Journal of Forest Research. 45(2): 141-152. doi:10.1139/cjfr-2014-0148

Kitzhaber, J.A.; Forsgren, H.; Zielinski, E. 2001. An 11-point strategy for restoring eastern Oregon forests, watersheds and communities (dated April 13). Three-page enclosure with a memorandum to Forest Supervisors and District Managers (file code 2510; dated June 14, 2001).

Kleinman, J.S.; Goode, J.D.; Fries, A.C.; Hart, J.L. 2019. Ecological consequences of compound disturbances in forest ecosystems: a systematic review. Ecosphere. 10(11): e02962 (18 p). doi:10.1002/ecs2.2962

Kliejunas, J.T. 2011. A risk assessment of climate change and the impact of forest diseases on forest ecosystems in the western United States and Canada. Gen. Tech. Rep. PSW-GTR-236. Albany, CA: USDA Forest Service, Pacific Southwest Research Station. 70 p. https://www.fs.usda.gov/treesearch/pubs/40137

Kliejunas, J.T.; Geils, B.W.; Glaeser, J.M.; Goheen, E.M.; Hennon, P.; Kim, M.-S.; Kope, H.; Stone, J.; Sturrock, R.; Frankel, S.J. 2009. Review of literature on climate change and forest diseases of western North America. Gen. Tech. Rep. PSW-GTR-225. Albany, CA: USDA Forest Service, Pacific Southwest Research Station. 54 p. http://www.treesearch.fs.fed.us/pubs/33904

Klinka, K.; Worrall, J.; Skoda, L.; Varga, P. 2000. The distribution and synopsis of ecological and silvical characteristics of tree species of British Columbia's forests. Coquitlam, BC: Canadian Cartographics Ltd. 180 p. isbn:0-921259-27-1

Klock, G.O.; Grier, C.C. 1979. Effects of fire on the long-term maintenance of forest productivity. In: Gessel, S.P.; Kenady, R.M.; Atkinson, W.A., eds. Proceedings, forest fertilization conference. Contribution No. 40. Seattle, WA: University of Washington, College of Forest Resources: 247-250.

Klopfenstein, N.B.; Kim, M.-S.; Hanna, J.W.; Richardson, B.A.; Lundquist, J.E. 2009. Approaches to predicting potential impacts of climate change on forest disease: an example with Armillaria root disease. Res. Pap. RMRS-RP-76. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 10 p. http://www.treesearch.fs.fed.us/pubs/33179

Knapp, E.C.; Keeley, J.E. 2006. Heterogeneity in fire severity within early season and late season prescribed burns in a mixed-conifer forest. International Journal of Wildland Fire. 15(1): 37-45. doi:10.1071/WF04068

Knapp, E.E.; Ritchie, M.W. 2016. Response of understory vegetation to salvage logging following a high-severity wildfire. Ecosphere. 7(11): e01550 (17 p). doi:10.1002/ecs2.1550

Kneeshaw, D.; Gauthier, S. 2003. Old growth in the boreal forest: a dynamic perspective at the stand and landscape level. Environmental Reviews. 11(S1): S99-S114. doi:10.1139/a03-010

Kneeshaw, D.D.; Leduc, A.; Drapeau, P.; Gauthier, S.; Paré, D.; Carignan, R.; Doucet, R.; Bouthillier, L.; Messier, C. 2000. Development of integrated ecological standards of sustainable forest management at an operational scale. Forestry Chronicle. 76(3): 481-493. doi:10.5558/tfc76481-3

Knight, R.L.; Bates, S.F. 1995. A new century for natural resources management. Washington, DC: Island Press. 398 p. isbn:1-55963-262-3

Knight, T.M.; McCoy, M.W.; Chase, J.M.; McCoy, K.A.; Holt, R.D. 2005. Trophic cascades across ecosystems. Nature. 437(7060): 880-883. doi:10.1038/nature03962

Knudson, R. 1980. Ancient peoples of the Columbia plateau. Journal of Forestry. 78(8): 477-479. doi:10.1093/jof/78.8.477

Knutson, D.; Tinnin, R. 1986. Effects of dwarf mistletoe on the response of young Douglas-fir to thinning. Canadian Journal of Forest Research. 16(1): 30-35. doi:10.1139/x86-006

Knutson, D.M.; Toevs, W.J. 1972. Dwarf mistletoe reduces root growth of ponderosa pine seedlings. Forest Science. 18(4): 323-324. doi:10.1093/forestscience/18.4.323

Kobziar, L.N.; McBride, J.R. 2006. Wildfire burn patterns and riparian vegetation response along two northern Sierra Nevada streams. Forest Ecology and Management. 222(1-3): 254-265. doi:10.1016/j.foreco.2005.10.024

Kobziar, L.; Moghaddas, J.; Stephens, S.L. 2006. Tree mortality patterns following prescribed fires in a mixed conifer forest. Canadian Journal of Forest Research. 36(12): 3222-3238. doi:10.1139/x06-183

Koch, P. 1996. Lodgepole pine in North America. Volume 1, part 1: background. Madison, WI: Forest Products Society. 343 p. isbn:978-0935018783

Kohm, K.A.; Franklin, J.F., eds. 1997. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press. 475 p. isbn:1-55963-399-9

Kolb, T.E.; Wagner, M.R.; Covington, W.W. 1994. Concepts of forest health: utilitarian and ecosystem perspectives. Journal of Forestry. 92(7): 10-15. doi:10.1093/jof/92.7.10

Kolb, T.E.; Holmberg, K.M.; Wagner, M.R.; Stone, J.E. 1998. Regulation of ponderosa pine foliar physiology and insect resistance mechanisms by basal area treatments. Tree Physiology. 18(6): 375-381. doi:10.1093/treephys/18.6.375

Kolb, S.E.; Fermanich, K.J.; Dornbush, M.E. 2009. Effect of charcoal quantity on microbial biomass and activity in temperate soils. Soil Science Society of America Journal. 73(4): 1173-1181. doi:10.2136/sssaj2008.0232

Kolden, C.A.; Lutz, J.A.; Key, C.H.; Kane, J.T.; van Wagtendonk, J.W. 2012. Mapped versus actual burned area within wildfire perimeters: characterizing the unburned. Forest Ecology and Management. 286: 38-47. doi:10.1016/j.foreco.2012.08.020

Kolden, C.A.; Abatzoglou, J.T.; Lutz, J.A.; Cansler, C.A.; Kane, J.T.; Wagtendonk, J.W.V.; Key, C.H. 2015. Climate contributors to forest mosaics: Ecological persistence following wildfire. Northwest Science. 89(3): 219-238. doi:10.3955/046.089.0305

Kondolf, G.M.; Boulton, A.J.; O'Daniel, S.; Poole, G.C.; Rahel, F.J.; Stanley, E.H.; Wohl, E.; Bång, A.; Carlstrom, J.; Cristoni, C.; Huber, H.; Koljonen, S.; Louhi, P.; Nakamura, K. 2006. Process-based ecological river restoration: Visualizing three-dimensional connectivity and dynamic vectors to recover lost linkages. Ecology and Society. 11(2): 5 (17 p). doi:10.5751/ES-01747-110205

Konkin, D.; Hopkins, K. 2009. Learning to deal with climate change and catastrophic forest disturbances. Unasylva. 60(231/232): 17-23. http://www.fao.org/docrep/011/i0670e/i0670e04.htm

Koontz, T.M.; Bodine, J. 2008. Implementing ecosystem management in public agencies: Lessons from the U.S. Bureau of Land Management and the Forest Service. Conservation Biology. 22(1): 60-69. doi:10.1111/j.1523-1739.2007.00860.x

Korb, J.E.; Fornwalt, P.J.; Stevens-Rumann, C.S. 2019. What drives ponderosa pine regeneration following wildfire in the western United States? Forest Ecology and Management. 454: 117663 (16 p). doi:10.1016/j.foreco.2019.117663

Kotar, J. 1997. Silviculture and ecosystem management. In: Boyce, M.S.; Haney, A., eds. Ecosystem management: applications for sustainable forest and wildlife resources. New Haven, CT: Yale University Press: 265-275. isbn:0-300-06902-2

Koyama, A.; Kavanagh, K.; Robinson, A. 2005. Marine nitrogen in central Idaho riparian forests: evidence from stable isotopes. Canadian Journal of Fisheries and Aquatic Sciences. 62(3): 518-526. doi:10.1139/f04-220

Krebs, C.J.; Boonstra, R.; Cowcill, K.; Kenney, A.J. 2009. Climatic determinants of berry crops in the boreal forest of the southwestern Yukon. Botany. 87(4): 401-408. doi:10.1139/B09-013

Kreider, M.R.; Mock, K.E.; Yocom, L.L. 2020. Methods for distinguishing aspen seedlings from suckers in the field. Journal of Forestry. 118(6): 561-568. doi:10.1093/jofore/fvaa030

Krist, F.J., Jr.; Sapio, F.J.; Tkacz, B.M. 2007. Mapping risk from forest insects and diseases. FHTET 2007-06. Washington, DC: USDA Forest Service, Forest Health Protection, Forest Health Technology Enterprise Team. 115 p. https://www.fs.fed.us/foresthealth/technology/pdfs/FHTET2007-06_RiskMap.pdf

Kronland, W.J.; Restani, M. 2012. Effects of post-fire salvage logging on cavity-nesting birds and small mammals in southeastern Montana. Canadian Field-Naturalist. 125(4): 316-326. doi:10.22621/cfn.v125i4.1260

Kuiters, A.T.; Mohren, G.M.J.; Van Wieren, S.E. 1996. Ungulates in temperate forest ecosystems. Forest Ecology and Management. 88(1-2): 1-5. doi:10.1016/s0378-1127(96)03876-5

Kulman, H.M. 1971. Effects of insect defoliation on growth and mortality of trees. Annual Review of Entomology. 16(1): 289-324. doi:10.1146/annurev.en.16.010171.001445

Kurz, W.A.; Dymond, C.C.; Stinson, G.; Rampley, G.J.; Neilson, E.T.; Carroll, A.L.; Ebata, T.; Safranyik, L. 2008. Mountain pine beetle and forest carbon feedback to climate change. Nature. 452(7190): 987-990. doi:10.1038/nature06777

Kushla, J.D.; Ripple, W.J. 1997. The role of terrain in a fire mosaic of a temperate coniferous forest. Forest Ecology and Management. 95(2): 97-107. doi:10.1016/S0378-1127(97)82929-5

Kuuluvainen, T. 2002. Natural variability of forests as a reference for restoring and managing biological diversity in boreal Fennoscandia. Silva Fennica. 36(1): 97-125.

doi:10.14214/sf.552

Kuuluvainen, T.; Grenfell, R. 2012. Natural disturbance emulation in boreal forest ecosystem management – theories, strategies, and a comparison with conventional even-aged management. Canadian Journal of Forest Research. 42(7): 1185-1203. doi:10.1139/X2012-064

Lacki, M.J.; Hayes, J.P.; Kurta, A., eds. 2007. Bats in forests: conservation and management. Baltimore, MD: John Hopkins University Press. 329 p. isbn:978-0-8018-8499-3

Laiho, R.; Prescott, C.E. 1999. The contribution of coarse woody debris to carbon, nitrogen, and phosphorus cycles in three Rocky Mountain coniferous forests. Canadian Journal of Forest Research. 29(10): 1592-1603. doi:10.1139/x99-132

Laiho, R.; Prescott, C.E. 2004. Decay and nutrient dynamics of coarse woody debris in northern coniferous forests: a synthesis. Canadian Journal of Forest Research. 34(4): 763-777. doi:10.1139/x03-241

Landhäusser, S.M.; Pinno, B.D.; Mock, K.E. 2019. Tamm Review: Seedling-based ecology, management, and restoration in aspen (Populus tremuloides). Forest Ecology and Management. 432: 231-245. doi:10.1016/j.foreco.2018.09.024

Landres, P.B.; Morgan, P.; Swanson, F.J. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications. 9(4): 1179-1188. doi:10.1890/1051-0761(1999)009[1179:OOTUON]2.0.CO;2

Landsberg, J.J.; Gower, S.T. 1997. Applications of physiological ecology to forest management. San Diego, CA: Academic Press. 354 p. isbn:0-12-435955-8

Lane, B.B.; Goheen, D.J. 1979. Incidence of root disease in bark beetle-infested eastern Oregon and Washington true firs. Plant Disease Reporter. 63(4): 262-266.

Langelier, L.A.; Garton, E.O. 1986. Management guidelines for increasing populations of birds that feed on western spruce budworm. Agric. Hand. No. 653. Washington, DC: USDA Forest Service, Cooperative State Research Service. 19 p. Spruce_Budworms_Handbook

Langenheim, J.H. 1990. Plant resins. American Scientist. 78 (January-February): 16-24. http://www.jstor.org/stable/29773859

Langille, H.D. 1906. Report on the proposed Blue Mountains Forest Reserve. Unpublished typescript report obtained from the National Archives, College Park, MD; record group 95. 40 p. On file with: USDA Forest Service, Umatilla National Forest, Supervisor's Office, Pendleton, OR. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015626.pdf

Langston, N. 2000. When sound science is not enough: regulating the Blues. Journal of Forestry. 98(11): 31-35. doi:10.1093/jof/98.11.31

Lapin, M.; Barnes, B.V. 1995. Using the landscape ecosystem approach to assess species and ecosystem diversity. Conservation Biology. 9(5): 1148-1158. doi:10.1046/j.1523-1739.1995.9051134.x-i1

Lapointe-Garant, M.-P.; Huang, J.-G.; Gea-Izquierdo, G.; Raulier, F.; Bernier, P.; Berninger, F. 2010. Use of tree rings to study the effect of climate change on trembling aspen in Québec. Global Change Biology. 16(7): 2039-2051. doi:10.1111/j.1365-2486.2009.02048.x

La Roi, G.H.; Hnatiuk, R.J. 1980. The Pinus contorta forests of Banff and Jasper National Parks: A study in comparative synecology and syntaxonomy. Ecological Monographs. 50(1): 1-29. doi:10.2307/2937244

Larsen, J.A. 1916. Silvical notes on western larch. Proceedings of the Society of American Foresters. 11(1): 434-440.

Larsen, J.A. 1922. Effect of removal of the virgin white pine stand upon the physical factors of site. Ecology. 3(4): 302-305. doi:10.2307/1929432

Larsen, J.A. 1923. Association of trees, shrubs, and other vegetation in the northern Idaho forests. Ecology. 4(1): 63-67. doi:10.2307/1929274

Larsen, J.A. 1924. Some factors affecting reproduction after logging in northern Idaho. Journal of Agricultural Research. 28(11): 1149-1157.

Larsen, J.A. 1925. Natural reproduction after forest fires in northern Idaho. Journal of Agricultural Research. 30(12): 1177-1197.

Larsen, J.A. 1929. Fires and forest succession in the Bitterroot Mountains of northern Idaho. Ecology. 10(1): 67-76. doi:10.2307/1940513

Larsen, J.A. 1930. Forest types of the northern Rocky Mountains and their climatic controls. Ecology. 11(4): 631-672. doi:10.2307/1932327

Larsen, J.A. 1940. Site factor variations and responses in temporary forest types in northern Idaho. Ecological Monographs. 10(1): 1-54. doi:10.2307/1943200

Larsen, J.B. 1995. Ecological stability of forests and sustainable silviculture. Forest Ecology and Management. 73(1-3): 85-96. doi:10.1016/0378-1127(94)03501-M

Larsen, J.B.; Nielsen, A.B. 2007. Nature-based forest management–Where are we going? Elaborating forest development types in and with practice. Forest Ecology and Management. 238(1-3): 107-117. doi:10.1016/j.foreco.2006.09.087

Larsen, M.J.; Jurgensen, M.F.; Harvey, A.E. 1978. N_2 fixation associated with wood decayed by some common fungi in western Montana. Canadian Journal of Forest Research. 8(3): 341-345. doi:10.1139/x78-050

Larsen, M.J.; Harvey, A.E.; Jurgensen, M.F. 1980. Residue decay processes and associated environmental functions in northern Rocky Mountain forests. In: USDA Forest Service. Environmental consequences of timber harvesting in Rocky Mountain coniferous forests. Gen. Tech. Rep. INT-90. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 157-174. https://archive.org/download/CAT80751102/CAT80751102.pdf

Larson, B.C. 1986. Development and growth of even-aged stands of Douglas-fir and grand fir. Canadian Journal of Forest Research. 16(2): 367-372. doi:10.1139/x86-063

Larson, A.J.; Churchill, D. 2008. Spatial patterns of overstory trees in late-successional conifer forests. Canadian Journal of Forest Research. 38(11): 2814-2825. doi:10.1139/X08-123

Larson, A.J.; Stover, K.C.; Keyes, C.R. 2012. Effects of restoration thinning on spatial heterogeneity in mixed-conifer forest. Canadian Journal of Forest Research. 42(8): 1505-1517. doi:10.1139/x2012-100

Latham, P.A.; Zuuring, H.R.; Coble, D.W. 1998. A method for quantifying vertical forest structure. Forest Ecology and Management. 104(1-3): 157-170. doi:10.1016/S0378-1127(97)00254-5

Latif, Q.S.; Ivan, J.S.; Seglund, A.E.; Pavlacky, D.L.; Truex, R.L. 2020. Avian relationships with bark beetle outbreaks and underlying mechanisms in lodgepole pine and spruce-fir forests of Colorado. Forest Ecology and Management. 464: 118043. doi:10.1016/j.foreco.2020.118043

Latta, G.; Temesgen, H.; Adams, D.; Barrett, T. 2009. Analysis of potential impacts of climate change on forests of the United States Pacific Northwest. Forest Ecology and Management. 259(4): 720-729. doi:10.1016/j.foreco.2009.09.003

Laudenslayer, W.F., Jr.; Shea, P.J.; Valentine, B.E.; Weatherspoon, C.P.; Lisle, T.E., eds. 2002. Proceedings of the symposium on the ecology and management of dead wood in western forests. Gen. Tech. Rep. PSW-GTR-181. Albany, CA: USDA Forest Service, Pacific Southwest Research Station. 949 p. http://www.treesearch.fs.fed.us/pubs/6718

Lausch, A.; Heurich, M.; Gordalla, D.; Dobner, H.J.; Gwillym-Margianto, S.; Salbach, C. 2013. Forecasting potential bark beetle outbreaks based on spruce forest vitality using hyperspectral remote-sensing techniques at different scales. Forest Ecology and Management. 308: 76-89. doi:10.1016/j.foreco.2013.07.043

Lawler, J.J.; Tear, T.H.; Pyke, C.; Shaw, M.R.; Gonzalez, P.; Kareiva, P.; Hansen, L.; Hannah, L.; Klausmeyer, K.; Aldous, A.; Bienz, C.; Pearsall, S. 2010. Resource management in a changing and uncertain climate. Frontiers in Ecology and the Environment. 8(1): 35-43. doi:10.1890/070146

Leach, W.D. 2002. Surveying diverse stakeholder groups. Society and Natural Resources. 15(7): 641-649. doi:10.1080/08941920290069245

Leaphart, C.D.; Denton, R.E. 1961. Needle discolorations of western larch. Forest Pest Leaflet 61. [Place of publication unknown]: USDA Forest Service. 7 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043563.pdf

Lee, K.N. 1999. Appraising adaptive management. Conservation Ecology. 3(2): article 3. http://www.ecologyandsociety.org/vol3/iss2/art3/

Leemans, R.; Eickhout, B. 2004. Another reason for concern: regional and global impacts on ecosystems for different levels of climate change. Global Environmental Change. 14(3): 219-228. doi:10.1016/j.gloenvcha.2004.04.009

Lehmkuhl, J.F.; Hessburg, P.F.; Everett, R.L.; Huff, M.H.; Ottmar, R.D. 1994. Historical and current forest landscapes of eastern Oregon and Washington; part I: vegetation pattern and insect and disease hazards. Gen. Tech. Rep. PNW-GTR-328. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 88 p. http://www.treesearch.fs.fed.us/pubs/6407

Lehmkuhl, J.F.; Lyons, A.L.; Bracken, E.; Leingang, J.; Gaines, W.L.; Dodson, E.K.; Singleton, P.H. 2013. Forage composition, productivity, and utilization in the eastern Washington Cascade Range. Northwest Science. 87(4): 267-291. doi:10.3955/046.087.0404

Lenihan, J.M.; Bachelet, D.; Neilson, R.P.; Drapek, R. 2008. Response of vegetation distribution, ecosystem productivity, and fire to climate change scenarios for California. Climatic Change. 87(1): 215-230. doi:10.1007/s10584-007-9362-0

Lentile, L.B.; Smith, F.W.; Shepperd, W.D. 2006. Influence of topography and forest structure on patterns of mixed severity fire in ponderosa pine forests of the South Dakota Black Hills, USA. International Journal of Wildland Fire. 15(4): 557-566. doi:10.1071/WF05096

Leopold, A. 1968. A Sand County almanac, and sketches from here and there. London, UK: Oxford University Press. 226 p. isbn:0-19-500777-8

Lertzman, K.P. 1995. Forest dynamics, differential mortality and variable recruitment probabilities. Journal of Vegetation Science. 6(2): 191-204. doi:10.2307/3236214

Lertzman, K.; Fall, J. 1998. From forest stands to landscapes: spatial scales and the roles of disturbances. In: Peterson, D.L.; Parker, V.T., eds. Ecological scale: theory and application. New York: Columbia University Press: 339-367. isbn:0-231-10503-7

Lertzman, K.P.; Fall, J.; Dorner, B. 1998. Three kinds of heterogeneity in fire regimes: at the crossroads of fire history and landscape ecology. Northwest Science. 72(Special Issue 1): 4-23.

Lesica, P. 1996. Using fire history models to estimate proportions of old growth forest in northwest Montana, USA. Biological Conservation. 77(1): 33-39. doi:10.1016/0006-3207(95)00130-1

Lesica, P.; McCune, B.; Cooper, S.V.; Hong, W.S. 1991. Differences in lichen and bryophyte communities between old-growth and managed second-growth forests in the Swan Valley, Montana. Canadian Journal of Botany. 69(8): 1745-1755. doi:10.1139/b91-222

Lewandowsky, S.; Gignac, G.E.; Vaughan, S. 2013. The pivotal role of perceived scientific consensus in acceptance of science. Nature Climate Change. 3(4): 399-404.

doi:10.1038/nclimate1720

Lewinsohn, E.; Gijzen, M.; Muzika, R.M.; Barton, K.; Croteau, R. 1993. Oleoresinosis in grand fir (*Abies grandis*) saplings and mature trees (modulation of this wound response by light and water stresses). Plant Physiology. 101(3): 1021-1028. doi:10.1104/pp.101.3.1021

Lewinsohn, D.; Lewinsohn, E.; Bertagnolli, C.L.; Partridge, A.D. 1994. Blue-stain fungi and their transport structures on the Douglas-fir beetle. Canadian Journal of Forest Research. 24(11): 2275-2283. doi:10.1139/x94-292

Lewis, H.T. 1985. Why Indians burned: specific versus general reasons. In: Lotan, J.E.; Kilgore, B.M.; Fischer, W.C.; Mutch, R.W., tech. coords. Proceedings–symposium and workshop on wilderness fire. Gen. Tech. Rep. INT-182. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 75-80. https://archive.org/download/CAT31118982/CAT31118982.pdf

Lewis, D. 2009. Stand and landscape-level simulations of mountain pine beetle (*Dendroctonus ponderosae*) and salvage logging effects on live tree and deadwood habitats in south-central British Columbia, Canada. Forest Ecology and Management. 258(Supplement): S24-S35. doi:10.1016/j.foreco.2009.08.031

Lewis, H.T.; Ferguson, T.A. 1988. Yards, corridors, and mosaics: How to burn a boreal forest. Human Ecology. 16(1): 57-77. doi:10.1007/bf01262026

Lexerød, N.L.; Eid, T. 2006. An evaluation of different diameter diversity indices based on criteria related to forest management planning. Forest Ecology and Management. 222(1-3): 17-28. doi:10.1016/j.foreco.2005.10.046

Li, C. 2000. Reconstruction of natural fire regimes through ecological modelling. Ecological Modelling. 134(2-3): 129-144. doi:10.1016/S0304-3800(00)00290-8

Li, C.; Barclay, H.J. 2001. Fire disturbance patterns and forest age structure. Natural Resource Modeling. 14(4): 495-521. doi:10.1111/j.1939-7445.2001.tb00071.x

Li, C.; Ter-Mikaelian, M.; Perera, A. 1997. Temporal fire disturbance patterns on a forest landscape. Ecological Modelling. 99(2-3): 137-150. doi:10.1016/S0304-3800(96)01944-8

Li, Z.; Apps, M.J.; Kurz, W.A.; Banfield, E. 2003. Temporal changes of forest net primary production and net ecosystem production in west central Canada associated with natural and anthropogenic disturbances. Canadian Journal of Forest Research. 33(12): 2340-2351. doi:10.1139/x03-168

Li, C.; Barclay, H.J.; Hawkes, B.C.; Taylor, S.W. 2005. Lodgepole pine forest age class dynamics and susceptibility to mountain pine beetle attack. Ecological Complexity. 2(3): 232-239. doi:10.1016/j.ecocom.2005.03.001

Li, C.; Hans, H.; Barclay, H.; Liu, J.; Carlson, G.; Campbell, D. 2008. Comparison of spatially explicit forest landscape fire disturbance models. Forest Ecology and Management. 254(3): 499-510. doi:10.1016/j.foreco.2007.07.022

Liang, J.; Buongiorno, J.; Monserud, R.A. 2004. CalPro: a spreadsheet program for the management of California mixed-conifer stands. Gen. Tech. Rep. PNW-GTR-619. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 32 p. https://www.fs.usda.gov/treesearch/pubs/7506

Liang, J.; Calkin, D.E.; Gebert, K.M.; Venn, T.J.; Silverstein, R.P. 2008. Factors influencing large wildland fire suppression expenditures. International Journal of Wildland Fire. 17(5): 650-659. doi:10.1071/WF07010

Liebhold, A.M. 2012. Forest pest management in a changing world. International Journal of Pest Management. 58(3): 289-295. doi:10.1080/09670874.2012.678405

Liebhold, A.M.; MacDonald, W.L.; Bergdahl, D.; Mastro, V.C. 1995. Invasion by exotic forest pests: a threat to forest ecosystems. Forest Science Monographs. 41(Supplement 1): 1-49.

doi:10.1093/forestscience/41.s1.a0001

Lieffers, S.M.; Lieffers, V.J.; Silins, U.; Bach, L. 2001. Effects of cold temperatures on breakage of lodgepole pine and white spruce twigs. Canadian Journal of Forest Research. 31(9): 1650-1653. doi:10.1139/x01-087

Lieffers, V.J.; Messier, C.; Burton, P.J.; Ruel, J.-C.; Grover, B.E. 2003. Nature-based silviculture for sustaining a variety of boreal forest values. Chapter 13. In: Burton, P.J.; Messier, C.; Smith, D.W.; Adamowicz, W.L., eds. Towards sustainable management of the boreal forest. Ottawa, ON: NRC Research Press: 481-530. isbn:0-660-18762-0

Lindblom, C.E. 1959. The science of "muddling through". Public Administration Review. 19(2): 79-88. doi:10.2307/973677

Lindenmayer, D.B.; Franklin, J.F. 2002. Conserving forest biodiversity: A comprehensive multiscaled approach. Washington, DC: Island Press. 351 p. isbn:9781559639354

Lindenmayer, D.B.; Hunter, M.L.; Burton, P.J.; Gibbons, P. 2009. Effects of logging on fire regimes in moist forests. Conservation Letters. 2(6): 271-277. doi:10.1111/j.1755-263X.2009.00080.x

Lindenmayer, D.B.; Westgate, M.J.; Scheele, B.C.; Foster, C.N.; Blair, D.P. 2019. Key perspectives on early successional forests subject to stand-replacing disturbances. Forest Ecology and Management. 454: 117656 (10 p). doi:10.1016/j.foreco.2019.117656

Lindh, B.C.; Muir, P.S. 2004. Understory vegetation in young Douglas-fir forests: does thinning help restore old-growth composition? Forest Ecology and Management. 192(2-3): 285-296. doi:10.1016/j.foreco.2004.01.018

Littell, J.S.; Gwozdz, R.B. 2011. Climatic water balance and regional fire years in the Pacific Northwest, USA: linking regional climate and fire at landscape scales. In: McKenzie, D.; Miller, C.; Falk, D.A., eds. The landscape ecology of fire. New York: Springer: 117-139. isbn:978-94-007-0300-1

Littell, J.S.; McKenzie, D.; Peterson, D.L.; Westerling, A.L. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916-2003. Ecological Applications. 19(4): 1003-1021. doi:10.1890/07-1183.1

Littell, J.; Oneil, E.; McKenzie, D.; Hicke, J.; Lutz, J.; Norheim, R.; Elsner, M. 2010. Forest ecosystems, disturbance, and climatic change in Washington State, USA. Climatic Change. 102(1): 129-158. doi:10.1007/s10584-010-9858-x

Little, C.E. 1995. The dying of the trees: the pandemic in America's forests. New York: Penguin Books. 275 p. isbn:0-670-84135-8

Little, S.N.; Klock, G.O. 1985. The influence of residue removal and prescribed fire on distributions of forest nutrients. Res. Pap. PNW-338. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 12 p. https://www.fs.usda.gov/treesearch/pubs/25240

Liu, J.; Ashton, P.S. 1995. Individual-based simulation models for forest succession and management. Forest Ecology and Management. 73(1-3): 157-175. doi:10.1016/0378-1127(94)03490-N

Liu, J.; Taylor, W.W., eds. 2002. Integrating landscape ecology into natural resource management. Cambridge, UK: Cambridge University Press. 480 p. isbn:0-521-78433-6

Liu, G.; Nelson, J.D.; Wardman, C.W. 2000. A target-oriented approach to forest ecosystem design – changing the rules of forest planning. Ecological Modelling. 127(2-3): 269-281. doi:10.1016/S0304-3800(99)00217-3

Liu, Y.; Stanturf, J.; Goodrick, S. 2010. Trends in global wildfire potential in a changing climate. Forest Ecology and Management. 259(4): 685-697. doi:10.1016/j.foreco.2009.09.002

Livingston, R.L.; Dewey, J.E.; Beckman, D.P.; Stipe, L.E. 2000. Distribution of the balsam woolly adelgid in Idaho. Western Journal of Applied Forestry. 15(4): 227-231.

doi:10.1093/wjaf/15.4.227

Loehle, C. 1988a. Forest decline: endogenous dynamics, tree defenses, and the elimination of spurious correlation. Vegetatio. 77(1): 65-78. https://www.jstor.org/stable/20038343

Loehle, C. 1988b. Tree life history strategies: the role of defenses. Canadian Journal of Forest Research. 18(2): 209-222. doi:10.1139/x88-032

Loehle, C. 2004. Applying landscape principles to fire hazard reduction. Forest Ecology and Management. 198(1-3): 261-267. doi:10.1016/j.foreco.2004.04.010

Loehle, C.; LeBlanc, D. 1996. Model-based assessments of climate change effects on forests: a critical review. Ecological Modelling. 90(1): 1-31. doi:10.1016/0304-3800(96)83709-4

Loehle, C.; MacCracken, J.G.; Runde, D.; Hicks, L. 2002. Forest management at landscape scales: solving the problems. Journal of Forestry. 100(6): 25-33. doi:10.1093/jof/100.6.25

Loehman, R.A.; Clark, J.A.; Keane, R.E. 2011. Modeling effects of climate change and fire management on western white pine (*Pinus monticola*) in the northern Rocky Mountains, USA. Forests 2(4):832-860. doi:10.3390/f2040832

Logan, J.A.; Powell, J.A. 2001. Ghost forests, global warming, and the mountain pine beetle (Coleoptera: Scolytidae). American Entomologist. 47(3): 160-173. doi:10.1093/ae/47.3.160

Logan, J.A.; Régniere, J.; Powell, J.A. 2003. Assessing the impacts of global warming on forest pest dynamics. Frontiers in Ecology and the Environment. 1(3): 130-137. doi:10.1890/1540-9295(2003)001[0130:ATIOGW]2.0.CO;2

Lomborg, B. 2001. The skeptical environmentalist: measuring the real state of the world. Cambridge, UK: Cambridge University Press. 515 p. isbn:0-521-01068-3

Long, J.N. 1998. Multiaged systems in the central and southern Rockies. Journal of Forestry. 96(7): 34-36. doi:10.1093/jof/96.7.34

Long, J.N. 2003. Diversity, complexity and interactions: an overview of Rocky Mountain forest ecosystems. Tree Physiology. 23(16): 1091-1099. doi:10.1093/treephys/23.16.1091

Long, J.N. 2009. Emulating natural disturbance regimes as a basis for forest management: a North American view. Forest Ecology and Management. 257(9): 1868-1873. doi:10.1016/j.foreco.2008.12.019

Long, J.W.; Lake, F.K. 2018. Escaping social-ecological traps through tribal stewardship on national forest lands in the Pacific Northwest, United States of America. Ecology and Society. 23(2): 10 (14 p). doi:10.5751/ES-10041-230210

Long, J.N.; Mock, K. 2012. Changing perspectives on regeneration ecology and genetic diversity in western quaking aspen: implications for silviculture. Canadian Journal of Forest Research. 42(12): 2011-2021. doi:10.1139/x2012-143

Long, J.N.; Shaw, J.D. 2010. The influence of compositional and structural diversity on forest productivity. Forestry. 83(2): 121-128. doi:10.1093/forestry/cpp033

Long, J.N.; Dean, T.J.; Roberts, S.D. 2004. Linkages between silviculture and ecology: examination of several important conceptual models. Forest Ecology and Management. 200(1-3): 249-261. doi:10.1016/j.foreco.2004.07.005

Long, R.A.; Rachlow, J.L.; Kie, J.G. 2008a. Effects of season and scale on response of elk and mule deer to habitat manipulation. Journal of Wildlife Management. 72(5): 1133-1142. doi:10.2193/2007-337

Long, R.A.; Rachlow, J.L.; Kie, J.G.; Vavra, M. 2008b. Fuels reduction in a western coniferous forest: effects on quantity and quality of forage for elk. Rangeland Ecology and Management. 61(3): 302-313. doi:10.2111/07-046.1

Long, J.W.; Tarnay, L.W.; North, M.P. 2018. Aligning smoke management with ecological and public health goals. Journal of Forestry. 116(1): 76-86. doi:10.5849/jof.16-042

Lopushinsky, W. 1991. Water relations of interior Douglas-fir. In: Baumgartner, D.M.; Lotan, J.E., comps. Interior Douglas-fir: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 35-46.

Lopushinsky, W.; Klock, G.O. 1980. Effect of defoliation on transpiration in grand fir. Canadian Journal of Forest Research. 10(1): 114-116. doi:10.1139/x80-020

Lorimer, C.G. 1985. Methodological considerations in the analysis of forest disturbance history. Canadian Journal of Forest Research. 15(1): 200-213. doi:10.1139/x85-038

Losin, N.; Floyd, C.H.; Todd, E.S.; Keller, S.J. 2006. Relationship between aspen heartwood rot and the location of cavity excavation by a primary cavity-nester, the red-naped sapsucker. The Condor. 108(3): 706-710. https://www.jstor.org/stable/4151090

Lotan, J.E. 1976. Cone serotiny–fire relationships in lodgepole pine. In: Proceedings of the annual Tall Timbers fire ecology conference no. 14. Tallahassee, FL: Tall Timbers Research Station: 267-278. https://talltimbers.org/wp-content/uploads/2018/09/267-Lotan1974_op.pdf

Lotan, J.E.; Critchfield, W.B. 1990. *Pinus contorta* Dougl. ex. Loud.; lodgepole pine. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America; volume 1, conifers. Agric. Handb. 654. Washington, DC: USDA Forest Service: 302-315. https://www.treesearch.fs.fed.us/pubs/1547

Lotan, J.E.; Perry, D.A. 1983. Ecology and regeneration of lodgepole pine. Agric. Handb. No. 606. Washington, DC: USDA Forest Service. 51 p. Agriculture_Handbook

Lotan, J.E.; Kilgore, B.M.; Fischer, W.C.; Mutch, R.W., tech. coords. 1985a. Proceedings–symposium and workshop on wilderness fire. Gen. Tech. Rep. INT-182. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 434 p. https://archive.org/download/CAT31118982/CAT31118982.pdf

Lotan, J.E.; Brown, J.K.; Neuenschwander, L.F. 1985b. Role of fire in lodgepole pine forests. In: Baumgartner, D.M.; Krebill, R.G.; Arnott, J.T.; Weetman, G.F., comps. Lodgepole pine: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Cooperative Extension, Office of Conferences and Institutes: 133-152.

Loucks, D.M.; Black, H.C.; Roush, M.L.; Radosevich, S.R. 1990. Assessment and management of animal damage in Pacific Northwest forests: an annotated bibliography. Gen. Tech. Rep. PNW-GTR-262. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 371 p. https://www.fs.usda.gov/treesearch/pubs/9109

Lowdermilk, W.C. 1925. Factors affecting reproduction of Engelmann spruce. Journal of Agricultural Research. 30(11): 995-1009. https://naldc-legacy.nal.usda.gov/naldc/download.xhtml?id=IND43967040&content=PDF

Lucas, O.W.R. 1991. The design of forest landscapes. Oxford, UK: Oxford University Press. 381 p. isbn:0-19-854280-1

Lucas, E. 1992. Bugged to death. Pacific Northwest (March): 7, 10.

Lucas, E. 1993. Burning question: thinning and fire prescribed for pest-infested forests. Horizon Magazine. 4(3): 12-13, 28, 30-31.

Ludwig, D. 1993. Forest management strategies that account for short-term and long-term consequences. Canadian Journal of Forest Research. 23(4): 563-572. doi:10.1139/x93-075

Ludwig, D.; Hilborn, R.; Walters, C. 1993. Uncertainty, resource exploitation, and conservation: lessons from history. Ecological Applications. 3(4): 547-549. doi:10.2307/1942074

Ludwig, D.; Walker, B.; Holling, C.S. 1997. Sustainability, stability, and resilience. Conservation Ecology [online]. 1(1): 1-25. http://www.consecol.org/vol1/iss1/art7/

Lugo, A.E.; Gucinski, H. 2000. Function, effects, and management of forest roads. Forest Ecology and Management. 133(3): 249-262. doi:10.1016/S0378-1127(99)00237-6

Luna, T. 2003. Propagation protocol for aspen using root cuttings. Native Plants Journal. 4(2): 129-131. https://www.jstor.org/stable/43310339

Lundholm, J.T.; Larson, D.W. 2004. Restoring artifacts as a metaphor for restoring ecosystems: A hands-on exercise for teaching restoration ecology. Ecological Restoration. 22(2): 126-130. doi:10.3368/er.22.2.126

Lundquist, J.E.; Beatty, J.S. 1999. A conceptual model for defining and assessing condition of forest stands. Environmental Management. 23(4): 519-525. doi:10.1007/s002679900206

Lundquist, J.E.; Beatty, J.S. 2002. A method for characterizing and mimicking forest canopy gaps caused by different disturbances. Forest Science. 48(3): 582-594. doi:10.1093/forestscience/48.3.582

Lundquist, J.E.; Goheen, E.M.; Goheen, D.J. 2002. Measuring positive, negative, and null impacts of forest disturbances: a case study using dwarf mistletoe on Douglas fir. Environmental Management. 30(6): 793-800. doi:10.1007/s00267-002-2756-x

Lutz, H.J. 1957. Applications of ecology in forest management. Ecology. 38(1): 46-49. doi:10.2307/1932125

Lutz, H.J. 1963. Forest ecosystems: Their maintenance, amelioration, and deterioration. Journal of Forestry. 61(8): 563-569. doi:10.1093/jof/61.8.563

Lutz, J.A.; Halpern, C.B. 2006. Tree mortality during early forest development: a long-term study of rates, causes, and consequences. Ecological Monographs. 76(2): 257-275. doi:10.1890/0012-9615(2006)076[0257:TMDEFD]2.0.CO;2

Lutz, J.A.; van Wagtendonk, J.W.; Franklin, J.F. 2010. Climatic water deficit, tree species ranges, and climate change in Yosemite National Park. Journal of Biogeography. 37(5): 936-950. doi:10.1111/j.1365-2699.2009.02268.x

Lutz, J.A.; Larson, A.J.; Swanson, M.E.; Freund, J.A. 2012. Ecological importance of large-diameter trees in a temperate mixed-conifer forest. PLoS ONE. 7(5): e36131 (15 p). doi:10.1371/journal.pone.0036131

Luyssaert, S.; Schulze, E.D.; Borner, A.; Knohl, A.; Hessenmoller, D.; Law, B.E.; Ciais, P.; Grace, J. 2008. Old-growth forests as global carbon sinks. Nature. 455(7210): 213-215. doi:10.1038/nature07276

Luyssaert, S.; Hessenmöller, D.; von Lüpke, N.; Kaiser, S.; Schulze, E.D. 2011. Quantifying land use and disturbance intensity in forestry, based on the self-thinning relationship. Ecological Applications. 21(8): 3272-3284. doi:10.1890/10-2395.1

Lydersen, J.; North, M. 2012. Topographic variation in structure of mixed-conifer forests under an active-fire regime. Ecosystems. 15(7): 1134-1146. doi:10.1007/s10021-012-9573-8

Lynch, H.J.; Moorcroft, P.R. 2008. A spatiotemporal Ripley's K-function to analyze interactions between spruce budworm and fire in British Columbia, Canada. Canadian Journal of Forest Research. 38(12): 3112-3119. doi:10.1139/x08-143

Lynch, H.J.; Renkin, R.A.; Crabtree, R.L.; Moorcroft, P.R. 2006. The influence of previous mountain pine beetle (*Dendroctonus ponderosae*) activity on the 1988 Yellowstone fires. Ecosystems. 9(8): 1318-1327. doi:10.1007/s10021-006-0173-3

Lynott, R.E.; Cramer, O.P. 1966. Detailed analysis of the 1962 Columbus Day windstorm in Oregon and Washington. Monthly Weather Review. 94(2): 105-117. doi:10.1175/1520-0493(1966)094%3C0105:DAOTCD%3E2.3.CO;2

Lyon, L.J. 1971. Vegetal development following prescribed burning of Douglas-fir in south-central Idaho. Res. Pap. INT-105. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 30 p. https://archive.org/download/vegetaldevelopme105lyon/vegetaldevelopme105lyon.pdf

Lyon, L.J. 1984. The Sleeping Child Burn–21 years of postfire change. Res. Pap. INT-330. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station.

17 p. http://digitalcommons.usu.edu/cgi/viewcontent.cgi?article=1014&context=govdocs_forest

Lyons-Tinsley, C.; Peterson, D.L. 2012. Surface fuel treatments in young, regenerating stands affect wildfire severity in a mixed conifer forest, eastside Cascade Range, Washington, USA. Forest Ecology and Management. 270: 117-125. doi:10.1016/j.foreco.2011.04.016

Ma, S.; Concilio, A.; Oakley, B.; North, M.; Chen, J. 2010. Spatial variability in microclimate in a mixed-conifer forest before and after thinning and burning treatments. Forest Ecology and Management. 259(5): 904-915. doi:10.1016/j.foreco.2009.11.030

Ma, Z.; Steele, D.; Cutler, A.; Newcomb, K. 2020. Promoting sustainability in public natural-resource agencies: Insights from the USDA Forest Service. Journal of Forestry. 118(2): 105-123. doi:10.1093/jofore/fvz067

MacArthur, R.H. 1984. Geographical ecology: patterns in the distribution of species. Princeton, NJ: Princeton University Press. 269 p. isbn:0-691-02382-4

MacArthur, R.H.; Wilson, E.O. 2001. The theory of island biogeography. Princeton, NJ: Princeton University Press. 203 p. isbn:0-691-08836-5

MacCleery, D.W. 2011. American forests: a history of resiliency and recovery. Durham, NC: Forest History Society. 71 p. https://foresthistory.org/wp-content/uploads/2016/12/American_Forests.pdf

MacCleery, D.W. 1995. The way to a healthy future for National Forest ecosystems in the West: What role can silviculture and prescribed fire play? In: Eskew, L.G., comp. Forest health through silviculture: proceedings of the 1995 National Silviculture Workshop. Gen. Tech. Rep. RM-GTR-267. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO: 37-45. https://www.fs.usda.gov/treesearch/pubs/23483

MacCleery, D.W.; Le Master, D.C. 1999. The historical foundation and evolving context for natural resource management on federal lands. In: Szaro, R.C.; Johnson, N.C.; Sexton, W.T.; Malk, A.J., eds. Ecological stewardship: a common reference for ecosystem management. Volume II. Oxford, UK: Elsevier Science, Ltd: 517-556. isbn:0-08-043206-9

MacFarlane, D.W.; Luo, A. 2009. Quantifying tree and forest bark structure with a bark-fissure index. Canadian Journal of Forest Research. 39(10): 1859-1870. doi:10.1139/X09-098

Mack, C.A. 2003. A burning issue: American Indian fire use on the Mt. Rainier Forest Reserve. Fire Management Today. 63(2): 20-24. https://www.fs.fed.us/file/40954/download?token=kYI6rtgO

MacKenzie, M.D.; DeLuca, T.H.; Sala, A. 2006. Fire exclusion and nitrogen mineralization in low elevation forests of western Montana. Soil Biology and Biochemistry. 38(5): 952-961. doi:10.1016/j.soilbio.2005.08.008

Maclauchlan, L.E.; Brooks, J.E.; Hodge, J.C. 2006. Analysis of historic western spruce budworm defoliation in south central British Columbia. Forest Ecology and Management. 226(1-3): 351-356. doi:10.1016/j.foreco.2006.02.003

MacLean, D.A. 2004. Predicting forest insect disturbance regimes for use in emulating natural disturbance. In: Perera, A.H.; Buse, L.J.; Weber, M.G., eds. Emulating natural forest landscape disturbances: concepts and applications. New York: Columbia University Press: 69-82. isbn:0-231-12916-5

MacLean, C.D.; Bolsinger, C.L. 1973. Estimating productivity on sites with a low stocking capacity. Res. Pap. PNW-152. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 18 p. https://www.archive.org/download/CAT92273128/CAT92273128.pdf

MacQuarrie, C.J.K.; Cooke, B.J. 2011. Density-dependent population dynamics of mountain pine beetle in thinned and unthinned stands. Canadian Journal of Forest Research.

41(5): 1031-1046. doi:10.1139/x11-007

Magill, A.W. 1992. Managed and natural landscapes: what do people like? Res. Pap. PSW-RP-213. Albany, CA: USDA Forest Service, Pacific Southwest Research Station. 28 p. https://www.fs.usda.gov/treesearch/pubs/28952

Magnuson, J.J. 1990. Long-term ecological research and the invisible present. BioScience. 40(7): 495-501. doi:10.2307/1311317

Mahon, C.L.; Steventon, J.D.; Martin, K. 2008. Cavity and bark nesting bird response to partial cutting in northern conifer forests. Forest Ecology and Management. 256(12): 2145-2153. doi:10.1016/j.foreco.2008.08.005

Maissurow, D.K. 1935. Fire as a necessary factor in the perpetuation of white pine. Journal of Forestry. 33(4): 373-378. doi:10.1093/jof/33.4.373

Mäkelä, A.; del Río, M.; Hynynen, J.; Hawkins, M.J.; Reyer, C.; Soares, P.; van Oijen, M.; Tomé, M. 2012. Using stand-scale forest models for estimating indicators of sustainable forest management. Forest Ecology and Management. 285: 164-178. doi:10.1016/j.foreco.2012.07.041

Malamud, B.D.; Millington, J.D.A.; Perry, G.L.W. 2005. Characterizing wildfire regimes in the United States. Proceedings of the National Academy of Sciences of the United States of America. 102(13): 4694-4699. doi:10.1073/pnas.0500880102

Malcolm, D.C.; Mason, W.L.; Clarke, G.C. 2001. The transformation of conifer forests in Britain – regeneration, gap size and silvicultural systems. Forest Ecology and Management. 151(1-3): 7-23. doi:10.1016/S0378-1127(00)00692-7

Malkinson, D.; Wittenberg, L. 2011. Post fire induced soil water repellency—Modeling short and long-term processes. Geomorphology. 125(1): 186-192. doi:10.1016/j.geomorph.2010.09.014

Mallams, K.M.; Chadwick, K.L.; Angwin, P.A. 2010. Decays of white, grand, and red firs. Forest Insect & Disease Leaflet 52. Portland, OR: USDA Forest Service, Pacific Northwest Region. 12 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_042842.pdf

Malmsheimer, R.W.; Heffernan, P.; Brink, S.; Crandall, D.; Deneke, F.; Galik, C.; Gee, E.; Helms, J.A.; McClure, N.; Mortimer, M.; Ruddell, S.; Smith, M.; Stewart, J. 2008. Forest management solutions for mitigating climate change in the United States. Journal of Forestry. 106(3): 115-171. doi:10.1093/jof/106.3.115

Maloney, P.E.; Smith, T.F.; Jensen, C.E.; Innes, J., Rizzo, D.M.; North, M.P. 2008. Initial tree mortality and insect and pathogen response to fire and thinning restoration treatments in an old-growth mixed-conifer forest of the Sierra Nevada, California. Canadian Journal of Forest Research. 38(12): 3011-3020. doi:10.1139/X08-141

Maloy, O.C.; Gross, H.L. 1963. Decay in young grand fir. Journal of Forestry. 61(11): 850-853. doi:10.1093/jof/61.11.850

Man, R.; Kayahara, G.J.; Dang, Q.-L.; Rice, J.A. 2009. A case of severe frost damage prior to budbreak in young conifers in northeastern Ontario: Consequence of climate change? Forestry Chronicle. 85(3): 453-462. doi:10.5558/tfc85453-3

Mandzak, J.M.; Moore, J.A. 1994. The role of nutrition in the health of inland western forests. Journal of Sustainable Forestry. 2(1/2): 191-210. doi:10.1300/J091v02n01_09

Manley, P.N.; Brogan, G.E.; Cook, C.; Flores, M.E.; Fullmer, D.G.; Husari, S.; Jimerson, T.M.; Lux, L.M.; McCain, M.E.; Rose, J.A.; Schmitt, G.; Schuyler, J.C.; Skinner, M.J. 1995. Sustaining ecosystems: a conceptual framework. Tech. Pub. R5-EM-TP-001. San Francisco, CA: USDA Forest Service, Pacific Southwest Region. 216 p. https://archive.org/download/CAT10857790/CAT10857790.pdf

Mann, C.C. 2006. 1491: new revelations of the Americas before Columbus. New York: Vintage Books. 538 p. isbn:1-4000-3205-9

Mann, L.K.; Johnson, D.W.; West, D.C.; Cole, D.W.; Hornbeck, J.W.; Martin, C.W.; Riekerk, H.; Smith, C.T.; Swank, W.T.; Tritton, L.M.; Van Lear, D.H. 1988. Effects of whole-tree and stem-only clearcutting on postharvest hydrologic losses, nutrient capital, and regrowth. Forest Science. 34(2): 412-428. doi:10.1093/forestscience/34.2.412

Mann, M.E.; Bradley, R.S.; Hughes, M.K. 1999. Northern hemisphere temperatures during the past millennium: inferences, uncertainties, and limitations. Geophysical Research Letters. 26(6): 759-762. doi:10.1029/1999gl900070

Mannan, R.W.; Meslow, E.C. 1984. Bird populations and vegetation characteristics in managed and old-growth forests, northeastern Oregon. Journal of Wildlife Management. 48(4): 1219-1238. doi:10.2307/3801783

Marcot, B.G.; Wisdom, M.J.; Li, H.W.; Castillo, G.C. 1994. Managing for featured, threatened, endangered, and sensitive species and unique habitats for ecosystem sustainability. Gen. Tech. Rep. PNW-GTR-329. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 39 p. https://www.fs.usda.gov/treesearch/pubs/6406

Marcoux, H.M.; Gergel, S.E.; Daniels, L.D. 2013. Mixed-severity fire regimes: How well are they represented by existing fire-regime classification systems? Canadian Journal of Forest Research. 43(7): 658-668. doi:10.1139/cjfr-2012-0449

Marlon, J.R.; Bartlein, P.J.; Carcaillet, C.; Gavin, D.G.; Harrison, S.P.; Higuera, P.E.; Joos, F.; Power, M.J.; Prentice, I.C. 2008. Climate and human influences on global biomass burning over the past two millennia. Nature Geoscience. 1(10): 697-702. doi:10.1038/ngeo313

Marlon, J.R.; Bartlein, P.J.; Walsh, M.K.; Harrison, S.P.; Brown, K.J.; Edwards, M.E.; Higuera, P.E.; Power, M.J.; Anderson, R.S.; Briles, C.; and others. 2009. Wildfire responses to abrupt climate change in North America. Proceedings of the National Academy of Sciences. 106(8): 2519-2524. doi:10.1073/pnas.0808212106

Marlon, J.R.; Bartlein, P.J.; Gavin, D.G.; Long, C.J.; Anderson, R.S.; Briles, C.E.; Brown, K.J.; Colombaroli, D.; Hallett, D.J.; Power, M.J.; and others. 2012. Long-term perspective on wildfires in the western USA. Proceedings of the National Academy of Sciences. 109(9): E535-E543. doi:10.1073/pnas.1112839109

Maron, M.; Simmonds, J.S.; Watson, J.E.M.; Sonter, L.J.; Bennun, L.; Griffiths, V.F.; Quétier, F.; von Hase, A.; Edwards, S.; Rainey, H.; Bull, J.W.; Savy, C.E.; Victurine, R.; Kiesecker, J.; Puydarrieux, P.; Stevens, T.; Cozannet, N.; Jones, J.P.G. 2020. Global no net loss of natural ecosystems. Nature Ecology & Evolution. 4(1): 46-49. doi:10.1038/s41559-019-1067-z

Marris, E. 2009. Planting the forest of the future. Nature. 459(7249): 906-908. doi:10.1038/459906a

Marron, N.; Epron, D. 2019. Are mixed-tree plantations including a nitrogen-fixing species more productive than monocultures? Forest Ecology and Management. 441: 242-252. doi:10.1016/j.foreco.2019.03.052

Marsh, A.; Mawdsley, J.; Negra, C. 2009. Forest observations and indicators needed to respond to climate change. Journal of Forestry. 107(5): 231-232. doi:10.1093/jof/107.5.231

Marshall, V.G. 2000. Impacts of forest harvesting on biological processes in northern forest soils. Forest Ecology and Management. 133(1-2): 43-60. doi:10.1016/S0378-1127(99)00297-2

Marshall, M.R.; DeCecco, J.A.; Williams, A.B.; Gale, G.A.; Cooper, R.J. 2003. Use of regenerating clearcuts by late-successional bird species and their young during the post-fledging period. Forest Ecology and Management. 183(1-3): 127-135. doi:10.1016/S0378-1127(03)00101-4

Martin, P.H. 1996. Will forest preserves protect temperate and boreal biodiversity from climate change? Forest Ecology and Management. 85(1-3): 335-341. doi:10.1016/S0378-1127(96)03767-X

Martinson, E.J.; Omi, P.N. 2013. Fuel treatments and fire severity: A meta-analysis. Res. Pap. RMRS-RP-103WWW. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 38 p. http://www.treesearch.fs.fed.us/pubs/43632

Maruoka, K.R. 1994. Fire history of *Pseudotsuga menziesii* and *Abies grandis* stands in the Blue Mountains of Oregon and Washington. M.S. thesis. Seattle, WA: University of Washington, College of Forest Resources. 73 p. https://www.fs.fed.us/rm/pubs/rmrs_gtr292/1994_maruoka.pdf

Marvel, C.C. 1988. Creating better looking national forests using natural-appearing cutting patterns. Western Journal of Applied Forestry. 3(4): 131-132. doi:10.1093/wjaf/3.4.131

Marzluff, J.M.; Raphael, M.G.; Sallabanks, R. 2000. Understanding the effects of forest management on avian species. Wildlife Society Bulletin. 28(4): 1132-1143. http://www.jstor.org/stable/3783874

Mason, D.T. 1915a. The life history of lodgepole pine in the Rocky Mountains. Bull. No. 154. Washington, DC: U.S. Department of Agriculture. 35 p. https://www.biodiversitylibrary.org/itempdf/131367

Mason, D.T. 1915b. Utilization and management of lodgepole pine in the Rocky Mountains. Bull. No. 234. Washington, DC: U.S. Department of Agriculture. 54 p. Utilization_and_Management_of_Lodgepole

Mason, R.R. 1992. Populations of arboreal spiders (Araneae) on Douglas-firs and true firs in the interior Pacific Northwest. Environmental Entomology. 21(1): 75-80. doi:10.1093/ee/21.1.75

Mason, R.R.; Wickman, B.E. 1984. The Douglas-fir tussock moth: ecological relationships, impact, and hazard reduction. In: Baumgartner, D.M.; Mitchell, R., comps. Silvicultural management strategies for pests of the interior Douglas-fir and grand fir forest types. Pullman, WA: Washington State University, Cooperative Extension: 95-102.

Mason, R.R.; Torgersen, T.R.; Wickman, B.E.; Paul, H.G. 1983. Natural regulation of a Douglas-fir tussock moth (Lepidoptera: Lymantriidae) population in the Sierra Nevada. Environmental Entomology. 12(2): 587-594. doi:10.1093/ee/12.2.587

Mason, R.R.; Wickman, B.E.; Paul, H.G.; Torgersen, T.R. 1998. A pilot experiment of forest fertilization during an outbreak of the western spruce budworm in northeastern Oregon. Res. Pap. PNW-RP-506. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 25 p. http://www.treesearch.fs.fed.us/pubs/26879

Mason, C.L.; Lippke, B.R.; Zobrist, K.W.; Bloxton, T.D.; Ceder, K.R.; Comnick, J.M.; McCarter, J.B.; Rogers, H.K. 2006. Investments in fuel removals to avoid forest fires result in substantial benefits. Journal of Forestry. 104(1): 27-31. doi:10.1093/jof/104.1.27

Mason, L.; White, G.; Morishima, G.; Alvarado, E.; Andrew, L.; Clark, F.; Durglo, M.; Durglo, J.; Eneas, J.; Erickson, J.; Friedlander, M.; Hamel, K.; Hardy, C.; Harwood, T.; Haven, F.; Isaac, E.; James, L.; Kenning, R.; Leighton, A.; Pierre, P.; Raish, C.; Shaw, B.; Smallsalmon, S.; Stearns, V.; Teasley, H.; Weingart, M.; Wilder, S. 2012. Listening and learning from traditional knowledge and western science: A dialogue on contemporary challenges of forest health and wildfire. Journal of Forestry. 110(4): 187-193. doi:10.5849/jof.11-006

Mataix-Solera, J.; Cerdà, A.; Arcenegui, V.; Jordán, A.; Zavala, L.M. 2011. Fire effects on soil aggregation: A review. Earth-Science Reviews. 109(1): 44-60. doi:10.1016/j.earscirev.2011.08.002

Mathiasen, R.L.; Nickrent, D.L.; Shaw, D.C.; Watson, D.M. 2008. Mistletoes: pathology, systematics, ecology, and management. Plant Disease. 92(7): 988-1006.

doi:10.1094/PDIS-92-7-0988

Mattson, W.J. 1996. Escalating arthropod stresses on forest ecosystems: forcing benign plant-insect interactions into new interaction trajectories. In: Korpilahti, E.; Mikkela, H.; Salonen, T., eds. Caring for the forests: research in a changing world. Congress Report, vol. 2. Finland: IUFRO World Congress Organizing Committee: 223-236.

Mattson, W.J.; Addy, N.D. 1975. Phytophagous insects as regulators of forest primary production. Science. 190(4214): 515-522. doi:10.1126/science.190.4214.515

Mattson, W.J.; Haack, R.A. 1987. The role of drought in outbreaks of plant-eating insects. BioScience. 37(2): 110-118. doi:10.2307/1310365

Maxwell, J.D.; Call, A.; St. Clair, S.B. 2019. Wildfire and topography impacts on snow accumulation and retention in montane forests. Forest Ecology and Management. 432: 256-263. doi:10.1016/j.foreco.2018.09.021

Mayer, A.L.; Rietkerk, M. 2004. The dynamic regime concept for ecosystem management and restoration. BioScience. 54(11): 1013-1020. doi:10.1641/0006-3568(2004)054[1013:TDRCFE]2.0.CO;2

Mazurek, M.J.; Zielinski, W.J. 2004. Individual legacy trees influence vertebrate wildlife diversity in commercial forests. Forest Ecology and Management. 193(3): 321-334. doi:10.1016/j.foreco.2004.01.013

McAfee, B.J.; de Camino, R.; Burton, P.J.; Eddy, B.; Fähser, L.; Messier, C.; Reed, M.G.; Spies, T.; Vides, R. 2010. Managing forested landscapes for socio-ecological resilience. In: Mery, G.; Katila, P.; Galloway, G.; Alfaro, R.I.; Kanninen, M.; Lobovikov, M.; Varjo, J., eds. Forests and society – responding to global drivers of change. IUFRO World Series Volume 25. Vienna, Austria: International Union of Forest Research Organizations: 401-439. https://www.iufro.org/science/special/wfse/forests-society-global-drivers/

McCaffrey, S.; Toman, E.; Stidham, M.; Shindler, B. 2013. Social science research related to wildfire management: an overview of recent findings and future research needs. International Journal of Wildland Fire. 22(1): 15-24. doi:10.1071/WF11115

McCann, K.S. 2000. The diversity-stability debate. Nature. 405(6783): 228-233. doi:10.1038/35012234

McCaskill, G.L. 2019. The Hungry Bob fire & fire surrogate study: A 20-year evaluation of the treatment effects. Forests. 10(1): 15 (12 p). doi:10.3390/f10010015

McCaughey, W.W.; Schmidt, W.C. 1982. Understory tree release following harvest cutting in spruce-fir forests of the Intermountain West. Res. Pap. INT-285. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 19 p. https://archive.org/download/understorytreere285mcca/understorytreere285mcca.pdf

McCaughey, W.W.; Schmidt, W.C. 1987. Seed dispersal of Engelmann spruce in the Intermountain West. Northwest Science. 61(1): 1-6. McCaugheyandSchmidt

McCaughey, W.W.; Schmidt, W.C.; Shearer, R.C. 1986. Seed-dispersal characteristics of conifers in the inland mountain West. In: Shearer, R.C., comp. Proceedings – conifer tree seed in the inland mountain west symposium. Gen. Tech. Rep. INT-203. Ogden, UT: USDA Forest Service, Intermountain Research Station: 50-62. https://archive.org/download/CAT86866655/CAT86866655.pdf

McCaughey, W.W.; Fiedler, C.E.; Schmidt, W.C. 1991. Twenty-year natural regeneration following five silvicultural prescriptions in spruce-fir forests of the Intermountain West. Res. Pap. INT-439. Ogden, UT: USDA Forest Service, Intermountain Research Station. 12 p. Twenty_year_Natural_Regeneration_Followi

McClanahan, T.R.; Wolfe, R.W. 1993. Accelerating forest succession in a fragmented landscape: the role of birds and perches. Conservation Biology. 7(2): 279-288. doi:10.1046/j.1523-1739.1993.07020279.x

McComb, B.C. 2008. Wildlife habitat management: concepts and applications in forestry. Boca Raton, FL: CRC Press. 319 p. isbn:0-8493-7489-8

McConnell, B.R.; Smith, J.G. 1971. Effect of ponderosa pine needle litter on grass seedling survival. Res. Note PNW-155. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 6 p. https://www.fs.usda.gov/treesearch/pubs/26376

McCullough, H.A. 1948. Plant succession on fallen logs in a virgin spruce-fir forest. Ecology. 29(4): 508-513. doi:10.2307/1932645

McCullough, D.G.; Werner, R.A.; Neumann, D. 1998. Fire and insects in northern and boreal forest ecosystems of North America. Annual Review of Entomology. 43(1): 107-127. doi:10.1146/annurev.ento.43.1.107

McCune, B. 1983. Fire frequency reduced two orders of magnitude in the Bitterroot canyons, Montana. Canadian Journal of Forest Research. 13(2): 212-218. doi:10.1139/x83-030

McCune, B. 1988. Ecological diversity in North American pines. American Journal of Botany. 75(3): 353-368. doi:10.1002/j.1537-2197.1988.tb13450.x

McCune, B.; Allen, T.F.H. 1985a. Forest dynamics in the Bitterroot Canyons, Montana. Canadian Journal of Botany. 63(3): 377-383. doi:10.1139/b85-044

McCune, B.; Allen, T.F.H. 1985b. Will similar forests develop on similar sites? Canadian Journal of Botany. 63(3): 367-376. doi:10.1139/b85-043

McCune, B.; Antos, J.A. 1981. Correlations between forest layers in the Swan Valley, Montana. Ecology. 62(5): 1196-1204. doi:10.2307/1937284

McDaniel, P.A.; Hipple, K.W. 2010. Mineralogy of loess and volcanic ash eolian mantles in Pacific Northwest (USA) landscapes. Geoderma. 154(3-4): 438-446. doi:10.1016/j.geoderma.2008.12.023

McDaniel, P.A.; Wilson, M.A. 2007. Physical and chemical characteristics of ash-influenced soils of inland Northwest forests. In: Page-Dumroese, D.; Miller, R.; Mital, J.; McDaniel, P.; Miller, D., tech. eds. Volcanic-ash-derived forest soils of the inland Northwest: properties and implications for management and restoration. Proceedings RMRS-P-44. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station: 31-45. https://www.fs.usda.gov/treesearch/pubs/26202

McDaniel, P.A.; Wilson, M.A.; Burt, R.; Lammers, D.; Thorson, T.D.; McGrath, C.L.; Peterson, N. 2005. Andic soils of the inland Pacific Northwest, USA: properties and ecological significance. Soil Science. 170(4): 300-311. doi:10.1097/01.ss.0000162293.71373.32

McDonald, G.I. 1991. Relationships among site quality, stand structure, and *Armillaria* root rot in Douglas-fir forests. In: Baumgartner, D.M.; Lotan, J.E., comps. Interior Douglas-fir: the species and its management; symposium proceedings. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 35-46.

McDonald, G.I.; Harvey, A.E.; Tonn, J.R. 2000. Fire, competition and forest pests: landscape treatment to sustain ecosystem function. In: Crossing the millenium: integrating spatial technologies and ecological principles for a new age in fire management: proceedings, the Joint Fire Science Conference and Workshop; volume 1. Moscow, ID: University of Idaho: 195-211. https://www.fs.usda.gov/treesearch/pubs/23565

McDowell, N.G.; Adams, H.D.; Bailey, J.D.; Hess, M.; Kolb, T.E. 2006. Homeostatic maintenance of ponderosa pine gas exchange in response to stand density changes. Ecological Applications. 16(3): 1164-1182. doi:10.1890/1051-0761(2006)016[1164:HMOPPG]2.0.CO;2

McDowell, N.; Pockman, W.T.; Allen, C.D.; Breshears, D.D.; Cobb, N.; Kolb, T.; Plaut, J.; Sperry, J.; West, A.; Williams, D.G.; Yepez, E.A. 2008. Mechanisms of plant survival and mortality during drought: why do some plants survive while others succumb to drought? New Phytologist. 178(4): 719-739. doi:10.1111/j.1469-8137.2008.02436.x

McElhinny, C.; Gibbons, P.; Brack, C.; Bauhus, J. 2005. Forest and woodland stand structural complexity: its definition and measurement. Forest Ecology and Management. 218(1-3): 1-24. doi:10.1016/j.foreco.2005.08.034

McIlroy, S.K.; Shinneman, D.J. 2020. Post-fire aspen (*Populus tremuloides*) regeneration varies in response to winter precipitation across a regional climate gradient. Forest Ecology and Management. 455: 117681 (9 p). doi:10.1016/j.foreco.2019.117681

McIntosh, B.A.; Sedell, J.R.; Smith, J.E.; Wissmar, R.C.; Clarke, S.E.; Reeves, G.H.; Brown, L.A. 1994. Management history of eastside ecosystems: changes in fish habitat over 50 years, 1935 to 1992. Gen. Tech. Rep. PNW-GTR-321. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 55 p. http://www.treesearch.fs.fed.us/pubs/6232

McKenney, D.W.; Pedlar, J.H.; Lawrence, K.; Campbell, K.; Hutchinson, M.F. 2007. Potential impacts of climate change on the distribution of North American trees. BioScience. 57(11): 939-948. doi:10.1641/B571106

McKenzie, D.; Kennedy, M.C. 2011. Scaling laws and complexity in fire regimes. In: McKenzie, D.; Miller, C.; Falk, D.A., eds. The landscape ecology of fire. New York: Springer: 27-49. isbn:978-94-007-0300-1

McKenzie, D.; Peterson, D.L.; Agee, J.K. 2000. Fire frequency in the interior Columbia River basin: building regional models from fire history data. Ecological Applications. 10(5): 1497-1516. doi:10.1890/1051-0761(2000)010[1497:FFITIC]2.0.CO;2

McKenzie, D.; Gedalof, Z.; Peterson, D.L.; Mote, P. 2004. Climatic change, wildfire, and conservation. Conservation Biology. 18(4): 890-902. doi:10.1111/j.1523-1739.2004.00492.x

McKenzie, D.; Miller, C.; Falk, D.A., eds. 2011a. The landscape ecology of fire. New York: Springer. 312 p. isbn:978-94-007-0300-1

McKenzie, D.; Miller, C.; Falk, D.A. 2011b. Toward a theory of landscape fire. In: McKenzie, D.; Miller, C.; Falk, D.A., eds. The landscape ecology of fire. New York: Springer: 3-25. isbn:978-94-007-0300-1

McKetta, C.; Blatner, K.A.; Graham, R.T.; Erickson, J.; Hamilton, S.S. 1994. Human dimensions of forest health choices. Journal of Sustainable Forestry. 2(1-2): 135-149. doi:10.1300/J091v02n01_06

McKinley, D.C.; Briggs, R.D.; Bartuska, A.M. 2012. When peer-reviewed publications are not enough! Delivering science for natural resource management. Forest Policy and Economics. 21: 1-11. doi:10.1016/j.forpol.2012.03.007

McLean, H.E. 1992. The Blue Mountains: forest out of control. American Forests. 98(9/10): 32, 34-35, 58, 61.

McMahon, S.M.; Arellano, G.; Davies, S.J. 2019. The importance and challenges of detecting changes in forest mortality rates. Ecosphere. 10(2): e02615 (10 p). doi:10.1002/ecs2.2615

McMinn, R.G. 1952. The role of soil drought in the distribution of vegetation in the northern Rocky Mountains. Ecology. 33(1): 1-15. doi:10.2307/1931247

McPherson, G.R.; DeStefano, S. 2003. Applied ecology and natural resource management. Cambridge, UK: Cambridge University Press. 165 p. isbn:0-521-00975-8

McRae, D.J.; Duchesne, L.C.; Freedman, B.; Lynham, T.J.; Woodley, S. 2001. Comparisons between wildfire and forest harvesting and their implications in forest management. Environmental Reviews. 9(4): 223-260. doi:10.1139/a01-010

McRoberts, R.E.; Winter, S.; Chirici, G.; Hauk, E.; Pelz, D.R.; Moser, W.K.; Hatfield, M.A. 2008. Large-scale spatial patterns of forest structural diversity. Canadian Journal of Forest Research. 38(3): 429-438. doi:10.1139/x07-154

McRoberts, R.E.; Winter, S.; Chirici, G.; LaPoint, E. 2012. Assessing forest naturalness. Forest Science. 58(3): 294-309. doi:10.5849/forsci.10-075

Meadows, P.S.; Meadows, A. 1991. The environmental impact of burrowing animals and animal burrows. Zoological Society of London Symposia 63. New York: Oxford Science Publications. 349 p. isbn:0-19-854680-7

Meddens, A.J.H.; Hicke, J.A.; Ferguson, C.A. 2012. Spatiotemporal patterns of observed bark beetle-caused tree mortality in British Columbia and the western United States. Ecological Applications. 22(7): 1876-1891. doi:10.1890/11-1785.1

Meddens, A.J.H.; Kolden, C.A.; Lutz, J.A. 2016. Detecting unburned areas within wildfire perimeters using Landsat and ancillary data across the northwestern United States. Remote Sensing of Environment. 186: 275-285. doi:10.1016/j.rse.2016.08.023

Meffe, G.K.; Carroll, C.R., and contributors. 1997. Principles of conservation biology. 2nd edition. Sunderland, MA: Sinauer Associates, Inc. 729 p. isbn:0-87893-521-5

Meffe, G.K.; Boersma, P.D.; Murphy, D.D.; Noon, B.R.; Pulliam, H.R.; Soule, M.E.; Waller, D.M. 1998. Independent scientific review in natural resource management. Conservation Biology. 12(2): 268-270. doi:10.1046/j.1523-1739.1998.012002268.x

Meffe, G.K.; Nielsen, L.A.; Knight, R.L.; Schenborn, D.A. 2002. Ecosystem management: adaptive, community-based conservation. Washington, DC: Island Press. 313 p. isbn:978-1-55963-824-1

Mehringer, P.J., Jr. 1997. Late Holocene fire and forest history from Lost Lake, Umatilla National Forest, Blue Mountains, Oregon. Final report for Challenge Cost-Share Agreement No. CCS-06-95-04-058. John Day, OR: USDA Forest Service, Malheur National Forest. 29 p.

Meigs, G.; Donato, D.; Campbell, J.; Martin, J.; Law, B. 2009. Forest fire impacts on carbon uptake, storage, and emission: the role of burn severity in the eastern Cascades, Oregon. Ecosystems. 12(8): 1246-1267. doi:10.1007/s10021-009-9285-x

Meigs, G.W.; Kennedy, R.E.; Cohen, W.B. 2011. A Landsat time series approach to characterize bark beetle and defoliator impacts on tree mortality and surface fuels in conifer forests. Remote Sensing of Environment. 115(12): 3707-3718. doi:10.1016/j.rse.2011.09.009

Meigs, G.W.; Kennedy, R.E.; Gray, A.N.; Gregory, M.J. 2015. Spatiotemporal dynamics of recent mountain pine beetle and western spruce budworm outbreaks across the Pacific Northwest Region, USA. Forest Ecology and Management. 339: 71-86. doi:10.1016/j.foreco.2014.11.030

Meigs, G.W.; Zald, H.S.J.; Campbell, J.L.; Keeton, W.S.; Kennedy, R.E. 2016. Do insect outbreaks reduce the severity of subsequent forest fires? Environmental Research Letters. 11(4): 045008 (10 p). doi:10.1088/1748-9326/11/4/045008

Meinecke, E.P. 1916. Forest pathology in forest regulation. Bull. No. 275. Washington, DC: U.S. Department of Agriculture. 62 p. https://archive.org/download/forestpathologyi275mein/forestpathologyi275mein.pdf

Merritt, M.L. 1911. Lodgepole pine in the Whitman National Forest. Unpublished typescript report obtained from the National Archives, College Park, MD; record group 95. 18 p. On file with: USDA Forest Service, Umatilla National Forest, Supervisor's Office, Pendleton, OR. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015534.pdf

Merschel, A.G. 2012. Mixed-conifer forests of central Oregon: structure, composition, history of establishment, and growth. M.S. thesis. Corvallis, OR: Oregon State University, Department of Forest Ecosystems and Society. 149 p. http://scholarsarchive.library.oregonstate.edu/xmlui/handle/1957/36236

Mery, G.; Katila, P.; Galloway, G.; Alfaro, R.I.; Kanninen, M.; Lobovikov, M.; Varjo, J., eds. 2010. Forests and society – responding to global drivers of change. IUFRO World Series Volume 25. Vienna, Austria: International Union of Forest Research Organizations. 509 p. https://www.iufro.org/science/special/wfse/forests-society-global-drivers/

Meslow, E.C.; Maser, C.; Verner, J. [1981]. Old-growth forests as wildlife habitat. In: Transactions of the 46th North American Wildlife and Natural Resources Conference. Washington, DC: Wildlife Management Institute: 329-335.

Messier, M.S.; Shatford, J.P.A.; Hibbs, D.E. 2012. Fire exclusion effects on riparian forest dynamics in southwestern Oregon. Forest Ecology and Management. 264: 60-71. doi:10.1016/j.foreco.2011.10.003

Messier, C.; Puettmann, K.J.; Coates, K.D. 2013. Managing forests as complex adaptive systems: Building resilience to the challenge of global change. Earthscan Forest Library. New York: Routledge. 353 p. isbn:978-0-415-51977-9

Meurisse, R.T.; Robbie, W.A.; Niehoff, J.; Ford, G. 1991. Dominant soil formation processes and properties in western-montane forest types and landscapes–some implications for productivity and management. In: Harvey, A.E.; Neuenschwander, L.F., comps. Proceedings–management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: USDA Forest Service, Intermountain Research Station: 7-19. http://www.treesearch.fs.fed.us/pubs/33908

Meyer, J.L. 1997. Conserving ecosystem function. In: Pickett, S.T.A.; Ostfeld, R.S.; Shachak, M.; Likens, G.E., eds. The ecological basis of conservation: heterogeneity, ecosystems, and biodiversity. New York: Chapman & Hall: 136-145. isbn:0-412-09851-2

Meyer, J.L.; Helfman, G.S. 1993. The ecological basis of sustainability. Ecological Applications. 3(4): 569-571. doi:10.2307/1942082

Meyer, M.D.; North, M.P.; Gray, A.N.; Zald, H.S.J. 2007. Influence of soil thickness on stand characteristics in a Sierra Nevada mixed-conifer forest. Plant and Soil. 294(1-2): 113-123. doi:10.1007/s11104-007-9235-3

Meyer, K.E.; Dwire, K.A.; Champ, P.A.; Ryan, S.E.; Riegel, G.M.; Burton, T.A. 2012. Burning questions for managers: fuels management practices in riparian areas. Fire Management Today. 72(2): 16-23. https://www.fs.usda.gov/treesearch/pubs/41413

Michaletz, S.T.; Johnson, E.A. 2007. How forest fires kill trees: A review of the fundamental biophysical processes. Scandinavian Journal of Forest Research. 22(6): 500-515. doi:10.1080/02827580701803544

Miesel, J.R. 2009. Restoring mixed-conifer forests with fire and mechanical thinning: effects on soil properties and mature conifer foliage. PhD dissertation. Columbus, OH: Ohio State University. 190 p. Miesel 2009 dissertation

Miesel, J.R.; Boerner, R.E.J.; Skinner, C.N. 2008. Mechanical restoration of California mixed-conifer forests: Does it matter which trees are cut? Restoration Ecology. 17(6): 784-795. doi:10.1111/j.1526-100X.2008.00414.x

Miesel, J.R.; Hockaday, W.C.; Kolka, R.K.; Townsend, P.A. 2015. Soil organic matter composition and quality across fire severity gradients in coniferous and deciduous forests of the southern boreal region. Journal of Geophysical Research: Biogeosciences. 120(6): 1124-1141. doi:10.1002/2015JG002959

Mika, P.; Moore, J. 1990. Foliar potassium status explains Douglas fir response to nitrogen fertilization in the inland Northwest, USA. Water, Air, and Soil Pollution. 54(3): 477-491. doi:10.1007/bf02385240

Mikkelson, K.M.; Dickenson, E.R.V.; Maxwell, R.M.; McCray, J.E.; Sharp, J.O. 2013. Water-quality impacts from climate-induced forest die-off. Nature Climate Change. 3(3): 218-222. doi:10.1038/nclimate1724

Mildrexler, D.J.; Berner, L.T.; Law, B.E.; Birdsey, R.A.; Moomaw, W.R. 2020. Large trees dominate carbon storage in forests east of the Cascade crest in the United States Pacific Northwest. Frontiers in Forests and Global Change. 3(127): 594274 (15 p). doi:10.3389/ffgc.2020.594274

Millar, C.I.; Woolfenden, W.B. 1999. The role of climate change in interpreting historical variability. Ecological Applications. 9(4): 1207-1216. doi:10.1890/1051-0761(1999)009[1207:TROCCI]2.0.CO;2

Miller, C.; Urban, D.L. 2000a. Connectivity of forest fuels and surface fire regimes. Landscape Ecology. 15(2): 145-154. doi:10.1023/A:1008181313360

Miller, C.; Urban, D.L. 2000b. Modeling the effects of fire management alternatives on Sierra Nevada mixed-conifer forests. Ecological Applications. 10(1): 85-94. doi:10.1890/1051-0761(2000)010[0085:MTEOFM]2.0.CO;2

Miller, J.; Safford, H.; Crimmins, M.; Thode, A. 2009. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. Ecosystems. 12(1): 16-32. doi:10.1007/s10021-008-9201-9

Miller, J.E.D.; Villella, J.; Stone, D.; Hardman, A. 2020. Using lichen communities as indicators of forest stand age and conservation value. Forest Ecology and Management. 475: 118436 (6 p). doi:10.1016/j.foreco.2020.118436

Mills, T.J.; Clark, R.N. 2001. Roles of research scientists in natural resource decision-making. Forest Ecology and Management. 153(1-3): 189-198. doi:10.1016/S0378-1127(01)00461-3

Milly, P.C.D.; Betancourt, J.; Falkenmark, M.; Hirsch, R.M.; Kundzewicz, Z.W.; Lettenmaier, D.P.; Stouffer, R.J. 2008. Stationarity is dead: Whither water management? Science. 319(5863): 573-574. doi:10.1126/science.1151915

Minckler, L.S. 1974. Prescribing silvicultural systems. Journal of Forestry. 72(5): 269-273. doi:10.1093/jof/72.5.269

Ministry of Agriculture and Forestry. 2009. Standards and guidelines for the sustainable management of indigenous forests. Fourth ed. Wellington, New Zealand: Ministry of Agriculture and Forestry, MAF Policy. 219 p.

Minore, D. 1979. Comparative autecological characteristics of northwestern tree species – a literature review. Gen. Tech. Rep. PNW-87. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 72 p. http://www.treesearch.fs.fed.us/pubs/25175

Mitchell, R.G. 1989. Mixed host strategies for mountain pine beetle control in Oregon. In: Amman, G.D., comp. Proceedings–symposium on the management of lodgepole pine to minimize losses to the mountain pine beetle. Gen. Tech. Rep. INT-262. Ogden, UT: USDA Forest Service, Intermountain Research Station: 60-63. https://archive.org/download/CAT90949101/CAT90949101.pdf

Mitchell, J.L. 1994. Commercial thinning of mature lodgepole pine to reduce susceptibility to mountain pine beetle. FRDA Report 224. Victoria, BC: Canadian Forest Service. 20 p. https://www.for.gov.bc.ca/hfd/pubs/docs/frr/Frr224.pdf

Mitchell, S.J. 1995. The windthrow triangle: a relative windthrow hazard assessment procedure for forest managers. Forestry Chronicle. 71(4): 446-450. doi:10.5558/tfc71446-4

Mitchell, S.J. 2013. Wind as a natural disturbance agent in forests: a synthesis. Forestry. 86(2): 147-157. doi:10.1093/forestry/cps058

Mitchell, R.G.; Buffam, P.E. 2001. Patterns of long-term balsam woolly adelgid infestation and effects in Oregon and Washington. Western Journal of Applied Forestry. 16(3): 121-126. doi:10.1093/wjaf/16.3.121

Mitchell, R.G.; Martin, R.E. 1980. Fire and insects in pine culture of the Pacific Northwest. In: Martin, R.E.; Edmonds, R.L.; Faulkner, D.A.; Harrington, J.B.; Fuquay, D.M.; Stocks, B.J.; Barr, S., eds. Proceedings: sixth conference on fire and forest meteorology. Washington, DC: Society of American Foresters: 182-190.

Mitchell, R.G.; Preisler, H.K. 1991. Analysis of spatial patterns of lodgepole pine attacked by outbreak populations of the mountain pine beetle. Forest Science. 37(5): 1390-1408. doi:10.1093/forestscience/37.5.1390

Mitchell, R.G.; Waring, R.H.; Pitman, G.B. 1983. Thinning lodgepole pine increases tree vigor and resistance to mountain pine beetle. Forest Science. 29(1): 204-211. doi:10.1093/forestscience/29.1.204

Mitchell, R.J.; Palik, B.J.; Hunter, M.L., Jr. 2002. Natural disturbance as a guide to silviculture. Forest Ecology and Management. 155(1-3): 315-317. doi:10.1016/S0378-1127(01)00568-0

Mitchell, S.R.; Harmon, M.E.; O'Connell, K.E.B. 2009. Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems. Ecological Applications. 19(3): 643-655. doi:10.1890/08-0501.1

Mladenoff, D.J.; Baker, W.L., eds. 1999. Spatial modeling of forest landscape change: approaches and applications. Cambridge, UK: Cambridge University Press. 352 p. isbn:0-521-63122-X

Mladenoff, D.J.; White, M.A.; Pastor, J. 1993. Comparing spatial pattern in unaltered old-growth and disturbed forest landscapes. Ecological Applications. 3(2): 294-306. doi:10.2307/1941832

Mladenoff, D.J.; White, M.A.; Crow, T.R.; Pastor, J. 1994. Applying principles of landscape design and management to integrate old-growth forest enhancement and commodity use. Conservation Biology. 8(3): 752-762. doi:10.1046/j.1523-1739.1994.08030752.x

Mlinsek, D. 1996. From clear-cutting to a close-to-nature silvicultural system. IUFRO News. 25(4): 6-8.

Mlot, C. 1991. Diversity and dwarf mistletoe. BioScience. 41(11): 755. doi:10.1093/bioscience/41.11.754

Mock, C.J. 1996. Climatic controls and spatial variations of precipitation in the western United States. Journal of Climate. 9(5): 1111-1125. doi:10.1175/1520-0442(1996)009<1111:CCASVO>2.0.CO;2

Mockta, T.K.; Fulé, P.Z.; Sánchez Meador, A.; Padilla, T.; Kim, Y.-S. 2018. Sustainability of culturally important teepee poles on Mescalero Apache Tribal Lands: Characteristics and climate change effects. Forest Ecology and Management. 430: 250-258. doi:10.1016/j.foreco.2018.08.017

Moerman, D.E. 2010. Native American ethnobotany. 9th edition. Portland, OR: Timber Press. 927 p. isbn:978-0-88192-453-4

Moghaddas, J.J.; Craggs, L. 2007. A fuel treatment reduces fire severity and increases suppression efficiency in a mixed conifer forest. International Journal of Wildland Fire. 16(6): 673-678. doi:10.1071/WF06066

Moghissi, A.A.; Love, B.R.; Straja, S.R.; McBride, D.K.; Swetnam, M.S. 2008. Best available science: its evolution, taxonomy, and application. Arlington, VA: Potomac Institute for Policy Studies. 93 p.

Monnig, E.; Byler, J. 1992. Forest health and ecological integrity in the northern Rockies. FPM Rep. 92-7. Missoula, MT: USDA Forest Service, Northern Region. 18 p.

Monserud, R.A.; Moody, U.; Breuer, D.W. 1990. A soil-site study for inland Douglas-fir. Canadian Journal of Forest Research. 20(6): 686-695. doi:10.1139/x90-092

Mooney, H.A.; Bonnicksen, T.M.; Christensen, N.L.; Lotan, J.E.; Reiners, W.A., tech. coords. 1981. Fire regimes and ecosystem properties; proceedings of the conference. Gen. Tech. Rep. WO-26. [Washington, DC]: USDA Forest Service. 594 p. https://archive.org/download/CAT83781017/CAT83781017.pdf

Moore, G.E. 1903. Principia ethica. Reprinted by several publishers in hardbound and paperback versions.

Moore, P.D. 2005. Roots of stability. Nature. 437(7061): 959-961. doi:10.1038/437959a

Moore, J.A.; Mika, P.G. 1997. Influence of soil parent material on the nutrition and health of established conifer stands in the inland northwest. In: Haase, D.L.; Rose, R., eds. Forest seedling nutrition from the nursery to the field. Corvallis, OR: Oregon State University, Department of Forest Science, Nursery Technology Cooperative: 112-117.

Moore, J.A.; Hamilton, D.A., Jr.; Xiao, Y.; Byrne, J. 2004. Bedrock type significantly affects individual tree mortality for various conifers in the inland Northwest, U.S.A. Canadian Journal of Forest Research. 34(1): 31-42. doi:10.1139/X03-196

Moore, J.A.; Mika, P.G.; Schwandt, J.W.; Shaw, T.M. 1994. Nutrition and forest health. In: Baumgartner, D.M.; Lotan, J.E.; Tonn, J.R., comps. Interior cedar-hemlock-white pine forests: ecology and management. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 157-163.

Moore, J.A.; Schenk, J.A.; Hatch, C.R. 1978. Validation and refinement of a hazard rating model for fir engraver-caused mortality in grand fir stands. Forest Science. 24(2): 309-312. doi:10.1093/forestscience/24.2.309

Moote, A. 2013. Closing the feedback loop: Evaluation and adaptation in collaborative resource management; a sourcebook for collaborative resource management groups. Flaggstaff, AZ: Northern Arizona University, Ecological Restoration Institute. 44 p. http://library.eri.nau.edu/gsdl/collect/erilibra/index/assoc/D2013003.dir/doc.pdf

Morgan, P. 2004. Back to the future: the value of history in understanding and managing dynamic landscapes. In: Gucinski, H.; Miner, C.; Bittner, B., eds. Proceedings: views from ridge – considerations for planning at the landscape scale. Gen. Tech. Rep. PNW-GTR-596. Portland, OR: USDA Forest Service, Pacific Northwest Research Station: 78-84. http://www.treesearch.fs.fed.us/pubs/6195

Morgan, P.; Parsons, R. 2001. Historical range of variability of forests of the Idaho southern batholith ecosystem. Unpub. Rep. Moscow, ID: University of Idaho, Department of Forest Resources. 34 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015498.pdf

Morgan, P.; Aplet, G.H.; Haufler, J.B.; Humphries, H.C.; Moore, M.M.; Wilson, W.D. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. Journal of Sustainable Forestry. 2(1-2): 87-111. doi:10.1300/J091v02n01_04

Morgan, P.; Bunting, S.C.; Black, A.E.; Merrill, T.; Barrett, S. 1998. Past and present fire regimes in the interior Columbia River basin. In: Fire management under fire (adapting to change); proceedings of the 1994 Interior West Fire Council meeting. Fairfield, WA: International Association of Wildland Fire: 77-82.

Morgan, P.; Hardy, C.C.; Swetnam, T.W.; Rollins, M.G.; Long, D.G. 2001. Mapping fire regimes across time and space: understanding coarse and fine-scale fire patterns. International Journal of Wildland Fire. 10(4): 329-342. doi:10.1071/WF01032

Morgan, P.; Heyerdahl, E.K.; Gibson, C.E. 2008. Multi-season climate synchronized forest fires throughout the 20th century, northern Rockies, USA. Ecology. 89(3): 717-728. doi:10.1890/06-2049.1

Morgantini, L.E.; Kansas, J.L. 2003. Differentiating mature and old-growth forests in the Upper Foothills and Subalpine Subregions of west-central Alberta. Forestry Chronicle. 79(3): 602-612. doi:10.5558/tfc79602-3

Moritz, M.A.; Stephens, S.L. 2008. Fire and sustainability: considerations for California's altered future climate. Climatic Change. 87(1): 265-271. doi:10.1007/s10584-007-9361-1

Moritz, M.A.; Hessburg, P.F.; Povak, N.A. 2011. Native fire regimes and landscape resilience. In: McKenzie, D.; Miller, C.; Falk, D.A., eds. The landscape ecology of fire. New York: Springer: 51-86. isbn:978-94-007-0300-1

Moritz, M.A.; Parisien, M.-A.; Batllori, E.; Krawchuk, M.A.; Van Dorn, J.; Ganz, D.J.; Hayhoe, K. 2012. Climate change and disruptions to global fire activity. Ecosphere. 3(6): art49. doi:10.1890/es11-00345.1

Morris, W.G. 1934. Lightning storms and fires on the national forests of Oregon and Washington. Monthly Weather Review. 62(10): 370-375. doi:10.1175/1520-0493(1934)62%3C370:LSAFOT%3E2.0.CO;2

Morris, D.M. 1997. The role of long-term site productivity in maintaining healthy ecosystems: a prerequisite of ecosystem management. Forestry Chronicle. 73(6): 731-740. doi:10.5558/tfc73731-6

Morrison, D.; Mallett, K. 1996. Silvicultural management of armillaria root disease in western Canadian forests. Canadian Journal of Plant Pathology. 18(2): 194-199. doi:10.1080/07060669609500645

Morton, J.N.; Sedam, J.B. 1938. Cutting operations to improve wildlife environment on forest areas. Journal of Wildlife Management. 2(4): 206-214. doi:10.2307/3795667

Moser, S.C.; Ekstrom, J.A. 2010. A framework to diagnose barriers to climate change adaptation. Proceedings of the National Academy of Sciences. 107(51): 22026-22031. doi:10.1073/pnas.1007887107

Mosgrove, J.L. 1980. The Malheur National Forest: an ethnographic history. John Day, OR: USDA Forest Service, Pacific Northwest Region, Malheur National Forest. 253 p. Malheur_National_Forest

Mote, P.W. 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters. 30(12): 1601. doi:10.1029/2003gl017258

Mote, P.W.; Parson, E.A.; Hamlet, A.F.; Keeton, W.S.; Lettenmaier, D.; Mantua, N.; Miles, E.L.; Peterson, D.W.; Peterson, D.L.; Slaughter, R.; Snover, A.K. 2003. Preparing for climatic change: the water, salmon, and forests of the Pacific Northwest. Climatic Change. 61(1-2): 45-88. doi:10.1023/A:1026302914358

Mott, D.G.; Nairn, L.D.; Cook, J.A. 1957. Radial growth in forest trees and effects of insect defoliation. Forest Science. 3(3): 286-304. doi:10.1093/forestscience/3.3.286

Moussaoui, L.; Leduc, A.; Fenton, N.J.; Lafleur, B.; Bergeron, Y. 2019. Changes in forest structure along a chronosequence in the black spruce boreal forest: Identifying structures to be reproduced through silvicultural practices. Ecological Indicators. 97: 89-99. doi:10.1016/j.ecolind.2018.09.059

Mowat, E.L. 1949. Preliminary guides for the management of lodgepole pine in Oregon and Washington. Res. Note No. 54. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 10 p. https://www.fs.usda.gov/treesearch/pubs/25571

Mroz, G.D.; Jurgensen, M.F.; Harvey, A.E.; Larsen, M.J. 1980. Effects of fire on nitrogen in forest floor horizons. Soil Science Society of America Journal. 44(2): 395-400. doi:10.2136/sssaj1980.03615995004400020038x

Mueggler, W.F. 1965. Ecology of seral shrub communities in the cedar-hemlock zone of northern Idaho. Ecological Monographs. 35(2): 165-185. doi:10.2307/1948416

Mueller-Dombois, D. 1987. Natural dieback in forests. BioScience. 37(8): 575-583. doi:10.2307/1310668

Muir, P.S. 1993. Disturbance effects on structure and tree species composition of *Pinus contorta* forests in western Montana. Canadian Journal of Forest Research. 23(8): 1617-1625. doi:10.1139/x93-201

Muir, J.A.; Robinson, D.C.E.; Geils, B.W. 2004. Characterizing the effects of dwarf mistletoe and other diseases for sustainable forest management. BC Journal of Ecosystems and Management. 3(2): article 6. Muir et al 2004

Munger, T.T. 1930. Ecological aspects of the transition from old forests to new. Science. 72(1866): 327-332. doi:10.1126/science.72.1866.327

Murphy, C.F.; Croft, B.A. 1990. Forest ant composition and foraging following aerial spraying of carbaryl to suppress western spruce budworm. Canadian Entomologist. 122(4): 595-606. doi:10.4039/Ent122595-7

Mutch, R.W.; Arno, S.F.; Brown, J.K.; Carlson, C.E.; Ottmar, R.D.; Peterson, J.L. 1993. Forest health in the Blue Mountains: a management strategy for fire-adapted ecosystems. Gen. Tech. Rep. PNW-GTR-310. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 14 p. http://www.treesearch.fs.fed.us/pubs/9056

Muzika, R.M.; Liebhold, A.M. 2000. A critique of silvicultural approaches to managing defoliating insects in North America. Agricultural and Forest Entomology. 2(2): 97-105. doi:10.1046/j.1461-9563.2000.00063.x

Mylek, M.R.; Schirmer, J. 2020. Understanding acceptability of fuel management to reduce wildfire risk: Informing communication through understanding complexity of thinking. Forest Policy and Economics. 113: 102120 (10 p). doi:10.1016/j.forpol.2020.102120

Nabuurs, G.J.; Masera, O.; Andrasko, K.; Benitez-Ponce, P.; Boer, R.; Dutschke, M.; Elsiddig, E.; Ford-Robertson, J.; Frumhoff, P.; Karjalainen, T.; Krankina, O.; Kurz, W.A.; Matsumoto, M.; Oyhantcabal, W.; Ravindranath, N.H.; Sanz Sanchez, M.J.; Zhang, X. 2007. Forestry. In: Climate change 2007: mitigation. Contribution of Working Group III to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK: Cambridge University Press: 541-584. https://www.ipcc.ch/site/assets/uploads/2018/02/ar4-wg3-chapter9-1.pdf

Nadrowski, K.; Wirth, C.; Scherer-Lorenzen, M. 2010. Is forest diversity driving ecosystem function and service? Current Opinion in Environmental Sustainability. 2(1-2): 75-79. doi:10.1016/j.cosust.2010.02.003

Naeem, S.; Bunker, D.E.; Hector, A.; Loreau, M.; Perrings, C., eds. 2009. Biodiversity, ecosystem functioning, and human wellbeing: an ecological and economic perspective. New York: Oxford University Press. 368 p. isbn:978-0-19-954796-8

Naiman, R.J.; Décamps, H. 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics. 28(1): 621-658. doi:10.1146/annurev.ecolsys.28.1.621

Namkoong, G. 2005. The misunderstood forest. Vancouver, BC: University of British Columbia, Forest Sciences Department. 132 p.

Natcher, D.C.; Hickey, C.G. 2002. Putting the community back into community-based resource management: a criteria and indicators approach to sustainability. Human Organization. 61(4): 350-363. http://www.metapress.com/content/DEM6FX3NPEP78XAQ

National Research Council. 2002. Ecological dynamics on Yellowstone's northern range. Washington, DC: National Academies Press. 180 p. isbn:0-309-083451

National Research Council. 2002. Riparian areas: Functions and strategies for management. Washington, DC: National Academies Press. 448 p. isbn:978-0-309-16977-6

Navarro-Cerrillo, R.M.; Sánchez-Salguero, R.; Rodriguez, C.; Duque Lazo, J.; Moreno-Rojas, J.M.; Palacios-Rodriguez, G.; Camarero, J.J. 2019. Is thinning an alternative when trees could die in response to drought? The case of planted *Pinus nigra* and *P. Sylvestris* stands in southern Spain. Forest Ecology and Management. 433: 313-324. doi:10.1016/j.foreco.2018.11.006

Nave, L.E.; Vance, E.D.; Swanston, C.W.; Curtis, P.S. 2010. Harvest impacts on soil carbon storage in temperate forests. Forest Ecology and Management. 259(5): 857-866. doi:10.1016/j.foreco.2009.12.009

Nealis, V.G.; Noseworthy, M.K.; Turnquist, R.; Waring, V.R. 2009. Balancing risks of disturbance from mountain pine beetle and western spruce budworm. Canadian Journal of Forest Research. 39(4): 839-848. doi:10.1139/X09-014

Neary, D.G.; Klopatek, C.C.; DeBano, L.F.; Ffolliott, P.F. 1999. Fire effects on belowground sustainability: a review and synthesis. Forest Ecology and Management. 122: 51-71. doi:10.1016/S0378-1127(99)00032-8

Nebeker, T.E.; Schmitz, R.F.; Tisdale, R.A.; Hobson, K.R. 1995. Chemical and nutritional status of dwarf mistletoe, Armillaria root rot, and Comandra blister rust infected trees which may influence tree susceptibility to bark beetle attack. Canadian Journal of Botany. 73(3): 360-369. doi:10.1139/b95-037

Neff, B.D.; Olden, J.D. 2006. Is peer review a game of chance? BioScience. 56(4): 333-340. doi:10.1641/0006-3568(2006)56[333:IPRAGO]2.0.CO;2

Negron, J.F.; Klutsch, J.G. 2017. Probability of infestation and extent of mortality models for mountain pine beetle in lodgepole pine forests in Colorado. Res. Note RMRS-RN-77. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 13 p. https://www.fs.usda.gov/treesearch/pubs/55502

Negron, J.F.; Schaupp, W.C., Jr.; Gibson, K.E.; Anhold, J.; Hansen, D.; Their, R.W.; Mocettinti, P. 1999. Estimating extent of mortality associated with the Douglas-fir beetle in the central and northern Rockies. Western Journal of Applied Forestry. 14(3): 121-127. doi:10.1093/wjaf/14.3.121

Nelson, E.E.; Martin, N.E.; Williams, R.E. 1981. Laminated root rot of western conifers. Forest Insect & Disease Leaflet 159. [Place of publication unknown]: USDA Forest Service. 6 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_042852.pdf

Neuenschwander, L.F.; Byler, J.W.; Harvey, A.E.; McDonald, G.I.; Ortiz, D.S.; Osborne, H.L.; Snyder, G.C.; Zack, A. 1999. White pine in the American west: a vanishing species – can we save it? Gen. Tech. Rep. RMRS-GTR-35. [Place of publication unknown]: USDA Forest Service, Rocky Mountain Research Station; University of Idaho. 20 p. https://www.fs.usda.gov/treesearch/pubs/4549

Neumann, M.; Starlinger, F. 2001. The significance of different indices for stand structure and diversity in forests. Forest Ecology and Management. 145(1-2): 91-106. doi:10.1016/s0378-1127(00)00577-6

Newton, A.C. 2007. Forest ecology and conservation: a handbook of techniques. Oxford, UK: Oxford University Press. 454 p. isbn:978-0-19-856744-8

Nichols, T.J. 1988. The relationship between western spruce budworm defoliation levels and growth of individual Douglas-fir and grand fir trees. Forest Science. 34(2): 496-504. doi:10.1093/forestscience/34.2.496

Nicoll, B.C.; Gardiner, B.A.; Rayner, B.; Peace, A.J. 2006. Anchorage of coniferous trees in relation to species, soil type, and rooting depth. Canadian Journal of Forest Research. 36(7): 1871-1883. doi:10.1139/x06-072

Niering, W.A. 1981. The role of fire management in altering ecosystems. In: Mooney, H.A.; Bonnicksen, T.M.; Christensen, N.L.; Lotan, J.E.; Reiners, W.A., tech. coords. Fire regimes and ecosystem properties; proceedings of the conference. Gen. Tech. Rep. WO-26. [Washington, DC]: USDA Forest Service: 489-510. https://archive.org/download/CAT83781017/CAT83781017.pdf

Niering, W.A. 1987. Vegetation dynamics (succession and climax) in relation to plant community management. Conservation Biology. 1(4): 287-295. doi:10.1111/j.1523-1739.1987.tb00049.x

Nigh, G.D.; Ying, C.C.; Qian, H. 2004. Climate and productivity of major conifer species in the interior of British Columbia, Canada. Forest Science. 50(5): 659-671. doi:10.1093/forestscience/50.5.659

Nigro, K.; Molinari, N. 2019. Status and trends of fire activity in southern California yellow pine and mixed conifer forests. Forest Ecology and Management. 441: 20-31. doi:10.1016/j.foreco.2019.01.020

Nilsson, S.G.; Niklasson, M.; Hedin, J.; Aronsson, G.; Gutowski, J.M.; Linder, P.; Ljungberg, H.; Mikusiński, G.; Ranius, T. 2002. Densities of large living and dead trees in old-growth temperate and boreal forests. Forest Ecology and Management. 161(1-3): 189-204. doi:10.1016/S0378-1127(01)00480-7

Nitschke, C.R. 2005. Does forest harvesting emulate fire disturbance? A comparison of effects on selected attributes in coniferous-dominated headwater systems. Forest Ecology and Management. 214(1-3): 305-319. doi:10.1016/j.foreco.2005.04.015

Nitschke, C.R.; Innes, J.L. 2008a. Climatic change and fire potential in south-central British Columbia, Canada. Global Change Biology. 14(4): 841-855. doi:10.1111/j.1365-2486.2007.01517.x

Nitschke, C.R.; Innes, J.L. 2008b. Integrating climate change into forest management in south-central British Columbia: an assessment of landscape vulnerability and development of a climate-smart framework. Forest Ecology and Management. 256(3): 313-327. doi:10.1016/j.foreco.2008.04.026

Nitschke, C.R.; Amoroso, M.; Coates, K.D.; Astrup, R. 2012. The influence of climate change, site type, and disturbance on stand dynamics in northwest British Columbia, Canada. Ecosphere. 3(1): art11. doi:10.1890/es11-00282.1

Niva, M.; Svensson, B.M.; Karlsson, P.S. 2006. Effects of light and water availability on shoot dynamics of the stoloniferous plant Linnaea borealis. Ecoscience. 13(3): 318-323. doi:10.2980/i1195-6860-13-3-318.1

Niwa, C.G.; Peck, R.W.; Torgersen, T.R. 2001. Soil, litter, and coarse woody debris habitats for arthropods in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 141-148. http://hdl.handle.net/2376/990

Noble, I.R.; Dirzo, R. 1997. Forests as human-dominated ecosystems. Science. 277(5325): 522-525. doi:10.1126/science.277.5325.522

Nolan, R.H.; Drew, D.M.; O'Grady, A.P.; Pinkard, E.A.; Paul, K.; Roxburgh, S.H.; Mitchell, P.J.; Bruce, J.; Battaglia, M.; Ramp, D. 2018. Safeguarding reforestation efforts against changes in climate and disturbance regimes. Forest Ecology and Management. 424: 458-467. doi:10.1016/j.foreco.2018.05.025

North, M., ed. 2012. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: USDA Forest Service, Pacific Southwest Research Station. 184 p. http://www.treesearch.fs.fed.us/pubs/40254

North, M.P.; Hurteau, M.D. 2011. High-severity wildfire effects on carbon stocks and emissions in fuels treated and untreated forest. Forest Ecology and Management. 261(6): 1115-1120. doi:10.1016/j.foreco.2010.12.039

North, M.; Innes, J.; Zald, H. 2007. Comparison of thinning and prescribed fire restoration treatments to Sierran mixed-conifer historic conditions. Canadian Journal of Forest Research. 37(2): 331-342. doi:10.1139/X06-236

North, M.; Stine, P.; O'Hara, K.; Zielinski, W.; Stephens, S. 2009a. An ecosystem management strategy for Sierran mixed-conifer forests. Gen. Tech. Rep. PSW-GTR-220. Albany, CA: USDA Forest Service, Pacific Southwest Research Station. 49 p. http://www.treesearch.fs.fed.us/pubs/32916

North, M.P.; Van de Water, K.M.; Stephens, S.L.; Collins, B.M. 2009b. Climate, rain shadow, and human-use influences on fire regimes in the eastern Sierra Nevada, California, USA. Fire Ecology. 5(3): 20-34. doi:10.4996/fireecology.0503020

North, M.; Hurteau, M.; Innes, J. 2009c. Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions. Ecological Applications. 19(6): 1385-1396. doi:10.1890/08-1173.1

North, M.; Stine, P.; Zielinski, W.; O'Hara, K.; Stephens, S. 2010. Harnessing fire for wildlife: fuels management in California's mixed-conifer forests. Wildlife Professional. 4(1): 30-33.

http://www.treesearch.fs.fed.us/pubs/34816

North, M.; Collins, B.M.; Stephens, S. 2012. Using fire to increase the scale, benefits, and future maintenance of fuels treatments. Journal of Forestry. 110(7): 392-401. doi:10.5849/jof.12-021

Norton, D.A. 1989. Tree windthrow and forest soil turnover. Canadian Journal of Forest Research. 19(3): 386-389. doi:10.1139/x89-059

Noss, R.F. 1983. A regional landscape approach to maintain diversity. BioScience. 33(11): 700-706. doi:10.2307/1309350

Noss, R.F.; Cooperrider, A.Y. 1994. Saving nature's legacy: protecting and restoring biodiversity. Washington, DC: Island Press. 416 p. isbn:1-55963-248-8

Odion, D.C.; Sarr, D.A. 2007. Managing disturbance regimes to maintain biological diversity in forested ecosystems of the Pacific Northwest. Forest Ecology and Management. 246(1): 57-65. doi:10.1016/j.foreco.2007.03.050

Odion, D.C.; Frost, E.J.; Strittholt, J.R.; Jiang, H.; Dellasala, D.A.; Moritz, M.A. 2004. Patterns of fire severity and forest conditions in the western Klamath Mountains, California. Conservation Biology. 18(4): 927-936. doi:10.1111/j.1523-1739.2004.00493.x

O'Farrell, P.J.; Anderson, P.M.L. 2010. Sustainable multifunctional landscapes: a review to implementation. Current Opinion in Environmental Sustainability. 2(1-2): 59-65. doi:10.1016/j.cosust.2010.02.005

OFRI (Oregon Forest Resources Institute). 2010. Federal forestland in Oregon. Portland, OR: Oregon Forest Resources Institute. 21 p. https://oregonforests.org/sites/default/files/2017-08/Federal_Forestlands.pdf

Ogden, A.E.; Innes, J.L. 2007. Incorporating climate change adaptation considerations into forest management and planning in the boreal forest. International Forestry Review. 9(3): 713-733. doi:10.1505/ifor.9.3.713

O'Hara, K.L. 1995. Early height development and species stratification across five climax series in the eastern Washington Cascade Range. New Forests. 9(1): 53-60. doi:10.1007/BF00028925

O'Hara, K.L. 1996. Dynamics and stocking-level relationships of multi-aged ponderosa pine stands. Forest Science. 42(Supplement 2): 1-34. doi:10.1093/forestscience/42.s2.a0001

O'Hara, K.L. 1998. Silviculture for structural diversity: a new look at multiaged systems. Journal of Forestry. 96(7): 4-10. doi:10.1093/jof/96.7.4a

O'Hara, K.L. 2014. Multiaged silviculture: Managing for complex forest stand structures. First ed. New York: Oxford University Press. 213 p. isbn:9780198703068 doi:10.1093/acprof:oso/9780198703068.001.0001

O'Hara, K.L.; Kollenberg, C.L. 2003. Stocking control procedures for multiaged lodgepole pine stands in the northern Rocky Mountains. Western Journal of Applied Forestry. 18(1): 15-21. doi:10.1093/wjaf/18.1.15

O'Hara, K.L.; Nagel, L.M. 2013. The stand: Revisiting a central concept in forestry. Journal of Forestry. 111(5): 335-340. doi:10.5849/jof.12-114

O'Hara, K.L.; Oliver, C.D. 1999. A decision system for assessing stand differentiation potential and prioritizing precommercial thinning treatments. Western Journal of Applied Forestry. 14(1): 7-13. doi:10.1093/wjaf/14.1.7

O'Hara, K.L.; Ramage, B.S. 2013. Silviculture in an uncertain world: utilizing multi-aged management systems to integrate disturbance. Forestry. 86(4): 401-410. doi:10.1093/forestry/cpt012

O'Hara, K.L.; Oliver, C.D.; Cobb, D.F. 1995. Development of western larch in mixed stands in Washington's Cascade Range: implications for prioritizing thinning treatments. In: Schmidt, W.C.; McDonald, K.J., comps. Ecology and management of Larix forests: a look ahead; proceedings of an international symposium. Gen. Tech. Rep. GTR-INT-319. Ogden, UT: USDA Forest Service, Intermountain Research Station: 259-264. https://archive.org/download/CAT10699418/CAT10699418.pdf

O'Hara, K.L.; Latham, P.A.; Hessburg, P.; Smith, B.G. 1996. A structural classification for inland Northwest forest vegetation. Western Journal of Applied Forestry. 11(3): 97-102. doi:10.1093/wjaf/11.3.97

O'Hara, K.L.; Youngblood, A.; Waring, K.M. 2010. Maturity selection versus improvement selection: lessons from a mid-20th century controversy in the silviculture of ponderosa pine. Journal of Forestry. 108(8): 397-407. doi:10.1093/jof/108.8.397

Ohlson, M.; Tryterud, E. 1999. Long-term spruce forest continuity – a challenge for a sustainable Scandinavian forestry. Forest Ecology and Management. 124(1): 27-34. doi:10.1016/S0378-1127(99)00053-5

Öhman, K. 2000. Creating continuous areas of old forest in long-term forest planning. Canadian Journal of Forest Research. 30(11): 1817-1823. doi:10.1139/x00-103

Økland, B.; Liebhold, A.; Bjørnstad, O.; Erbilgin, N.; Krokene, P. 2005. Are bark beetle outbreaks less synchronous than forest Lepidoptera outbreaks? Oecologia. 146(3): 365-372. doi:10.1007/s00442-005-0221-2

O'Laughlin, J. 1994. Assessing forest health conditions in Idaho with forest inventory data. Journal of Sustainable Forestry. 2(3-4): 221-247. doi:10.1300/J091v02n03_01

O'Laughlin, J.; Cook, P.S. 2003. Inventory-based forest health indicators: implications for national forest management. Journal of Forestry. 101(2): 11-17. doi:10.1093/jof/101.2.11

O'Laughlin, J.; Livingston, R.L.; Thier, R.; Thornton, J.; Toweill, D.E.; Morelan, L. 1994. Defining and measuring forest health. Journal of Sustainable Forestry. 2(1/2): 65-85. doi:10.1300/J091v02n01_03

Oliver, C.D. 1980. Even-aged development of mixed-species stands. Journal of Forestry. 78(4): 201-203. doi:10.1093/jof/78.4.201

Oliver, C.D. 1981. Forest development in North America following major disturbances. Forest Ecology and Management. 3: 153-168. doi:10.1016/0378-1127(80)90013-4

Oliver, C.D. 1992. A landscape approach: Achieving and maintaining biodiversity and economic productivity. Journal of Forestry. 90(9): 20-25. doi:10.1093/jof/90.9.20

Oliver, W.W. 1995. Is self-thinning in ponderosa pine ruled by *Dendroctonus* bark beetles? In: Eskew, L.G., comp. Forest health through silviculture; proceedings of the 1995 National Silviculture Workshop. Gen. Tech. Rep. RM-GTR-267. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station: 213-218. http://www.treesearch.fs.fed.us/pubs/23504

Oliver, C.D.; Deal, R.L. 2007. A working definition of sustainable forestry and means of achieving it at different spatial scales. Journal of Sustainable Forestry. 24(2-3): 141-163. doi:10.1300/J091v24n02_03

Oliver, W.W.; Dolph, K.L. 1992. Mixed-conifer seedling growth varies in response to overstory release. Forest Ecology and Management. 48(1-2): 179-183. doi:10.1016/0378-1127(92)90130-2

Oliver, C.D.; Kenady, R.M., eds. 1982. Proceedings of the biology and management of true fir in the Pacific Northwest symposium. Contribution No. 45. Seattle, WA: University of Washington, College of Forest Resources, Institute of Forest Resources. 344 p.

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics. Update edition. New York: John Wiley and Sons. 520 p. isbn:0-471-13833-9

Oliver, W.W.; Ryker, R.A. 1990. *Pinus ponderosa* Dougl. ex Laws.; ponderosa pine. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America; volume 1, conifers. Agric. Handb. 654. Washington, DC: USDA Forest Service: 413-424.

https://www.treesearch.fs.fed.us/pubs/1547

Oliver, C.D.; Irwin, L.L.; Knapp, W.H. 1994a. Eastside forest management practices: historical overview, extent of their applications, and their effects on sustainability of ecosystems. Gen. Tech. Rep. PNW-GTR-324. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 73 p. http://www.treesearch.fs.fed.us/pubs/6294

Oliver, C.D.; Harrington, C.; Bickford, M.; Gara, R.; Knapp, W.; Lightner, G.; Hicks, L. 1994b. Maintaining and creating old growth structural features in previously disturbed stands typical of the eastern Washington Cascades. Journal of Sustainable Forestry. 2(3-4): 353-387. doi:10.1300/J091v02n03_09

Oliver, C.D.; Ferguson, D.E.; Harvey, A.E.; Malany, H.S.; Mandzak, J.M.; Mutch, R.W. 1994. Managing ecosystems for forest health: an approach and the effects on uses and values. Journal of Sustainable Forestry. 2(1-2): 113-133. doi:10.1300/J091v02n01_05

O'Loughlin, J.; Pfister, R.D., eds. 1983. Management of second-growth forests: the state of knowledge and research needs. Missoula, MT: Montana Forest and Conservation Experiment Station, School of Forestry, University of Montana.

Olsen, C.S.; Shindler, B.A. 2010. Trust, acceptance, and citizen–agency interactions after large fires: influences on planning processes. International Journal of Wildland Fire. 19(1): 137-147. doi:10.1071/WF08168

Olson, J.S. 1963. Energy storage and the balance of producers and decomposers in ecological systems. Ecology. 44(2): 322-331. doi:10.2307/1932179

Olson, J.S. 1981. Carbon balance in relation to fire regimes. In: Mooney, H.A.; Bonnicksen, T.M.; Christensen, N.L.; Lotan, J.E.; Reiners, W.A., tech. coords. Fire regimes and ecosystem properties; proceedings of the conference. Gen. Tech. Rep. WO-26. [Washington, DC]: USDA Forest Service: 327-378. https://archive.org/download/CAT83781017/CAT83781017.pdf

Olson, D.L. 2000. Fire in riparian zones: a comparison of historical fire occurrence in riparian and upslope forests in the Blue Mountains and southern Cascades of Oregon. M.S. thesis. Seattle, WA: University of Washington. 274 p. https://www.fs.fed.us/rm/pubs/rmrs_gtr292/2000_olsen.pdf

Olson, D.L.; Agee, J.K. 2005. Historical fires in Douglas-fir dominated riparian forests of the southern Cascades, Oregon. Fire Ecology. 1(1): 50-74. doi:10.4996/fireecology.0101050

Olsson, P.; Folke, C.; Berkes, F. 2004. Adaptive comanagement for building resilience in social-ecological systems. Environmental Management. 34(1): 75-90. doi:10.1007/s00267-003-0101-7

Omernik, J.M. 1995. Ecoregions: a spatial framework for environmental management. In: Davis, W.S.; Simon, T.P., eds. Biological assessment and criteria: tools for water resource planning and decision making. Ann Arbor (MI): Lewis Publishers: 49-65.

Omi, P.N. 1996. The role of fuelbreaks. In: Proceedings of the seventeenth annual forest vegetation management conference; 1996 January 16-18; Redding, CA: 89-96. http://www.fvmc.org/PDF/FVMCProc17th(1996).pdf

O'Neill, R.V.; DeAngelis, D.L.; Waide, J.B.; Allen, T.F.H. 1986. A hierarchical concept of ecosystems. Monographs in Population Biology No. 23. Princeton, NJ: Princeton University Press. 253 p. isbn:0-691-08437-8

Ontl, T.A.; Janowiak, M.K.; Swanston, C.W.; Daley, J.; Handler, S.; Cornett, M.; Hagenbuch, S.; Handrick, C.; Mccarthy, L.; Patch, N. 2020. Forest management for carbon sequestration and climate adaptation. Journal of Forestry. 118(1): 86-101. doi:10.1093/jofore/fvz062

Oswald, B.P.; Wellner, K.; Boyce, R.; Neuenschwander, L.F. 1999. Germination and initial growth of four coniferous species on varied duff depths in northern Idaho. Journal of Sustainable Forestry. 8(1): 11-21. doi:10.1300/J091v08n01_02

Oswalt, S.N.; Smith, W.B.; Miles, P.D.; Pugh, S.A. 2019. Forest resources of the United States, 2017: A technical document supporting the Forest Service 2020 RPA assessment. Gen. Tech. Rep. WO-97. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 223 p. https://www.fs.usda.gov/treesearch/pubs/57903

Ottmar, R.D.; Sandberg, D.V. 2001. Wildland fire in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 46-54. http://hdl.handle.net/2376/980

Overpeck, J.T.; Rind, D.; Goldberg, R. 1990. Climate-induced changes in forest disturbance and vegetation. Nature. 343(6253): 51-53. doi:10.1038/343051a0

Owen, S.M.; Sieg, C.H.; Gehring, C.A.; Bowker, M.A. 2009. Above- and belowground responses to tree thinning depend on the treatment of tree debris. Forest Ecology and Management. 259(1): 71-80. doi:10.1016/j.foreco.2009.09.044

Pacala, S.W.; Deutschman, D.H. 1995. Details that matter: the spatial distribution of individual trees maintains forest ecosystem function. Oikos. 74(3): 357-365. doi:10.2307/3545980

Page, W.G.; Jenkins, M.J. 2007. Predicted fire behavior in selected mountain pine beetle-infested lodgepole pine. Forest Science. 53(6): 662-674. doi:10.1093/forestscience/53.6.662

Page, H.N.; Bork, E.W.; Newman, R.F. 2005. Understorey responses to mechanical restoration and drought within montane forests of British Columbia. BC Journal of Ecosystems and Management. 6(1): 8-21. https://jem-online.org/forrex/index.php/jem/article/view/304

Page, W.G.; Jenkins, M.J.; Runyon, J.B. 2012. Mountain pine beetle attack alters the chemistry and flammability of lodgepole pine foliage. Canadian Journal of Forest Research. 42(8): 1631-1647. doi:10.1139/x2012-094

Page-Dumroese, D.S.; Jurgensen, M.F. 2006. Soil carbon and nitrogen pools in mid- to late-successional forest stands of the northwestern United States: potential impact of fire. Canadian Journal of Forest Research. 36(9): 2270-2284. doi:10.1139/X06-125

Page-Dumroese, D.; Harvey, A.; Jurgensen, M.; Graham, R. 1991. Organic matter function in the western-montane forest soil system. In: Harvey, A.E.; Neuenschwander, L.F., comps. Proceedings–management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: USDA Forest Service, Intermountain Research Station: 95-100. https://www.fs.usda.gov/treesearch/pubs/33908

Page-Dumroese, D.; Jurgensen, M.; Elliot, W.; Rice, T.; Nesser, J.; Collins, T.; Meurisse, R. 2000. Soil quality standards and guidelines for forest sustainability in northwestern North America. Forest Ecology and Management. 138(1-3): 445-462. doi:10.1016/S0378-1127(00)00430-8

Page-Dumroese, D.S.; Miller, R.E.; Mital, J.; McDaniel, P.; Miller, D., tech. eds. 2007. Volcanic-ash-derived forest soils of the inland Northwest: properties and implications for management and restoration. Proceed. RMRS-P-44. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 220 p. https://www.fs.usda.gov/treesearch/pubs/26202

Page-Dumroese, D.S.; Abbott, A.M.; Rice, T.M. 2009. Forest soil disturbance monitoring protocol: Volume I: Rapid assessment. Gen. Tech. Rep. WO-GTR-82a. Washington, DC: USDA Forest Service. 31 p. http://www.treesearch.fs.fed.us/pubs/34427

Page-Dumroese, D.S.; Jurgensen, M.; Terry, T. 2010a. Maintaining soil productivity during forest or biomass-to-energy thinning harvests in the western United States. Western Journal of Applied Forestry. 25(1): 5-11. doi:10.1093/wjaf/25.1.5

Page-Dumroese, D.; Jurgensen, M.; Neary, D.; Curran, M.; Trettin, C. 2010b. Soil quality is fundamental to ensuring healthy forests. In: Pye, J.M.; Rauscher, H.M.; Sands, Y.; Lee, D.C.; Beatty, J.S., tech. eds. Advances in threat assessment and their application to forest and rangeland management–Volume 1. Gen. Tech. Rep. PNW-GTR-802. Portland, OR: USDA Forest Service, Pacific Northwest and Southern Research Stations: 27-36.

http://www.treesearch.fs.fed.us/pubs/36650

Paine, T.D.; Raffa, K.F.; Harrington, T.C. 1997. Interactions among Scolytid bark beetles, their associated fungi, and live host conifers. Annual Review of Entomology. 42(1): 179-206. doi:10.1146/annurev.ento.42.1.179

Paine, R.T.; Tegner, M.J.; Johnson, E.A. 1998. Compounded perturbations yield ecological surprises. Ecosystems. 1(6): 535-545. doi:10.1007/s100219900049

Painter, L.E.; Beschta, R.L.; Larsen, E.J.; Ripple, W.J. 2015. Recovering aspen follow changing elk dynamics in Yellowstone: evidence of a trophic cascade? Ecology. 96(1): 252-263. doi:10.1890/14-0712.1

Palik, B.J.; Pregitzer, K.S. 1991. The relative influence of establishment time and height-growth rates on species vertical stratification during secondary forest succession. Canadian Journal of Forest Research. 21(10): 1481-1490. doi:10.1139/x91-208

Palik, B.J.; Pregitzer, K.S. 1995. Variability in early height growth rate of forest trees: Implications for retrospective studies of stand dynamics. Canadian Journal of Forest Research. 25(5): 767-776. doi:10.1139/x95-084

Palik, B.J.; Goebel, P.C.; Kirkman, L.K.; West, L. 2000. Using landscape hierarchies to guide restoration of disturbed ecosystems. Ecological Applications. 10(1): 189-202. doi:10.1890/1051-0761(2000)010[0189:ULHTGR]2.0.CO;2

Palmer, M.A.; Bernhardt, E.S.; Allan, J.D.; Lake, P.S.; Alexander, G.; Brooks, S.; Carr, J.; Clayton, S.; Dahm, C.N.; Follstad Shah, J.; Galat, D.L.; Loss, S.G.; Goodwin, P.; Hart, D.D.; Hassett, B.; Jenkinson, R.; Kondolf, G.M.; Lave, R.; Meyer, J.L.; O'Donnell, T.K.; Pagano, L.; Sudduth, E. 2005. Standards for ecologically successful river restoration. Journal of Applied Ecology. 42(2): 208-217. doi:10.1111/j.1365-2664.2005.01004.x

Paquette, A.; Messier, C. 2009. The role of plantations in managing the world's forests in the Anthropocene. Frontiers in Ecology and the Environment. 8(1): 27-34. doi:10.1890/080116

Parisien, M.-A.; Parks, S.; Miller, C.; Krawchuk, M.; Heathcott, M.; Moritz, M. 2011. Contributions of ignitions, fuels, and weather to the spatial patterns of burn probability of a boreal landscape. Ecosystems. Online First: 1-15. doi:10.1007/s10021-011-9474-2

Parisien, M.-A.; Snetsinger, S.; Greenberg, J.A.; Nelson, C.R.; Schoennagel, T.; Dobrowski, S.Z.; Moritz, M.A. 2012. Spatial variability in wildfire probability across the western United States. International Journal of Wildland Fire. 21(4): 313-327. doi:10.1071/WF11044

Parker, J. 1955. Annual trends in cold hardiness of ponderosa pine and grand fir. Ecology. 36(3): 377-380. doi:10.2307/1929571

Parker, J. 1956. Drought resistance in woody plants. Botanical Review. 22(4): 241-289. doi:10.1007/BF02872473

Parker, D.L. 1973. Trend of a mountain pine beetle outbreak. Journal of Forestry. 71(11): 698-700. doi:10.1093/jof/71.11.698

Parker, A.J. 1982. The topographic relative moisture index: an approach to soil-moisture assessment in mountain terrain. Physical Geography. 3(2): 160-168. doi:10.1080/02723646.1982.10642224

Parker, G.G. 1983. Throughfall and stemflow in the forest nutrient cycle. Advances in Ecological Research. 13: 57-133. doi:10.1016/S0065-2504(08)60108-7

Parker, T.J.; Clancy, K.M.; Mathiasen, R.L. 2006. Interactions among fire, insects and pathogens in coniferous forests of the interior western United States and Canada. Agricultural and Forest Entomology. 8(3): 167-189. doi:10.1111/j.1461-9563.2006.00305.x

Parks, C.G. 1993. The influence of induced host moisture stress on the growth and development of western spruce budworm and *Armillaria ostoyae* on grand fir seedlings. PhD dissertation. Corvallis, OR: Oregon State University, Department of Forest Resources. 154 p. Parks_Catherine_Gray_1993

Parks, C.G.; Flanagan, P.T. 2001. Dwarf mistletoes (*Arceuthobium* spp.), rust diseases, and stem decays in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 31-37. http://hdl.handle.net/2376/978

Parks, C.G.; Shaw, D.C. 1996. Death and decay: a vital part of living canopies. Northwest Science. 70(Special Issue): 46-53. http://hdl.handle.net/2376/1285

Parks, C.G.; Bednar, L.; Tiedemann, A.R. 1998. Browsing ungulates–an important consideration in dieback and mortality of Pacific yew (*Taxus brevifolia*) in a northeastern Oregon stand. Northwest Science. 72(3): 190-197. http://hdl.handle.net/2376/1215

Parks, C.G.; Bull, E.L.; Tinnin, R.O.; Shepherd, J.F.; Blumton, A.K. 1999. Wildlife use of dwarf mistletoe brooms in Douglas-fir in northeast Oregon. Western Journal of Applied Forestry. 14(2): 100-105. doi:10.1093/wjaf/14.2.100

Parks, S.A.; Parisien, M.-A.; Miller, C. 2012. Spatial bottom-up controls on fire likelihood vary across western North America. Ecosphere. 3(1): art12. doi:10.1890/es11-00298.1

Parks, S.A.; Miller, C.; Parisien, M.-A.; Holsinger, L.M.; Dobrowski, S.Z.; Abatzoglou, J. 2015. Wildland fire deficit and surplus in the western United States, 1984-2012. Ecosphere. 6(12): 275 (13 p). doi:10.1890/es15-00294.1

Parks, S.A.; Miller, C.; Holsinger, L.M.; Baggett, L.S.; Bird, B.J. 2016. Wildland fire limits subsequent fire occurrence. International Journal of Wildland Fire. 25(2): 182-190. doi:10.1071/WF15107

Parks, S.A.; Holsinger, L.M.; Miller, C.; Parisien, M.-A. 2018a. Analog-based fire regime and vegetation shifts in mountainous regions of the western US. Ecography. 41(6): 910-921. doi:10.1111/ecog.03378

Parks, S.A.; Holsinger, L.M.; Panunto, M.H.; Jolly, W.M.; Dobrowski, S.Z.; Dillon, G.K. 2018b. High-severity fire: evaluating its key drivers and mapping its probability across western US forests. Environmental Research Letters. 13(4): 044037 (13 p). doi:10.1088/1748-9326/aab791

Parmeter, J.R., Jr. 1978. Forest stand dynamics and ecological factors in relation to dwarf mistletoe spread, impact, and control. In: Scharpf, R.F.; Parmeter, J.R., Jr., tech. coords. Proceedings of the symposium on dwarf mistletoe control through forest management. Gen. Tech. Rep. PSW-31. Berkeley, CA: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station: 16-30. https://www.fs.usda.gov/treesearch/pubs/24083

Parminter, J. 1998. Natural disturbance ecology. Chapter 1. In: Voller, J.; Harrison, S., eds. Conservation biology principles for forested landscapes. Vancouver, BC: UBC Press: 3-41. isbn:0-7748-0630-3

Parminter, J. 2014. Natural disturbance bibliography for British Columbia. Tech. Rep. 080. Victoria, BC: Province of British Columbia, Ministry of Forests, Lands and Natural Resource Operations. 170 p. http://www.for.gov.bc.ca/hfd/pubs/Docs/Tr/TR080.htm

Parry, D.L.; Filip, G.M.; Willits, S.A.; Parks, C.G. 1996. Lumber recovery and deterioration of beetle-killed Douglas-fir and grand fir in the Blue Mountains of eastern Oregon. Gen. Tech. Rep. PNW-GTR-376. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 24 p. https://www.fs.usda.gov/treesearch/pubs/3058

Parry, M.L.; Canziani, O.F.; Palutikof, J.P.; and others. 2007. Technical summary. In: Parry, M.L.; Canziani, O.F.; Palutikof, J.P.; van der Linden, P.J.; Hanson, C.E., eds. Climate change 2007: impacts, adaptation and vulnerability. Contribution of Working Group II to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, UK: Cambridge University Press: 23-78. https://www.ipcc.ch/site/assets/uploads/2018/02/ar4-wg2-ts-1.pdf

Parsons, D.J. 1981. The role of fire management in maintaining natural ecosystems. In: Mooney, H.A.; Bonnicksen, T.M.; Christensen, N.L.; Lotan, J.E.; Reiners, W.A., tech. coords. Fire regimes and ecosystem properties; proceedings of the conference. Gen. Tech. Rep. WO-26. [Washington, DC]: USDA Forest Service: 469-488. https://archive.org/download/CAT83781017/CAT83781017.pdf

Parsons, D.J.; DeBenedetti, S.H. 1979. Impact of fire suppression on a mixed-conifer forest. Forest Ecology and Management. 2: 21-33. doi:10.1016/0378-1127(79)90034-3

Patry, C.; Kneeshaw, D.; Wyatt, S.; Grenon, F.; Messier, C. 2013. Forest ecosystem management in North America: From theory to practice. Forestry Chronicle. 89(4): 525-537. doi:10.5558/tfc2013-093

Paul, E.A., ed. 2007. Soil microbiology, ecology, and biochemistry. Burlington, MA: Academic Press. 532 p. isbn:978-0-12-546807-7

Paustian, S.J.; Hemstrom, M.; Dennis, J.G.; Dittberner, P.; Brookes, M.H. 1999. Ecosystem processes and functions: management considerations. In: Szaro, R.C.; Johnson, N.C.; Sexton, W.T.; Malk, A.J., eds. Ecological stewardship: a common reference for ecosystem management. Volume II. Biological and ecological dimensions; Humans as agents of ecological change. Oxford, UK: Elsevier Science, Ltd: 255-279. isbn:0-08-043206-9

Pearson, P.N.; Palmer, M.R. 2000. Atmospheric carbon dioxide concentrations over the past 60 million years. Nature. 406(6797): 695-699. doi:10.1038/35021000

Pedersen, B.S. 1998. Modeling tree mortality in response to short- and long-term environmental stresses. Ecological Modelling. 105(2-3): 347-351. doi:10.1016/S0304-3800(97)00162-2

Peet, R.K.; Christensen, N.L. 1987. Competition and tree death. BioScience. 37(8): 586-595. doi:10.2307/1310669

Peipoch, M.; Brauns, M.; Hauer, F.R.; Weitere, M.; Valett, H.M. 2015. Ecological simplification: Human influences on riverscape complexity. BioScience. 65(11): 1057-1065. doi:10.1093/biosci/biv120

Pelren, E.C.; Crawford, J.A. 1999. Blue grouse nesting parameters and habitat associations in northeastern Oregon. Great Basin Naturalist. 59(4): 368-373. https://www.jstor.org/stable/41713138

Peltonen, M.; Liebhold, A.M.; Bjornstad, O.N.; Williams, D.W. 2002. Spatial synchrony in forest insect outbreaks: roles of regional stochasticity and dispersal. Ecology. 83(11): 3120-3129. doi:10.1890/0012-9658(2002)083[3120:SSIFIO]2.0.CO;2

Pelz, K.A.; Smith, F.W. 2012. Thirty year change in lodgepole and lodgepole/mixed conifer forest structure following 1980s mountain pine beetle outbreak in western Colorado, USA. Forest Ecology and Management. 280: 93-102. doi:10.1016/j.foreco.2012.05.032

Pelz, K.A.; Smith, F.W. 2018. Effects of stand structure, browsing, and biophysical conditions on regeneration following mountain pine beetle in mixed lodgepole pine and aspen forests of the southern Rockies. Forests. 9(9): 525 (12 p). doi:10.3390/f9090525

Pelz, K.A.; Rhoades, C.C.; Hubbard, R.M.; Battaglia, M.A.; Smith, F.W. 2015. Species composition influences management outcomes following mountain pine beetle in lodgepole pine-dominated forests. Forest Ecology and Management. 336: 11-20. doi:10.1016/j.foreco.2014.09.034

Pelz, K.A.; Rhoades, C.C.; Hubbard, R.M.; Smith, F.W. 2018. Severity of overstory mortality influences conifer recruitment and growth in mountain pine beetle-affected forests. Forests. 9(9): 536 (11 p). doi:10.3390/f9090536

Perera, A.H.; Cui, W. 2010. Emulating natural disturbances as a forest management goal: lessons from fire regime simulations. Forest Ecology and Management. 259(7): 1328-1337. doi:10.1016/j.foreco.2009.03.018

Perera, A.H.; Baldwin, D.J.B.; Yemshanov, D.G.; Schnekenburger, F.; Weaver, K.; Boychuk, D. 2003. Predicting the potential for old-growth forests by spatial simulation of landscape ageing patterns. Forestry Chronicle. 79(3): 621-631. doi:10.5558/tfc79621-3

Perera, A.H.; Buse, L.J.; Weber, M.G., eds. 2004. Emulating natural forest landscape disturbances: concepts and applications. New York: Columbia University Press. 315 p. isbn:0-231-12916-5

Perera, A.H.; Buse, L.J.; Crow, T.R. 2006. Forest landscape ecology: transferring knowledge to practice. New York: Springer. 223 p. isbn:0-387-34242-7

Pernot, J.F. 1913a. Insect control policy for District 6. Unpublished typescript report obtained from the National Archives, College Park, MD; record group 95. 14 p. On file with: USDA Forest Service, Umatilla National Forest, Supervisor's Office, Pendleton, OR.

Pernot, J.F. 1913b. The insect situation in 1913 on the Whitman National Forest. Unpublished typescript report obtained from the National Archives, College Park, MD; record group 95. 13 p. On file with: USDA Forest Service, Umatilla National Forest, Supervisor's Office, Pendleton, OR.
https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5414671.pdf

Perrakis, D.D.B.; Agee, J.K. 2006. Seasonal fire effects on mixed-conifer forest structure and ponderosa pine resin properties. Canadian Journal of Forest Research. 36(1): 238-254. doi:10.1139/x05-212

Perry, D.A. 1988. Landscape pattern and forest pests. Northwest Environmental Journal. 4(2): 213-228.

Perry, D.A. 1998. The scientific basis of forestry. Annual Review of Ecology and Systematics. 29: 435-466. doi:10.1146/annurev.ecolsys.29.1.435

Perry, D.A.; Amaranthus, M.P. 1997. Disturbance, recovery, and stability. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press: 31-56. isbn:1-55963-399-9

Perry, G.L.W.; Millington, J.D.A. 2008. Spatial modelling of succession-disturbance dynamics in forest ecosystems: concepts and examples. Perspectives in Plant Ecology, Evolution and Systematics. 9(3-4): 191-210. doi:10.1016/j.ppees.2007.07.001

Perry, D.A.; Wheeler, C.T.; Helgerson, O.T. [1979.] Nitrogen-fixing plants for silviculture: some genecological considerations. In: Gordon, J.C.; Wheeler, C.T.; Perry, D.A., eds. Symbiotic nitrogen fixation in the management of temperate forests; proceedings of a workshop held April 2-5, 1979. Corvallis, OR: Oregon State University, Forest Research Laboratory: 243-252.

Perry, D.A.; Amaranthus, M.P.; Borchers, J.G.; Borchers, S.L.; Brainerd, R.E. 1989. Bootstrapping in ecosystems. BioScience. 39(4): 230-237. doi:10.2307/1311159

Perry, D.A.; Meurisse, R.; Thomas, B.; Miller, R.; Boyle, J.; Means, J.; Perry, C.R.; Powers, R.F. 1989b. Maintaining the long-term productivity of Pacific Northwest forest ecosystems. Portland, OR: Timber Press. 256 p. isbn:0-88192-144-0

Perry, D.A.; Jing, H.; Youngblood, A.; Oetter, D.R. 2004. Forest structure and fire susceptibility in volcanic landscapes of the eastern high Cascades, Oregon. Conservation Biology. 18(4): 913-926. doi:10.1111/j.1523-1739.2004.00530.x

Perry, D.A.; Oren, R.; Hart, S.C. 2008. Forest ecosystems. 2nd edition. Baltimore, MD: Johns Hopkins University Press. 606 p. isbn:978-0-8018-8840-3

Perry, D.A.; Hessburg, P.F.; Skinner, C.N.; Spies, T.A.; Stephens, S.L.; Taylor, A.H.; Franklin, J.F.; McComb, B.; Riegel, G. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and northern California. Forest Ecology and Management. 262(5): 703-717. doi:10.1016/j.foreco.2011.05.004

Peters, R.L. 1990. Effects of global warming on forests. Forest Ecology and Management.

35(1-2): 13-33. doi:10.1016/0378-1127(90)90229-5

Peters, R.S.; Waller, D.M.; Noon, B.; Pickett, S.T.A.; Murphy, D.; Cracraft, J.; Kiester, R.; Kuhlmann, W.; Houck, O.; Snape, W.J., III. 1997. Standard scientific procedures for implementing ecosystem management on public lands. In: Pickett, S.T.A.; Ostfeld, R.S.; Shachak, M.; Likens, G.E., eds. The ecological basis of conservation: heterogeneity, ecosystems, and biodiversity. New York: Chapman & Hall: 320-336. isbn:0-412-09851-2

Petersen, J.D. 1992. Grey ghosts in the Blue Mountains. Evergreen (Jan./Feb.): 3-8.

Peterson, C.J. 2000. Catastrophic wind damage to North American forests and the potential impact of climate change. Science of The Total Environment. 262(3): 287-311. doi:10.1016/S0048-9697(00)00529-5

Peterson, W.C.; Hibbs, D.E. 1989. Adjusting stand density management guides for sites with low stocking potential. Western Journal of Applied Forestry. 4(2): 62-65. doi:10.1093/wjaf/4.2.62

Peterson, D.L.; Johnson, M.C. 2007. Science-based strategic planning for hazardous fuel treatment. Fire Management Today. 67(3): 13-18. https://www.fs.fed.us/fire/fmt/fmt_pdfs/FMT67-3.pdf

Peterson, C.J.; Leach, A.D. 2008. Salvage logging after windthrow alters microsite diversity, abundance and environment, but not vegetation. Forestry. 81(3): 361-376. doi:10.1093/forestry/cpn007

Peterson, D.L.; Parker, V.T., eds. 1998. Ecological scale: theory and applications. New York: Columbia University Press. 615 p. isbn:0-231-10503-7

Peterson, D.L.; Innes, J.L.; O'Brian, K., eds. 2004. Climate change, carbon, and forestry in northwestern North America: proceedings of a workshop. Gen. Tech. Rep. PNW-GTR-614. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 117 p. https://www.fs.usda.gov/treesearch/pubs/6596

Peterson, D.L.; Halofsky, J.E.; Johnson, M.C. 2011a. Managing and adapting to changing fire regimes in a warmer climate. In: McKenzie, D.; Miller, C.; Falk, D.A., eds. The landscape ecology of fire. New York: Springer: 249-267. isbn:978-94-007-0300-1

Peterson, D.L.; Millar, C.I.; Joyce, L.A.; Furniss, M.J.; Halofsky, J.E.; Neilson, R.P.; Morelli, T.L. 2011b. Responding to climate change in national forests: a guidebook for developing adaptation options. Gen. Tech. Rep. PNW-GTR-855. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 109 p. http://www.treesearch.fs.fed.us/pubs/39884

Petraitis, P.S.; Latham, R.E.; Niesenbaum, R.A. 1989. The maintenance of species diversity by disturbance. Quarterly Review of Biology. 64(4): 393-418. https://www.jstor.org/stable/2830102

Pettit, N.; Naiman, R. 2007. Fire in the riparian zone: characteristics and ecological consequences. Ecosystems. 10(5): 673-687. doi:10.1007/s10021-007-9048-5

Pettit, J.M.; Burton, J.I.; DeRose, R.J.; Long, J.N.; Voelker, S.L. 2019. Epidemic spruce beetle outbreak changes drivers of Engelmann spruce regeneration. Ecosphere. 10(11): e02912 (18 p). doi:10.1002/ecs2.2912

Pfeifer, E. 2005. Bear Creek prescribed burn: Trial monitoring and evaluation report. Walla Walla, WA: USDA Forest Service, Umatilla National Forest, Walla Walla Ranger District. 20 p.

Pfister, R.D. 1993. The need and potential for ecosystem management in forests of the inland west. In: Aplet, G.H.; Johnson, N.; Olson, J.T.; Sample, V.A., eds. Defining sustainable forestry. Washington, DC: Island Press: 217-239. isbn:1-55963-234-8

Pfister, R.D.; Daubenmire, R. 1975. Ecology of lodgepole pine (*Pinus contorta* Douglas). In: Baumgartner, D.M., ed. Management of lodgepole pine ecosystems; symposium proceedings. Pullman, WA: Washington State University, Cooperative Extension Service: 27-46.

Pharis, R.P.; Ferrell, W.K. 1966. Differences in drought resistance between coastal and inland sources of Douglas-fir. Canadian Journal of Botany. 44(12): 1651-1659. doi:10.1139/b66-177

Phillips, J. 1995. The crisis in our forests. Sunset. 195(1): 87-92.

Phillips, M.A.; Croteau, R.B. 1999. Resin-based defenses in conifers. Trends in Plant Science. 4(5): 184-190. doi:10.1016/S1360-1385(99)01401-6

Phillips, I.D.; Cobb, T.P.; Spence, J.R.; Brigham, R.M. 2006. Salvage logging, edge effects, and carabid beetles: connections to conservation and sustainable forest management. Environmental Entomology. 35(4): 950-957. doi:10.1603/0046-225X-35.4.950

Phoebus, I.; Segelbacher, G.; Stenhouse, G.B. 2017. Do large carnivores use riparian zones? Ecological implications for forest management. Forest Ecology and Management. 402: 157-165. doi:10.1016/j.foreco.2017.07.037

Pickett, S.T.A. 1976. Succession: an evolutionary interpretation. American Naturalist. 110(971): 107-119. https://www.jstor.org/stable/2459880

Pickett, S.T.A. 1980. Non-equilibrium coexistence of plants. Bulletin of the Torrey Botanical Club. 107(2): 238-248. doi:10.2307/2484227

Pickett, S.T.A.; Cadenasso, M.L. 1995. Landscape ecology: spatial heterogeneity in ecological systems. Science. 269(5222): 331-334. doi:10.1126/science.269.5222.331

Pickett, S.T.A.; Ostfeld, R.S. 1995. The shifting paradigm in ecology. In: Knight, R.L.; Bates, S.F., eds. A new century for natural resources management. Washington, DC: Island Press: 261-278. isbn:1-55963-262-3

Pickett, S.T.A.; Parker, V.T. 1994. Avoiding the old pitfalls: opportunities in a new discipline. Restoration Ecology. 2(2): 75-79. doi:10.1111/j.1526-100X.1994.tb00044.x

Pickett, S.T.A.; Collins, S.L.; Armesto, J.J. 1987a. A hierarchical consideration of causes and mechanisms of succession. Vegetatio. 69(1/3): 109-114. https://www.jstor.org/stable/20038107

Pickett, S.T.A.; Collins, S.L.; Armesto, J.J. 1987b. Models, mechanisms and pathways of succession. Botanical Review. 53(3): 335-371. doi:10.1007/BF02858321

Pickett, S.T.A.; Kolasa, J.; Armesto, J.J.; Collins, S.L. 1989. The ecological concept of disturbance and its expression at various hierarchical levels. Oikos. 54(2): 129-136. doi:10.2307/3565258

Pickett, S.T.A.; Ostfeld, R.S.; Shachak, M.; Likens, G.E., eds. 1997. The ecological basis of conservation: heterogeneity, ecosystems, and biodiversity. New York: Chapman & Hall. 466 p. isbn:0-412-09851-2

Pielke, R.A.; Avissar, R. 1990. Influence of landscape structure on local and regional climate. Landscape Ecology. 4(2/3): 133-155. doi:10.1007/BF00132857

Pierce, J.L.; Meyer, G.A.; Jull, A.J.T. 2004. Fire-induced erosion and millennial-scale climate change in northern ponderosa pine forests. Nature. 432(7013): 87-90. doi:10.1038/nature03058

Pierce, D.W.; Barnett, T.P.; Hidalgo, H.G.; Das, T.; Bonfils, C.; Santer, B.D.; Bala, G.; Dettinger, M.D.; Cayan, D.R.; Mirin, A.; Wood, A.W.; Nozawa, T. 2008. Attribution of declining western US snowpack to human effects. Journal of Climate. 21(23): 6425-6444. doi:10.1175/2008JCLI2405.1

Pierotti, R.; Wildcat, D. 2000. Traditional ecological knowledge: the third alternative. Ecological Applications. 10(5): 1333-1340. doi:10.1890/1051-0761(2000)010[1333:TEKTTA]2.0.CO;2

Pinkard, E.A.; Battaglia, M.; Roxburgh, S.; O'Grady, A.P. 2011. Estimating forest net primary production under changing climate: adding pests into the equation. Tree Physiology. 31(7): 686-699. doi:10.1093/treephys/tpr054

Pitman, G.B.; Perry, D.A.; Emmingham, W.H. 1982. Thinning to prevent mountain pine beetles in lodgepole and ponderosa pine. Exten. Circ. 1106. Corvallis, OR: Oregon State University, Extension Service. 4 p. https://ir.library.oregonstate.edu/concern/administrative_report_or_publications/4f16c295t

Pohl, K.A.; Hadley, K.S.; Arabas, K.B. 2002. A 545-year drought reconstruction for central Oregon. Physical Geography. 23(4): 302-320. doi:10.2747/0272-3646.23.4.302

Pollack, H.N. 2005. Uncertain science...uncertain world. Cambridge, UK: Cambridge University Press. 243 p. isbn:0-521-61910-6

Pollock, M.M.; Beechie, T.J.; Imaki, H. 2012. Using reference conditions in ecosystem restoration: an example for riparian conifer forests in the Pacific Northwest. Ecosphere. 3(11): art98 (23 p). doi:10.1890/ES12-00175.1

Pommerening, A.; Sánchez Meador, A.J. 2018. Tamm review: Tree interactions between myth and reality. Forest Ecology and Management. 424: 164-176. doi:10.1016/j.foreco.2018.04.051

Portier, J.; Gauthier, S.; Cyr, G.; Bergeron, Y. 2018. Does time since fire drive live aboveground biomass and stand structure in low fire activity boreal forests? Impacts on their management. Journal of Environmental Management. 225: 346-355. doi:10.1016/j.jenvman.2018.07.100

Potts, D.F. 1985. Water potential of forest duff and its possible relationship to regeneration success in the northern Rocky Mountains. Canadian Journal of Forest Research. 15(2): 464-468. doi:10.1139/x85-074

Povak, N.A.; Hessburg, P.F.; Salter, R.B. 2018. Evidence for scale-dependent topographic controls on wildfire spread. Ecosphere. 9(10): e02443 (18 p). doi:10.1002/ecs2.2443

Powell, D.C. 1994. Effects of the 1980s western spruce budworm outbreak on the Malheur National Forest in northeastern Oregon. Tech. Pub. R6-FI&D-TP-12-94. Portland, OR: USDA Forest Service, Pacific Northwest Region. 176 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5358589.pdf

Powell, D.C. 1995. Annual forest monitoring and evaluation report for FY1994; file designation 1920-2-3 memorandum to Planning Staff Officer. Pendleton, OR: USDA Forest Service, Umatilla National Forest, Supervisor's Office. 18 p.

Powell, D.C. 1997. Forest vegetation report; Tower Fire Ecosystem Analysis. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest, Supervisor's Office. 56 p. https://fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015965.pdf

Powell, D.C. 1998. Upland forest vegetation analysis; Desolation Watershed ecosystem analysis. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest, Supervisor's Office. 77 p. http://hdl.handle.net/1794/6649

Powell, D.C. 1999a. Historical references about vegetation conditions: a bibliography with abstracts. Tech. Pub. F14-SO-TP-05-99. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 310 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprd3798058.pdf

Powell, D.C. 1999b. Suggested stocking levels for forest stands in northeastern Oregon and southeastern Washington: an implementation guide for the Umatilla National Forest. Tech. Pub. F14-SO-TP-03-99. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 300 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5405482.pdf

Powell, D.C. 2000. Potential vegetation, disturbance, plant succession, and other aspects of forest ecology. Tech. Pub. F14-SO-TP-09-00. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 88 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5358579.pdf

Powell, D.C. 2008a. Aspen community types of the Pike and San Isabel National Forests in south-central Colorado. R2-ECOL-88-01. Update edition. Lakewood, CO: USDA Forest Service, Rocky Mountain Region. 278 p. https://fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_016041.pdf

Powell, D.C. 2008b. Recording the changes: Field guide to establishing and maintaining permanent camera point systems. Tech. Pub. FS-14-SO-09-06. Second edition. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 23 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015665.pdf

Powell, D.C. 2010. Estimating crown fire susceptibility for project planning. Fire Management Today. 70(3): 8-15. http://www.fs.fed.us/fire/fmt/fmt_pdfs/FMT70-3.pdf

Powell, D.C. 2012a. A stage is a stage is a stage…or is it? Successional stages, structural stages, seral stages. White Pap. F14-SO-WP-Silv-10. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 15 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd546903.pdf

Powell, D.C. 2012b. Historical fires in headwaters portion of Tucannon River watershed. White Pap. F14-SO-WP-Silv-21. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 59 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5413729.pdf

Powell, D.C. 2013a. Description of composite vegetation database. White Pap. F14-SO-WP-Silv-2. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 39 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5326218.pdf

Powell, D.C. 2013b. Stand density protocol for mid-scale assessments. White Pap. F14-SO-WP-Silv-36. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 67 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5413734.pdf

Powell, D.C. 2014. Active management of Blue Mountains dry forests: silvicultural considerations. White Pap. F14-SO-WP-Silv-4. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest, Supervisor's Office. 238 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd491870.pdf

Powell, D.C. 2019a. Historical vegetation mapping. White Pap. F14-SO-WP-Silv-23. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 103 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5413730.pdf

Powell, D.C. 2019b. Range of variation recommendations for dry, moist, and cold forests. White Pap. F14-SO-WP-Silv-3. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 79 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5326219.pdf

Powell, D.C. 2019c. Using General Land Office survey notes to characterize historical vegetation conditions for Umatilla National Forest. White Pap. F14-SO-WP-Silv-41. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 112 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5413735.pdf

Powell, E.N.; Raffa, K.F. 2011. Fire injury reduces inducible defenses of lodgepole pine against mountain pine beetle. Journal of Chemical Ecology. 37(11): 1184-1192. doi:10.1007/s10886-011-0031-4

Powell, D.C.; Rockwell, V.A.; Townsley, J.J.; Booser, J.; Bulkin, S.P.; Martin, T.H.; Obedzinski, B.; Zensen, F. 2001. Forest density management: recent history and trends for the Pacific Northwest Region. Tech. Pub. R6-NR-TM-TP-05-01. Portland, OR: USDA Forest Service, Pacific Northwest Region. 21 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprd3798353.pdf

Powell, D.C.; Johnson, C.G., Jr.; Crowe, E.A.; Wells, A.; Swanson, D.K. 2007. Potential vegetation hierarchy for the Blue Mountains section of northeastern Oregon, southeastern

Washington, and west-central Idaho. Gen. Tech. Rep. PNW-GTR-709. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 87 p. http://www.treesearch.fs.fed.us/pubs/27598

Powell, E.N.; Townsend, P.A.; Raffa, K.F. 2012. Wildfire provides refuge from local extinction but is an unlikely driver of outbreaks by mountain pine beetle. Ecological Monographs. 82(1): 69-84. doi:10.1890/11-0607.1

Power, M.; Marlon, J.; Ortiz, N.; Bartlein, P.; Harrison, S.; Mayle, F.; Ballouche, A.; Bradshaw, R.; Carcaillet, C.; Cordova, C.; and others. 2008. Changes in fire regimes since the Last Glacial Maximum: an assessment based on a global synthesis and analysis of charcoal data. Climate Dynamics. 30(7): 887-907. doi:10.1007/s00382-007-0334-x

Powers, R.F. 1989. Maintaining long-term forest productivity in the Pacific Northwest: defining the issues. In: Perry, D.A.; Meurisse, R.; Thomas, B.; Miller, R.; Boyle, J.; Means, J.; Perry, C.R.; Powers, R.F., eds. Maintaining the long-term productivity of Pacific Northwest forest ecosystems. Portland, OR: Timber Press: 3-16. isbn:0-88192-144-0

Powers, R.F., tech. ed. 2007. Restoring fire-adapted ecosystems: proceedings of the 2005 National Silviculture Workshop. Gen. Tech. Rep. PSW-GTR-203. Albany, CA: USDA Forest Service, Pacific Southwest Research Station. 306 p. https://www.fs.usda.gov/treesearch/pubs/25886

Powers, R.F.; Scott, D.A.; Sanchez, F.G.; Voldseth, R.A.; Page-Dumroese, D.; Elioff, J.D.; Stone, D.M. 2005. The North American long-term soil productivity experiment: findings from the first decade of research. Forest Ecology and Management. 220: 31-50. doi:10.1016/j.foreco.2005.08.003

Preisler, H.K.; Mitchell, R.G. 1993. Colonization patterns of the mountain pine beetle in thinned and unthinned lodgepole pine stands. Forest Science. 39(3): 528-545. doi:10.1093/forestscience/39.3.528

Preisler, H.K.; Burgan, R.E.; Eidenshink, J.C.; Klaver, J.M.; Klaver, R.W. 2009. Forecasting distributions of large federal-lands fires utilizing satellite and gridded weather information. International Journal of Wildland Fire. 18(5): 508-516. doi:10.1071/WF08032

Prescott, C.E.; Corbin, J.P.; Parkinson, D. 1989. Biomass, productivity, and nutrient-use efficiency of aboveground vegetation in four Rocky Mountain coniferous forests. Canadian Journal of Forest Research. 19(3): 309-317. doi:10.1139/x89-046

Prescott, C.E.; Corbin, J.P.; Parkinson, D. 1992. Availability of nitrogen and phosphorus in the forest floors of Rocky Mountain coniferous forests. Canadian Journal of Forest Research. 22(4): 593-600. doi:10.1139/x92-079

Prescott, C.E.; Zabek, L.M.; Staley, C.L.; Kabzems, R. 2000a. Decomposition of broadleaf and needle litter in forests of British Columbia: influences of litter type, forest type, and litter mixtures. Canadian Journal of Forest Research. 30(11): 1742-1750. doi:10.1139/x00-097

Prescott, C.E.; Blevins, L.L.; Staley, C.L. 2000b. Effects of clear-cutting on decomposition rates of litter and forest floor in forests of British Columbia. Canadian Journal of Forest Research. 30(11): 1751-1757. doi:10.1139/x00-102

Prescott, C.E.; Maynard, D.G.; Laiho, R. 2000c. Humus in northern forests: friend or foe? Forest Ecology and Management. 133(1-2): 23-36. doi:10.1016/s0378-1127(99)00295-9

Price, O.F. 2012. The drivers of effectiveness of prescribed fire treatment. Forest Science. 58(6): 606-617. doi:10.5849/forsci.11-002

Prober, S.M.; Dunlop, M. 2011. Climate change: a cause for new biodiversity conservation objectives but let's not throw the baby out with the bathwater. Ecological Management and Restoration. 12(1): 2-3. doi:10.1111/j.1442-8903-2011.00563.x

Puettmann, K.J.; Coates, K.D.; Messier, C. 2009. A critique of silviculture: managing for complexity. Washington, DC: Island Press. 189 p. isbn:978-1-59726-146-3

Pugnaire, F.I.; Morillo, J.A.; Peñuelas, J.; Reich, P.B.; Bardgett, R.D.; Gaxiola, A.; Wardle, D.A.; van der Putten, W.H. 2019. Climate change effects on plant-soil feedbacks and consequences for biodiversity and functioning of terrestrial ecosystems. Science Advances. 5(11): eaaz1834 (12 p). doi:10.1126/sciadv.aaz1834

Puritch, G.S. 1973. Effect of water stress on photosynthesis, respiration, and transpiration of four *Abies* species. Canadian Journal of Forest Research. 3(2): 293-298. doi:10.1139/x73-040

Putz, F.E.; Parker, G.G.; Archibald, R.M. 1984. Mechanical abrasion and intercrown spacing. American Midland Naturalist. 112(1): 24-28. doi:10.2307/2425452

Pyne, S.J. 1997. America's fires: management on wildlands and forests. Durham, NC: Forest History Society. 54 p. isbn:0-89030-053-4

Pyne, S.J.; Andrews, P.L.; Laven, R.D. 1996. Introduction to wildland fire. 2nd edition. New York: John Wiley & Sons. 769 p. isbn:0-471-54913-4

Quaempts, E.J. 1993. An analysis of huckleberry treatment alternatives for Black Mountain. Unpublished report prepared for Technical Fire Management program. Walla Walla, WA: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest, Walla Walla Ranger District. 48 p.

Quaempts, E.J.; Jones, K.L.; O'Daniel, S.J.; Beechie, T.J.; Poole, G.C. 2018. Aligning environmental management with ecosystem resilience: a First Foods example from the Confederated Tribes of the Umatilla Indian Reservation, Oregon, USA. Ecology and Society. 23(2): 29 (20 p). doi:10.5751/ES-10080-230229

Quigley, T.M. 1992. Forest health in the Blue Mountains: social and economic perspectives. Gen. Tech. Rep. PNW-GTR-296. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 9 p. http://www.treesearch.fs.fed.us/pubs/9033

Quigley, T.M.; Arbelbide, S.J., tech. eds. 1997. An assessment of ecosystem components in the Interior Columbia Basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 4 volumes: 1-2066. http://www.treesearch.fs.fed.us/pubs/24921

Quigley, T.M.; Haynes, R.W.; Graham, R.T. 1996. Integrated scientific assessment for ecosystem management in the interior Columbia basin. Gen. Tech. Rep. PNW-GTR-382. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 303 p. http://www.treesearch.fs.fed.us/pubs/25384

Quigley, T.M.; Hayes, J.L.; Starr, L.; Daterman, G.E. 2001. Improving forest health and productivity in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 234-251. http://hdl.handle.net/2376/999

Quine, C.P.; Gardiner, B.A. 2007. Understanding how the interaction of wind and trees results in windthrow, stem breakage, and canopy gap formation. In: Johnson, E.A.; Miyanishi, K., eds. Plant disturbance ecology: the process and the response. Amsterdam: Academic Press: 103-155. isbn:978-0-12-088778-1

Quine, C.P.; Humphrey, J.W.; Ferris, R. 1999. Should the wind disturbance patterns observed in natural forests be mimicked in planted forests in the British uplands? Forestry. 72(4): 337-358. doi:10.1093/forestry/72.4.337

Radeloff, V.C.; Mladenoff, D.J.; Boyce, M.S. 2000. Effects of interacting disturbances on landscape patterns: budworm defoliation and salvage logging. Ecological Applications. 10(1): 233-247. doi:10.1890/1051-0761(2000)010[0233:EOIDOL]2.0.CO;2

Raffa, K.F.; Berryman, A.A. 1982. Physiological differences between lodgepole pines resistant and susceptible to the mountain pine beetle and associated microorganisms. Environmental Entomology. 11(2): 486-492. doi:10.1093/ee/11.2.486

Raffa, K.F.; Berryman, A.A. 1983. The role of host plant resistance in the colonization behavior and ecology of bark beetles (Coleoptera: Scolytidae). Ecological Monographs. 53(1): 27-

49. doi:10.2307/1942586

Raffa, K.F.; Aukema, B.H.; Bentz, B.J.; Carroll, A.L.; Hicke, J.A.; Turner, M.G.; Romme, W.H. 2008. Cross-scale drivers of natural disturbances prone to anthropogenic amplification: the dynamics of bark beetle eruptions. BioScience. 58(6): 501-517. doi:10.1641/B580607

Ragenovich, I.R.; Mitchell, R.G. 2006. Balsam woolly adelgid. Forest Insect & Disease Leaflet 118; R6-NR-FID-PR-04-2006. Portland, OR: USDA Forest Service. 11 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043667.pdf

Rainville, R.; White, R.; Barbour, J., tech. eds. 2008. Assessment of timber availability from forest restoration within the Blue Mountains of Oregon. Gen. Tech. Rep. PNW-GTR-752. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 65 p. http://www.treesearch.fs.fed.us/pubs/30559

Rambo, T.R.; North, M.P. 2009. Canopy microclimate response to pattern and density of thinning in a Sierra Nevada forest. Forest Ecology and Management. 257(2): 435-442. doi:10.1016/j.foreco.2008.09.029

Rambo, T.; Schowalter, T.; North, M. 2014. Canopy arthropod responses to thinning and burning treatments in old-growth mixed-conifer forest in the Sierra Nevada, California. Forest Ecology and Management. 326: 91-100. doi:10.1016/j.foreco.2014.04.014

Rammig, A.; Fahse, L.; Bebi, P.; Bugmann, H. 2007. Wind disturbance in mountain forests: Simulating the impact of management strategies, seed supply, and ungulate browsing on forest succession. Forest Ecology and Management. 242(2-3): 142-154. doi:10.1016/j.foreco.2007.01.036

Rancourt, S.J.; Rule, M.I.; O'Connell, M.A. 2007. Maternity roost site selection of big brown bats in ponderosa pine forests of the channeled scablands of northeastern Washington State, USA. Forest Ecology and Management. 248(3): 183-192. doi:10.1016/j.foreco.2007.05.005

Rapp, V. 2002a. Restoring complexity: second-growth forests and habitat diversity. Portland, OR: Science Update. 1: 1-10. http://www.fs.fed.us/pnw/pubs/science-update1.pdf

Rapp, V. 2002b. Fire risk in east-side forests. Portland, OR: Science Update. 2: 1-11. http://www.fs.fed.us/pnw/scienceupdate2.pdf

Rapport, D.J.; Whitford, W.G. 1999. How ecosystems respond to stress. BioScience. 49(3): 193-203. doi:10.2307/1313509

Rapport, D.J.; Regier, H.A.; Hutchinson, T.C. 1985. Ecosystem behavior under stress. American Naturalist. 125(5): 617-640. doi:10.1086/284368

Rasmussen, M.; Lord, R.; Vickery, B.; McKetta, C.; Green, D.; Green, M.; Hemstrom, M.; Potiowsky, T.; Renfro, J. 2012. National forest health restoration: an economic assessment of forest restoration on Oregon's eastside national forests. Place of publication unknown: Publisher unknown. 84 p. http://www.oregon.gov/gov/docs/OR_Forest_Restoration_Econ_Assessment_Nov_2012.pdf

Raymond, C.L.; McKenzie, D. 2012. Carbon dynamics of forests in Washington, USA: 21st century projections based on climate-driven changes in fire regimes. Ecological Applications. 22(5): 1589-1611. doi:10.1890/11-1851.1

Raymond, P.; Royo, A.A.; Prévost, M.; Dumais, D. 2018. Assessing the single-tree and small group selection cutting system as intermediate disturbance to promote regeneration and diversity in temperate mixedwood stands. Forest Ecology and Management. 430: 21-32. doi:10.1016/j.foreco.2018.07.054

Reaves, J.L.; Shaw, C.G., III; Martin, R.E.; Mayfield, J.E. 1984. Effects of ash leachates on growth and development of *Armillaria mellea* in culture. Res. Note PNW-418. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 11 p. https://www.fs.usda.gov/treesearch/pubs/26797

Reaves, J.L.; Shaw, C.G., III; Mayfield, J.E. 1990. The effects of *Trichoderma* spp. isolated from burned and non-burned forest soils on the growth and development of *Armillaria ostoyae* in culture. Northwest Science. 64(1): 39-44. http://hdl.handle.net/2376/1656

Redak, R.A.; Cates, R.G. 1984. Douglas-fir (*Pseudotsuga menziesii*)–spruce budworm (*Choristoneura occidentalis*) interactions: the effect of nutrition, chemical defenses, tissue phenology, and tree physical parameters on budworm success. Oecologia. 62(1): 61-67. doi:10.1007/bf00377374

Redmond, M.D.; Kelsey, K.C. 2018. Topography and overstory mortality interact to control tree regeneration in spruce-fir forests of the southern Rocky Mountains. Forest Ecology and Management. 427: 106-113. doi:10.1016/j.foreco.2018.05.057

Reed, W.J. 1994. Estimating the historic probability of stand-replacement fire using the age-class distribution of undisturbed forest. Forest Science. 40(1): 104-119. doi:10.1093/forestscience/40.1.104

Reed, W.J.; McKelvey, K.S. 2002. Power-law behaviour and parametric models for the size-distribution of forest fires. Ecological Modelling. 150(3): 239-254. doi:10.1016/S0304-3800(01)00483-5

Reed, W.J.; Larsen, C.P.S.; Johnson, E.A.; MacDonald, G.M. 1998. Estimation of temporal variations in historical fire frequency from time-since-fire map data. Forest Science. 44(3): 465-475. doi:10.1093/forestscience/44.3.465

Reeves, G.H.; Duncan, S.L. 2009. Ecological history vs. social expectations: managing aquatic ecosystems. Ecology and Society. 14(2): article 8 [online]. http://www.ecologyandsociety.org/vol14/iss2/art8/

Reeves, G.H.; Bisson, P.A.; Rieman, B.E.; Benda, L.E. 2006. Postfire logging in riparian areas. Conservation Biology. 20(4): 994-1004. doi:10.1111/j.1523-1739.2006.00502.x

Rehfeldt, G.E. 1994. Evolutionary genetics, the biological species, and the ecology of the interior cedar-hemlock forests. In: Baumgartner, D.M.; Lotan, J.E.; Tonn, J.R., comps. Interior cedar-hemlock-white pine forests: ecology and management. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 91-100.

Rehfeldt, G.E. 1995. Genetic variation, climate models and the ecological genetics of *Larix occidentalis*. Forest Ecology and Management. 78(1-3): 21-37. doi:10.1016/0378-1127(95)03602-4

Rehfeldt, G.E.; Crookston, N.L.; Warwell, M.V.; Evans, J.S. 2006. Empirical analyses of plant-climate relationships for the western United States. International Journal of Plant Sciences. 167(6): 1123-1150. doi:10.1086/507711

Rehfeldt, G.E.; Ferguson, D.E.; Crookston, N.L. 2008. Quantifying the abundance of co-occurring conifers along inland northwest (USA) climate gradients. Ecology. 89(8): 2127-2139. doi:10.1890/06-2013.1

Rehfeldt, G.E.; Ferguson, D.E.; Crookston, N.L. 2009. Aspen, climate, and sudden decline in western USA. Forest Ecology and Management. 258(11): 2353-2364. doi:10.1016/j.foreco.2009.06.005

Reice, S.R. 1994. Nonequilibrium determinants of biological community structure. American Scientist. 82(5): 424-435. https://www.jstor.org/stable/29775279

Reich, P.B.; Bakken, P.; Carlson, D.; Frelich, L.E.; Friedman, S.K.; Grigal, D.F. 2001. Influence of logging, fire, and forest type on biodiversity and productivity in southern boreal forests. Ecology. 82(10): 2731-2748. doi:10.1890/0012-9658(2001)082[2731:IOLFAF]2.0.CO;2

Reid, M. 2008. Mountain pine beetle dispersal through managed and unmanaged landscapes. Mountain Pine Beetle Working Paper 2008-17. Victoria, BC: Natural Resources Canada, Canadian Forest Service, Pacific Forestry Centre. 17 p. https://cfs.nrcan.gc.ca/pubwarehouse/pdfs/29221.pdf

Reid, R.W.; Whitney, H.S.; Watson, J.A. 1967. Reactions of lodgepole pine to attack by *Dendroctonus ponderosae* Hopkins and blue stain fungi. Canadian Journal of Botany. 45(7): 1115-1126. doi:10.1139/b67-116

Reimoser, F.; Gossow, H. 1996. Impact of ungulates on forest vegetation and its dependence on the silvicultural system. Forest Ecology and Management. 88(1-2): 107-119. doi:10.1016/S0378-1127(96)03816-9

Reimoser, F.; Armstrong, H.; Suchant, R. 1999. Measuring forest damage of ungulates: what should be considered. Forest Ecology and Management. 120(1-3): 47-58. doi:10.1016/S0378-1127(98)00542-8

Reinhardt, E.; Holsinger, L. 2010. Effects of fuel treatments on carbon-disturbance relationships in forests of the northern Rocky Mountains. Forest Ecology and Management. 259(8): 1427-1435. doi:10.1016/j.foreco.2010.01.015

Reinhardt, E.D.; Graham, R.T.; Jain, T.B.; Simmerman, D.G. 1994. Short-term effects of prescribed fire in grand fir-white pine-western hemlock slash fuels. In: Baumgartner, D.M.; Lotan, J.E.; Tonn, J.R., comps. Interior cedar-hemlock-white pine forests: ecology and management. Pullman, WA: Washington State University, Department of Natural Resource Sciences: 221-225.

Reinhardt, E.D.; Keane, R.E.; Calkin, D.E.; Cohen, J.D. 2008. Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. Forest Ecology and Management. 256(12): 1997-2006. doi:10.1016/j.foreco.2008.09.016

Remy, C.C.; Fouquemberg, C.; Asselin, H.; Andrieux, B.; Magnan, G.; Brossier, B.; Grondin, P.; Bergeron, Y.; Talon, B.; Girardin, M.P.; Blarquez, O.; Bajolle, L.; Ali, A.A. 2018. Guidelines for the use and interpretation of palaeofire reconstructions based on various archives and proxies. Quaternary Science Reviews. 193: 312-322. doi:10.1016/j.quascirev.2018.06.010

Repetto, R. 1988. The forest for the trees? Government policies and the misuse of forest resources. Washington, DC: World Resources Institute. 105 p. isbn:0-915825-25-2

Restaino, J.C.; Peterson, D.L. 2013. Wildfire and fuel treatment effects on forest carbon dynamics in the western United States. Forest Ecology and Management. 303: 46-60. doi:10.1016/j.foreco.2013.03.043

Rhemtulla, J.M.; Mladenoff, D.J.; Clayton, M.K. 2009. Historical forest baselines reveal potential for continued carbon sequestration. Proceedings of the National Academy of Sciences. 106(15): 6082-6087. doi:10.1073/pnas.0810076106

Richards, B. 1992. As fires sear the west, Forest Service policies come under scrutiny. New York: Wall Street Journal, Tuesday, October 6th, pages A1-A2, A9.

Rickard, W.H. 1960. The distribution of small mammals in relation to the climax vegetation mosaic in eastern Washington and northern Idaho. Ecology. 41(1): 99-106. doi:10.2307/1931943

Ricklefs, R.E. 1987. Community diversity: relative roles of local and regional processes. Science. 235(4785): 167-171. doi:10.1126/science.235.4785.167

Rieman, B.; Dunham, J.; Clayton, J. 2006. Emerging concepts for management of river ecosystems and challenges to applied integration of physical and biological sciences in the Pacific Northwest, USA. International Journal of River Basin Management. 4(2): 85-97. doi:10.1080/15715124.2006.9635279

Rietveld, W.J. 1975. Phytotoxic grass residues reduce germination and initial root growth of ponderosa pine. Res. Pap. RM-153. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 15 p. https://www.fs.usda.gov/treesearch/pubs/23932

Riggs, R.A.; Tiedemann, A.R.; Cook, J.C.; Ballard, T.M.; Edgerton, P.J.; Vavra, M.; Krueger, W.C.; Hall, F.C.; Bryant, L.D.; Irwin, L.L.; Delcurto, T. 2000. Modification of mixed-conifer forests by ruminant herbivores in the Blue Mountains ecological province. Res. Pap. PNW-RP-527. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 77 p. http://www.treesearch.fs.fed.us/pubs/2941

Riley, K.L.; Loehman, R.A. 2016. Mid-21st-century climate changes increase predicted fire occurrence and fire season length, northern Rocky Mountains, United States. Ecosphere. 7(11): e01543 (19 p). doi:10.1002/ecs2.1543

Ripple, W.J.; Beschta, R.L. 2003. Wolf reintroduction, predation risk, and cottonwood recovery in Yellowstone National Park. Forest Ecology and Management. 184(1-3): 299-313. doi:10.1016/S0378-1127(03)00154-3

Ripple, W.J.; Beschta, R.L. 2004. Wolves, elk, willows, and trophic cascades in the upper Gallatin Range of southwestern Montana, USA. Forest Ecology and Management. 200(1-3): 161-181. doi:10.1016/j.foreco.2004.06.017

Ripple, W.J.; Beschta, R.L. 2005. Willow thickets protect young aspen from elk browsing after wolf reintroduction. Western North American Naturalist. 65(1): 118-122. https://scholarsarchive.byu.edu/wnan/vol65/iss1/14/

Ripple, W.J.; Beschta, R.L. 2006. Linking wolves to willows via risk-sensitive foraging by ungulates in the northern Yellowstone ecosystem. Forest Ecology and Management. 230(1-3): 96-106. doi:10.1016/j.foreco.2006.04.023

Ripple, W.J.; Beschta, R.L. 2007. Restoring Yellowstone's aspen with wolves. Biological Conservation. 138(3-4): 514-519. doi:10.1016/j.biocon.2007.05.006

Ripple, W.J.; Beschta, R.L. 2012. Trophic cascades in Yellowstone: The first 15 years after wolf reintroduction. Biological Conservation. 145(1): 205-213. doi:10.1016/j.biocon.2011.11.005

Ripple, W.J.; Larsen, E.J. 2000. Historic aspen recruitment, elk, and wolves in northern Yellowstone National Park, USA. Biological Conservation. 95(3): 361-370. doi:10.1016/S0006-3207(00)00014-8

Ripple, W.J.; Larsen, E.J. 2001. The role of postfire coarse woody debris in aspen regeneration. Western Journal of Applied Forestry. 16(2): 61-64. doi:10.1093/wjaf/16.2.61

Ripple, W.J.; Larsen, E.J.; Renkin, R.A.; Smith, D.W. 2001. Trophic cascades among wolves, elk and aspen on Yellowstone National Park's northern range. Biological Conservation. 102(3): 227-234. doi:10.1016/S0006-3207(01)00107-0

Ripple, W.J.; Beschta, R.L.; Painter, L.E. 2015. Trophic cascades from wolves to alders in Yellowstone. Forest Ecology and Management. 354: 254-260. doi:10.1016/j.foreco.2015.06.007

Rippy, R.C.; Stewart, J.E.; Zambino, P.J.; Klopfenstein, N.B.; Tirocke, J.M.; Kim, M.-S.; Thies, W.G. 2005. Root diseases in coniferous forests of the inland West: potential implications of fuel treatments. Gen. Tech. Rep. RMRS-GTR-141. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 32 p. http://www.treesearch.fs.fed.us/pubs/20469

Rist, L.; Moen, J. 2013. Sustainability in forest management and a new role for resilience thinking. Forest Ecology and Management. 310: 416-427. doi:10.1016/j.foreco.2013.08.033

Robbins, W.G. 1997. Landscapes of promise: the Oregon story, 1800-1940. Seattle, WA: University of Washington Press. 392 p. isbn:0-295-97632-2

Robbins, W.G.; Wolf, D.W. 1994. Landscape and the intermontane Northwest: an environmental history. Gen. Tech. Rep. PNW-GTR-319. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 32 p. http://www.treesearch.fs.fed.us/pubs/6224

Roberts, M.R. 2004. Response of the herbaceous layer to natural disturbance in North American forests. Canadian Journal of Botany. 82(9): 1273-1283. doi:10.1139/b04-091

Roberts, M.R. 2007. A conceptual model to characterize disturbance severity in forest harvests. Forest Ecology and Management. 242(1): 58-64. doi:10.1016/j.foreco.2007.01.043

Roberts, M.R.; Gilliam, F.S. 1995. Patterns and mechanisms of plant diversity in forested ecosystems: implications for forest management. Ecological Applications. 5(4): 969-977. doi:10.2307/2269348

Robichaud, P.R.; Jordan, P.; Lewis, S.A.; Ashmun, L.E.; Covert, S.A.; Brown, R.E. 2013a. Evaluating the effectiveness of wood shred and agricultural straw mulches as a treatment to reduce post-wildfire hillslope erosion in southern British Columbia, Canada. Geomorphology. 197: 21-33. doi:10.1016/j.geomorph.2013.04.024

Robichaud, P.R.; Lewis, S.A.; Wagenbrenner, J.W.; Ashmun, L.E.; Brown, R.E. 2013b. Postfire mulching for runoff and erosion mitigation: Part I: Effectiveness at reducing hillslope erosion rates. Catena. 105: 75-92. doi:10.1016/j.catena.2012.11.015

Robichaud, P.R.; Wagenbrenner, J.W.; Pierson, F.B.; Spaeth, K.E.; Ashmun, L.E.; Moffet, C.A. 2016. Infiltration and interrill erosion rates after a wildfire in western Montana, USA. Catena. 142: 77-88. doi:10.1016/j.catena.2016.01.027

Robinson, D. 1994. The responses of plants to non-uniform supplies of nutrients. New Phytologist. 127(4): 635-674. doi:10.1111/j.1469-8137.1994.tb02969.x

Roby, K.; Azuma, D. 1995. Changes in a reach of a northern California stream following wildfire. Environmental Management. 19(4): 591-600. doi:10.1007/bf02471970

Rochelle, J.A.; Lehmann, L.A.; Wisniewski, J., eds. 1999. Forest fragmentation: wildlife and management implications. Leiden, Netherlands: Brill Academic Publishers. 301 p. isbn:90-04-11388-6

Rodman, K.C.; Veblen, T.T.; Saraceni, S.; Chapman, T.B. 2019. Wildfire activity and land use drove 20th-century changes in forest cover in the Colorado Front Range. Ecosphere. 10(2): e02594 (27 p). doi:10.1002/ecs2.2594

Roe, A.L. 1967. Productivity indicators in western larch forests. Res. Note INT-59. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 4 p. https://archive.org/download/productivityindi59roea/productivityindi59roea.pdf

Roe, A.L.; Schmidt, W.C. 1965. Thinning western larch. Res. Pap. INT-16. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 10 p. https://archive.org/download/thinningwesternl16roea/thinningwesternl16roea.pdf

Roe, A.L.; Alexander, R.R.; Andrews, M.D. 1970. Engelmann spruce regeneration practices in the Rocky Mountains. Prod. Res. Rep. No. 115. Washington, DC: USDA Forest Service. 32 p. https://archive.org/details/CAT87210675

Roerick, T.M.; Cain, J.W.; Gedir, J.V. 2019. Forest restoration, wildfire, and habitat selection by female mule deer. Forest Ecology and Management. 447: 169-179. doi:10.1016/j.foreco.2019.05.067

Roeser, J., Jr. 1924. A study of Douglas fir reproduction under various cutting methods. Journal of Agricultural Research. 28(12): 1233-1242. https://naldc-legacy.nal.usda.gov/naldc/download.xhtml?id=IND43966833&content=PDF

Rogan, J.; Miller, J. 2007. Integrating GIS and remotely sensed data for mapping forest disturbance and change. In: Wulder, M.A.; Franklin, S.E., eds. Understanding forest disturbance and spatial pattern. Boca Raton, FL: CRC Press: 133-171.

Rogers, P. 1996. Disturbance ecology and forest management: a review of the literature. Gen. Tech. Rep. INT-GTR-336. Ogden, UT: USDA Forest Service, Intermountain Research Station. 16 p. https://www.fs.usda.gov/treesearch/pubs/42431

Rogers, B.M.; Neilson, R.P.; Drapek, R.; Lenihan, J.M.; Wells, J.R.; Bachelet, D.; Law, B.E.

2011. Impacts of climate change on fire regimes and carbon stocks of the U.S. Pacific Northwest. Journal of Geophysical Research. 116(G3): G03037 (13 p). doi:10.1029/2011JG001695

Rollins, G. 1982. Management experience with grand fir in the northern Blue Mountains. In: Oliver, C.D.; Kenady, R.M., eds. Proceedings of the biology and management of true fir in the Pacific Northwest symposium. Contribution No. 45. Seattle, WA: University of Washington, College of Forest Resources, Institute of Forest Resources: 253-255.

Rollins, M.G.; Keane, R.E.; Parsons, R.A. 2004. Mapping fuels and fire regimes using remote sensing, ecosystem simulation, and gradient modeling. Ecological Applications. 14(1): 75-95. doi:10.1890/02-5145

Roloff, G.J.; Mealey, S.P.; Clay, C.; Barry, J.; Yanish, C.; Neuenschwander, L. 2005. A process for modeling short- and long-term risk in the southern Oregon Cascades. Forest Ecology and Management. 211(1-2): 166-190. doi:10.1016/j.foreco.2005.02.006

Román-Cuesta, R.M.; Gracia, M.; Retana, J. 2009. Factors influencing the formation of unburned forest islands within the perimeter of a large forest fire. Forest Ecology and Management. 258(2): 71-80. doi:10.1016/j.foreco.2009.03.041

Romme, W.H.; Despain, D.G. 1989. Historical perspective on the Yellowstone fires of 1988. BioScience. 39(10): 695-699. doi:10.2307/1311000

Romme, W.H.; Knight, D.H. 1981. Fire frequency and subalpine forest succession along a topographic gradient in Wyoming. Ecology. 62(2): 319-326. doi:10.2307/1936706

Romme, W.H.; Knight, D.H.; Yavitt, J.B. 1986. Mountain pine beetle outbreaks in the Rocky Mountains: Regulators of primary productivity? American Naturalist. 127(4): 484-494. doi:10.1086/284497

Romme, W.H.; Everham, E.H.; Frelich, L.E.; Moritz, M.A.; Sparks, R.E. 1998. Are large, infrequent disturbances qualitatively different from small, frequent disturbances? Ecosystems. 1(6): 524-534. doi:10.1007/s100219900048

Romme, W.H.; Boyce, M.S.; Gresswell, R.; Merrill, E.H.; Minshall, G.W.; Whitlock, C.; Turner, M.G. 2011. Twenty years after the 1988 Yellowstone fires: lessons about disturbance and ecosystems. Ecosystems. 14(7): 1196-1215. doi:10.1007/s10021-011-9470-6

Ronco, F. 1972. Planting Engelmann spruce. Research Paper RM-89. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 24 p. https://archive.org/download/CAT92273281/CAT92273281.pdf

Rorig, M.L.; Ferguson, S.A. 1999. Characteristics of lightning and wildland fire ignition in the Pacific Northwest. Journal of Applied Meteorology. 38(11): 1565-1575. doi:10.1175/1520-0450(1999)038<1565:COLAWF>2.0.CO;2

Rose, S.L.; Perry, D.A.; Pilz, D.; Schoeneberger, M.M. 1983. Allelopathic effects of litter on the growth and colonization of mycorrhizal fungi. Journal of Chemical Ecology. 9(8): 1153-1162. doi:10.1007/BF00982218

Rosenberg, A.A.; Fogarty, M.J.; Sissenwine, M.P.; Beddington, J.R.; Shepherd, J.G. 1993. Achieving sustainable use of renewable resources. Science. 262(5135): 828-829. doi:10.1126/science.262.5135.828

Ross, D.W.; Daterman, G.E. 1994. Reduction of Douglas-fir beetle infestation of high-risk stands by antiaggregation and aggregation pheromones. Canadian Journal of Forest Research. 24(11): 2184-2190. doi:10.1139/x94-281

Ross, R.L.; Hunter, H.E. 1976. Climax vegetation of Montana based on soils and climate. Bozeman, MT: USDA Soil Conservation Service. 64 p. https://archive.org/download/climaxvegetation00ross/climaxvegetation00ross.pdf

Rotarangi, S.J.; Stephenson, J. 2014. Resilience pivots: Stability and identity in a social-ecological-cultural system. Ecology and Society. 19(1): 28 (10 p).

doi:10.5751/ES-06262-190128

Rothe, A.; Binkley, D. 2001. Nutritional interactions in mixed species forests: a synthesis. Canadian Journal of Forest Research. 31(11): 1855-1870. doi:10.1139/x01-120

Rothermel, R.C. 1991. Predicting behavior and size of crown fires in the northern Rocky Mountains. Res. Pap. INT-438. Ogden, UT: USDA Forest Service, Intermountain Research Station. 46 p. https://www.fs.usda.gov/treesearch/pubs/26696

Rowe, J.S. 1983. Concepts of fire effects on plant individuals and species. In: Wein, R.W.; MacLean, D.A., eds. The role of fire in northern circumpolar ecosystems. New York: John Wiley & Sons Ltd: 135-154.

Rowe, J.S. 1992. The ecosystem approach to forestland management. Forestry Chronicle. 68(2): 222-224. doi:10.5558/tfc68222-2

Rowe, J.S. 2000. Biodiversity at the landscape level. In: Bunnell, F.L.; Johnson, J.F., eds. Policy and practices for biodiversity in managed forests: the living dance. Vancouver, BC: UBC Press: 82-95.

Royo, A.A.; Carson, W.P. 2006. On the formation of dense understory layers in forests worldwide: consequences and implications for forest dynamics, biodiversity, and succession. Canadian Journal of Forest Research. 36(6): 1345-1362. doi:10.1139/X06-025

Rubin, B.D.; Manion, P.D.; Faber-Langendoen, D. 2006. Diameter distributions and structural sustainability in forests. Forest Ecology and Management. 222(1-3): 427-438. doi:10.1016/j.foreco.2005.10.049

Rubio, A.; Gavilán, R.; Escudero, A. 1999. Are soil characteristics and understorey composition controlled by forest management? Forest Ecology and Management. 113(2-3): 191-200. doi:10.1016/S0378-1127(98)00425-3

Ruddell, S.; Sampson, R.; Smith, M.; Giffen, R.; Cathcart, J.; Hagan, J.; Sosland, D.; Godbee, J.; Heissenbuttel, J.; Lovett, S.; Helms, J.; Price, W.; Simpson, R. 2007. The role for sustainably managed forests in climate change mitigation. Journal of Forestry. 105(6): 314-319. doi:10.1093/jof/105.6.314

Rudnicki, M.; Lieffers, V.J.; Silins, U. 2003. Stand structure governs the crown collisions of lodgepole pine. Canadian Journal of Forest Research. 33(7): 1238-1244. doi:10.1139/x03-055

Ruel, J-C. 1995. Understanding windthrow: silvicultural implications. Forestry Chronicle. 71(4): 434-445. doi:10.5558/tfc71434-4

Russell, E.W.B. 1983. Indian-set fires in the forests of the northeastern United States. Ecology. 64(1): 78-88. doi:10.2307/1937331

Russell, C.E.; Berryman, A.A. 1976. Host resistance to the fir engraver beetle. 1. Monoterpene composition of *Abies grandis* pitch blisters and fungus-infected wounds. Canadian Journal of Botany. 54(1-2): 14-18. doi:10.1139/b76-003

Russell, W.H.; McBride, J.R. 2001. The relative importance of fire and watercourse proximity in determining stand composition in mixed conifer riparian forests. Forest Ecology and Management. 150(3): 259-265. doi:10.1016/s0378-1127(00)00586-7

Ryan, R.B. 1983. Population density and dynamics of larch casebearer (Lepidoptera: Coleophoridae) in the Blue Mountains of Oregon and Washington before the build-up of exotic parasites. Canadian Entomologist. 115(9): 1095-1102. doi:10.4039/Ent1151095-9

Ryan, K.C.; Pickford, S.G. 1978. Physical properties of woody fuels in the Blue Mountains of Oregon and Washington. Res. Note PNW-315. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 10 p. Physical_properties_of_woody_fuels_in_th

Ryan, S.E.; Dwire, K.A.; Dixon, M.K. 2011. Impacts of wildfire on runoff and sediment loads at Little Granite Creek, western Wyoming. Geomorphology. 129(1): 113-130.

doi:10.1016/j.geomorph.2011.01.017

Ryerson, D.E.; Swetnam, T.W.; Lynch, A.M. 2003. A tree-ring reconstruction of western spruce budworm outbreaks in the San Juan Mountains, Colorado, USA. Canadian Journal of Forest Research. 33(6): 1010-1028. doi:10.1139/x03-026

Ryker, R.A. 1975. A survey of factors affecting regeneration of Rocky Mountain Douglas-fir. Res. Pap. INT-174. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 19 p. https://archive.org/download/surveyoffactorsa174ryke/surveyoffactorsa174ryke.pdf

Sackville Hamilton, N.R.; Matthew, C.; Lemaire, G. 1995. In defence of the -3/2 boundary rule: a re-evaluation of self-thinning concepts and status. Annals of Botany. 76(6): 569-577. doi:10.1006/anbo.1995.1134

Safford, H.D.; Stevens, J.T.; Merriam, K.; Meyer, M.D.; Latimer, A.M. 2012. Fuel treatment effectiveness in California yellow pine and mixed conifer forests. Forest Ecology and Management. 274: 17-28. doi:10.1016/j.foreco.2012.02.013

Safranyik, L.; Nevill, R.; Morrison, D. 1998. Effects of stand density management on forest insects and diseases. Tech. Tran. Note No. 12. Victoria, BC: Natural Resources Canada, Canadian Forest Service, Pacific Forestry Center. 4 p. http://bookstore.cfs.nrcan.gc.ca/detail_e.php?recid=35719

SAF Task Force. 2011. Forest carbon stocks and flows. Journal of Forestry. 109(Suppl_1): S14-S20. doi:10.1093/jof/109.s1.S14

Sagarin, R.; Pauchard, A. 2010. Observational approaches in ecology open new ground in a changing world. Frontiers in Ecology and the Environment. 8(7): 379-386. doi:10.1890/090001

Sala, A. 2006. Hydraulic compensation in northern Rocky Mountain conifers: does successional position and life history matter? Oecologia. 149(1): 1-11. doi:10.1007/s00442-006-0420-5

Sala, A.; Carey, E.V.; Callaway, R.M. 2001. Dwarf mistletoe affects whole-tree water relations of Douglas fir and western larch primarily through changes in leaf to sapwood ratios. Oecologia. 126(1): 42-52. doi:10.1007/s004420000503

Salathé, E.P., Jr. 2006. Influences of a shift in North Pacific storm tracks on western North American precipitation under global warming. Geophysical Research Letters. 33(19): L19820 (4 p). doi:10.1029/2006gl026882

Sallabanks, R.; Arnett, E.B. 2005. Accommodating birds in managed forests of North America: a review of bird-forestry relationships. In: Ralph, C.J.; Rich, T.D., eds. Bird conservation implementation and integration in the Americas: proceedings of the third International Partners in Flight conference. Gen. Tech. Rep. PSW-GTR-191. Albany, CA: USDA Forest Service, Pacific Southwest Research Station: 345-372. http://www.treesearch.fs.fed.us/pubs/31705

Sallabanks, R.; Christoffersen, N.D.; Weatherford, W.W.; Anderson, R. 2005. Restoring high priority habitats for birds: Aspen and pine in the interior west. In: Ralph, C.J.; Rich, T.D., eds. Bird conservation implementation and integration in the Americas: proceedings of the third International Partners in Flight conference. Gen. Tech. Rep. PSW-GTR-191. Albany, CA: USDA Forest Service, Pacific Southwest Research Station: 391-404. https://www.fs.usda.gov/treesearch/pubs/31708

Sallabanks, R.; Riggs, R.A.; Cobb, L.E. 2002. Bird use of forest structural classes in grand fir forests of the Blue Mountains, Oregon. Forest Science. 48(2): 311-321. doi:10.1093/forestscience/48.2.311

Sallabanks, R.; Riggs, R.A.; Cobb, L.E.; Dodson, S.W. 2006. Bird-habitat relationships in grand fir forests of the Blue Mountains, Oregon. Forest Science. 52(5): 489-502. doi:10.1093/forestscience/52.5.489

Sambaraju, K.R.; Carroll, A.L.; Zhu, J.; Stahl, K.; Moore, R.D.; Aukema, B.H. 2012. Climate change could alter the distribution of mountain pine beetle outbreaks in western Canada. Ecography. 35(3): 211-223. doi:10.1111/j.1600-0587.2011.06847.x

Sambaraju, K.R.; Carroll, A.L.; Aukema, B.H. 2019. Multiyear weather anomalies associated with range shifts by the mountain pine beetle preceding large epidemics. Forest Ecology and Management. 438: 86-95. doi:10.1016/j.foreco.2019.02.011

Sampson, R.N.; Adams, D.L., eds. 1994. Assessing forest ecosystem health in the inland west. Binghamton, NY: Food Products Press (Haworth Press). 461 p. isbn:1-56022-052-X

Sampson, R.N.; Adams, D.L.; Hamilton, S.S.; Mealey, S.P.; Steele, R.; Van De Graaff, D. 1994. Assessing forest ecosystem health in the inland west. Journal of Sustainable Forestry. 2(1-2): 3-10. doi:10.1300/J091v02n01_01

Sampson, R.N.; Atkinson, R.D.; Lewis, J.W., eds. 2000. Mapping wildfire hazards and risks. New York: Food Products Press. 343 p. isbn:1-56022-073-2

Savage, M. 1997. The role of anthropogenic influences in a mixed-conifer forest mortality episode. Journal of Vegetation Science. 8(1): 95-104. doi:10.2307/3237247

Savory, A. 1988. Holistic resource management. Washington, DC: Island Press. 564 p. isbn:0-933280-61-0

Saxe, S.; Hogue, T.S.; Hay, L. 2018. Characterization and evaluation of controls on post-fire streamflow response across western US watersheds. Hydrology and Earth System Sciences. 22(2): 1221-1237. doi:10.5194/hess-22-1221-2018

Schaaf, M.D.; Sandberg, D.V.; Schreuder, M.D.; Riccardi, C.L. 2007. A conceptual framework for ranking crown fire potential in wildland fuelbeds. Canadian Journal of Forest Research. 37(12): 2464-2478. doi:10.1139/X07-102

Schaedel, M.S.; Larson, A.J.; Affleck, D.L.R.; Belote, R.T.; Goodburn, J.M.; Page-Dumroese, D.S. 2017. Early forest thinning changes aboveground carbon distribution among pools, but not total amount. Forest Ecology and Management. 389(Supplement C): 187-198. doi:10.1016/j.foreco.2016.12.018

Schaedel, M.S.; Larson, A.J.; Affleck, D.L.R.; Belote, R.T.; Goodburn, J.M.; Wright, D.K.; Sutherland, E.K. 2017. Long-term precommercial thinning effects on Larix occidentalis (western larch) tree and stand characteristics. Canadian Journal of Forest Research. 47: 861-874. doi:10.1139/cjfr-2017-0074

Schaetzl, R.J. 1986. Complete soil profile inversion by tree uprooting. Physical Geography. 7(2): 181-189. doi:10.1080/02723646.1986.10642290

Schaetzl, R.J.; Burns, S.F.; Johnson, D.L.; Small, T.W. 1989. Tree uprooting: review of impacts on forest ecology. Vegetatio. 79(3): 165-176. doi:10.1007/BF00044908

Schaetzl, R.J.; Johnson, D.L.; Burns, S.F.; Small, T.W. 1989. Tree uprooting: review of terminology, process, and environmental implications. Canadian Journal of Forest Research. 19(1): 1-11. doi:10.1139/x89-001

Scharpf, R.F.; Parmeter, Jr., J.R., tech. coords. 1978. Proceedings of the symposium on dwarf mistletoe control through forest management. Berkeley, CA: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station. 190 p. https://www.fs.usda.gov/treesearch/pubs/24083

Scheller, R.M.; Mladenoff, D.J. 2007. An ecological classification of forest landscape simulation models: tools and strategies for understanding broad-scale forested ecosystems. Landscape Ecology. 22(4): 491-505. doi:10.1007/s10980-006-9048-4

Scheller, R.M.; Mladenoff, D.J.; Crow, T.R.; Sickley, T.A. 2005. Simulating the effects of fire reintroduction versus continued fire absence on forest composition and landscape structure in the Boundary Waters Canoe Area, northern Minnesota, USA. Ecosystems. 8(4): 396-411. doi:10.1007/s10021-003-0087-2

Schenk, J.A.; Mahoney, R.L.; Moore, J.A.; Adams, D.L. 1976. Understory plants as indicators of grand fir mortality due to the fir engraver. Journal of the Entomological Society of British Columbia. 73: 21-24. https://journal.entsocbc.ca/index.php/journal/article/view/1881

Schenk, J.A.; Moore, J.A.; Adams, D.L.; Mahoney, R.L. 1977. A preliminary hazard rating of grand fir stands for mortality by the fir engraver. Forest Science. 23(1): 103-110. doi:10.1093/forestscience/23.1.103

Schenk, J.A.; Mahoney, R.L.; Moore, J.A.; Adams, D.L. 1980. A model for hazard rating lodgepole pine stands for mortality by mountain pine beetle. Forest Ecology and Management. 3: 57-68. doi:10.1016/0378-1127(80)90005-5

Scher, J.S. 2002. *Larix occidentalis*. In: Fire Effects Information System [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us/database/feis/plants/tree/larlya/all.html

Schieck, J.; Lertzman, K.; Nyberg, B.; Page, R. 1995. Effects of patch size on birds in old-growth montane forests. Conservation Biology. 9(5): 1072-1084. doi:10.1046/j.1523-1739.1995.9051059.x-i1

Schmid, J.M.; Frye, R.H. 1976. Stand ratings for spruce beetles. Res. Note RM-309. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 4 p. https://digitalcommons.usu.edu/cgi/viewcontent.cgi?article=1128&context=barkbeetles

Schmid, J.M.; Frye, R.H. 1977. Spruce beetle in the Rockies. Gen. Tech. Rep. RM-49. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. 33 p. https://www.fs.usda.gov/treesearch/pubs/35507

Schmidt, W.C. 1969. Seedbed treatments influence seedling development in western larch forests. Res. Note INT-93. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 7 p. https://www.fs.fed.us/rm/pubs_exp_forests/coram/rmrs_1969_schmidt_w001.pdf

Schmidt, W.C., comp. 1988. Proceedings–future forests of the mountain west: a stand culture symposium. Gen. Tech. Rep. INT-243. Ogden, UT: USDA Forest Service, Intermountain Research Station. 402 p. https://archive.org/download/CAT88897341/CAT88897341.pdf

Schmidt, W.C.; McDonald, K.J., comps. 1995. Ecology and management of Larix forests: a look ahead; proceedings of an international symposium. Gen. Tech. Rep. GTR-INT-319. Ogden, UT: USDA Forest Service, Intermountain Research Station. 521 p. https://archive.org/download/CAT10699418/CAT10699418.pdf

Schmidt, W.C.; Shearer, R.C. 1990. *Larix occidentalis* Nutt.; western larch. In: Burns, R.M.; Honkala, B.H., tech. coords. Silvics of North America; volume 1, conifers. Agric. Handb. 654. Washington, DC: USDA Forest Service: 160-172. https://www.treesearch.fs.fed.us/pubs/1547 https://www.srs.fs.usda.gov/pubs/misc/ag_654/volume_1/larix/occidentalis.htm

Schmidt, W.C.; Shearer, R.C.; Roe, A.L. 1976. Ecology and silviculture of western larch forests. Tech. Bull. 1520. Washington, DC: USDA Forest Service. 96 p. https://www.fs.fed.us/rm/pubs_exp_forests/coram/rmrs_1976_schmidt_w001.pdf

Schmidt, T.; Boche, M.; Blackwood, J.; Richmond, B. 1993. Blue Mountains ecosystem restoration strategy: a report to the Regional Forester. Unnumbered Report. Portland, OR: USDA Forest Service, Pacific Northwest Region. 12 p (plus appendices).

Schmidt, K.M.; Menakis, J.P.; Hardy, C.C.; Hann, W.J.; Bunnell, D.L. 2002. Development of coarse-scale spatial data for wildland fire and fuel management. Gen. Tech. Rep. RMRS-GTR-87. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 41 p (and CD). http://www.treesearch.fs.fed.us/pubs/4590

Schmidt, L.; Hille, M.G.; Stephens, S.L. 2006. Restoring northern Sierra Nevada mixed conifer forest composition and structure with prescribed fires of varying intensities. Fire

Ecology. 2(2): 20-33. doi:10.4996/fireecology.0202020

Schmiegelow, F.K.A.; Stepnisky, D.P.; Stambaugh, C.A.; Koivula, M. 2006. Reconciling salvage logging of boreal forests with a natural-disturbance management model. Conservation Biology. 20(4): 971-983. doi:10.1111/j.1523-1739.2006.00496.x

Schmitt, C.L. 1996. Recommendations for managing the Douglas-fir and larch dwarf mistletoe components in stands with regard to wildife use. Unnum. Rep. La Grande, OR: USDA Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest, Blue Mountains Pest Management Service Center. 10 p.

Schmitt, C.L. 1997. Management of Douglas-fir infected with dwarf mistletoe in the Blue Mountains of northeastern Oregon and southeastern Washington. Management Guide BMZ-97-02. La Grande, OR: USDA Forest Service, Pacific Northwest Region, Blue Mountains Pest Management Service Center. 26 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_026324.pdf

Schmitt, C.L. 1999. Effects of stocking level control on the occurrence and severity of conifer diseases in the Blue Mountains of northeastern Oregon and southeastern Washington. BMZ-99-04. La Grande, OR: USDA Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest, Blue Mountains Pest Management Service Center. 9 p.

Schmitt, C.L. 2001. Root diseases in conifer forested communities in the Blue Mountains of northeastern Oregon and southeastern Washington–detection and management, values and impacts. Management Guide BMPMSC-02-01. [La Grande, OR]: USDA Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest, Blue Mountains Pest Management Service Center. 44 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_026030.pdf

Schmitt, C.L. 2010. Upland moist forests and associated disease disturbances in the Blue Mountains – cases for management action. La Grande, OR: USDA Forest Service, Wallowa-Whitman National Forest, Blue Mountains Pest Management Service Center. 29 p.

Schmitt, C.L.; Hadfield, J.S. 2009. Larch dwarf mistletoe – management strategies and general marking guide for eastern Washington and the Blue Mountains, and visual guide to determining dwarf mistletoe rating (DMR). BMPMSC-09-01. La Grande, OR: USDA Forest Service, Pacific Northwest Region, Blue Mountains Pest Management Service Center. 18 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_026107.pdf

Schmitt, C.L.; Powell, D.C. 2005. Rating forest stands for insect and disease susceptibility: a simplified approach; version 2.0. Pub. BMPMSC-05-01. La Grande, OR: USDA Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest, Blue Mountains Pest Management Service Center. 20 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015659.pdf

Schmitt, C.L.; Powell, D.C. 2012. Historical range of variability estimates for insect and disease susceptibility. White Paper F14-SO-WP-Silv-22. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 16 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5358588.pdf

Schmitt, C.L.; Scott, D.W. 1993. Catastrophic stand conditions in the Blue Mountains: discussion, guidelines, and rating system. BMZ-93-05. Baker City, OR: USDA Forest Service, Pacific Northwest Region, Blue Mountain Pest Management Zone, Wallowa-Whitman National Forest. 12 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_026435.pdf

Schmitt, C.L.; Parmeter, J.R.; Kliejunas, J.T. 2000. Annosus root disease of western conifers. Forest Insect & Disease Leaflet 172; R6-NR-FID-PR-002-00. [Place of publication unknown]: USDA Forest Service. 9 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043457.pdf

Schmitz, R.F.; McGregor, M.D.; Amman, G.D.; Oakes, R.D. 1989. Effect of partial cutting treatments of lodgepole pine stands on the abundance and behavior of flying mountain pine beetles. Canadian Journal of Forest Research. 19(5): 566-574. doi:10.1139/x89-089

Schneider, D.C. 2001. The rise of the concept of scale in ecology. BioScience. 51(7): 545-553. doi:10.1641/0006-3568(2001)051[0545:TROTCO]2.0.CO;2

Schneider, E.E.; Affleck, D.L.R.; Larson, A.J. 2019. Tree spatial patterns modulate peak snow accumulation and snow disappearance. Forest Ecology and Management. 441: 9-19. doi:10.1016/j.foreco.2019.03.031

Schnepf, C.; Graham, R.T.; Kegley, S.; Jain, T.B. 2009. Managing organic debris for forest health: reconciling fire hazard, bark beetles, wildlife, and forest nutrition needs. PNW 609. Moscow, ID: University of Idaho; Oregon State University; Washington State University. 60 p. https://www.fs.usda.gov/treesearch/pubs/33615

Schoennagel, T.; Nelson, C.R. 2011. Restoration relevance of recent National Fire Plan treatments in forests of the western United States. Frontiers in Ecology and the Environment. 9(5): 271-277. doi:10.1890/090199

Schoennagel, T.; Veblen, T.T.; Romme, W.H. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience. 54(7): 661-676. doi:10.1641/0006-3568(2004)054[0661:TIOFFA]2.0.CO;2

Schoennagel, T.; Veblen, T.T.; Negron, J.F.; Smith, J.M. 2012. Effects of mountain pine beetle on fuels and expected fire behavior in lodgepole pine forests, Colorado, USA. PLoS ONE. 7(1): e30002. doi:10.1371/journal.pone.0030002

Schowalter, T.D. 1981. Insect herbivore relationship to the state of the host plant: biotic regulation of ecosystem nutrient cycling through ecological succession. Oikos. 37(1): 126-130. doi:10.2307/3544083

Schowalter, T.D. 1988. Forest pest management: a synopsis. Northwest Environmental Journal 4: 313-318.

Schowalter, T.D. 2008. Insect herbivore responses to management practices in conifer forests in North America. Journal of Sustainable Forestry. 26(3): 204-222. doi:10.1080/10549810701879727

Schowalter, T. 2017. Arthropod diversity and functional importance in old-growth forests of North America. Forests. 8(4): 97 (17 p). doi:10.3390/f8040097

Schowalter, T.D.; Means, J.E. 1988. Pest response to simplification of forest landscapes. Northwest Environmental Journal. 4(2): 342-343. http://andrewsforest.oregonstate.edu/pubs/pdf/pub1000.pdf

Schowalter, T.D.; Withgott, J. 2001. Rethinking insects: What would an ecosystem approach look like? Conservation Biology In Practice. 2(4): 10-16. doi:10.1111/j.1526-4629.2001.tb00017.x

Schowalter, T.D.; Hargrove, W.W.; Crossley, D.A., Jr. 1986. Herbivory in forested ecosystems. Annual Review of Entomology. 31: 177-196. doi:10.1146/annurev.en.31.010186.001141

Schowalter, T.; Hansen, E.; Molina, R.; Zhang, Y. 1997. Integrating the ecological roles of phytophagous insects, plant pathogens, and mycorrhizae in managed forests. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press: 171-189. isbn:1-55963-399-9

Schreuder, H.T.; Thomas, C.E. 1991. Establishing cause-effect relationships using forest survey data. Forest Science. 37(6): 1497-1512. doi:10.1093/forestscience/37.6.1497

Schroeder, T.A.; Healey, S.P.; Moisen, G.G.; Frescino, T.S.; Cohen, W.B.; Huang, C.; Kennedy, R.E.; Yang, Z. 2014. Improving estimates of forest disturbance by combining observations from Landsat time series with U.S. Forest Service Forest Inventory and Analysis data. Remote Sensing of Environment. 154: 61-73. doi:10.1016/j.rse.2014.08.005

Schuler, L.J.; Bugmann, H.; Snell, R.S. 2017. From monocultures to mixed-species forests: is tree diversity key for providing ecosystem services at the landscape scale? Landscape Ecology. 32(7): 1499-1516. doi:10.1007/s10980-016-0422-6

Schulte, L.A.; Mladenoff, D.J. 2005. Severe wind and fire regimes in northern forests: historical variability at the regional scale. Ecology. 86(2): 431-445. doi:10.1890/03-4065

Schulze, E.-D.; Beck, E.; Müller-Hohenstein, K. 2005. Plant ecology. Heidelberg, Germany: Springer-Verlag. 702 p. isbn:978-3-540-20833-4

Schütt, P.; Cowling, E.B. 1985. Waldsterben, a general decline of forests in central Europe: symptoms, development, and possible causes. Plant Disease. 69(7): 548-558. doi:10.1094/PD-69-548

Schwandt, J.W. 1984. Dwarf mistletoe management strategies for inland Douglas-fir and grand fir forest types. In: Baumgartner, D.M.; Mitchell, R., comps. Silvicultural management strategies for pests of the interior Douglas-fir and grand fir forest types. Pullman, WA: Washington State University, Cooperative Extension: 67-71.

Schwandt, J.W.; Lockman, I.B.; Kliejunas, J.T.; Muir, J.A. 2010. Current health issues and management strategies for white pines in the western United States and Canada. Forest Pathology. 40(3-4): 226-250. doi:10.1111/j.1439-0329.2010.00656.x

Schwilk, D.W.; Ackerly, D.D. 2001. Flammability and serotiny as strategies: correlated evolution in pines. Oikos. 94(2): 326-336. doi:10.1034/j.1600-0706.2001.940213.x

Schwilk, D.W.; Knapp, E.E.; Ferrenberg, S.M.; Keeley, J.E.; Caprio, A.C. 2006. Tree mortality from fire and bark beetles following early and late season prescribed fires in a Sierra Nevada mixed-conifer forest. Forest Ecology and Management. 232(1-3): 36-45. doi:10.1016/j.foreco.2006.05.036

Schwilk, D.W.; Keeley, J.E.; Knapp, E.E.; McIver, J.; Bailey, J.D.; Fettig, C.J.; Fiedler, C.E.; Harrod, R.J.; Moghaddas, J.J.; Outcalt, K.W.; Skinner, C.N.; Stephens, S.L.; Waldrop, T.A.; Yaussy, D.A.; Youngblood, A. 2009. The national Fire and Fire Surrogate study: effects of fuel reduction methods on forest vegetation structure and fuels. Ecological Applications. 19(2): 285-304. doi:10.1890/07-1747.1

Scott, D.W. 1996. A rationale and procedure for determining imminent susceptibility of stands to insects in the Blue and Wallowa Mountains of southeastern Washington and northeastern Oregon. BMZ-96-15. Baker City, OR: USDA Forest Service, Pacific Northwest Region, Blue Mountain Pest Management Zone, Wallowa-Whitman National Forest. 17 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_025923.pdf

Scott, D.W. 2010. Upland moist forests and associated insect disturbances in the Blue Mountains: supporting science for management action. Rep. BMPMSC-10-01. La Grande, OR: USDA Forest Service, Wallowa-Whitman National Forest, Blue Mountains Pest Management Service Center. 101 p.

Scott, R.E.; Mitchell, S.J. 2005. Empirical modelling of windthrow risk in partially harvested stands using tree, neighbourhood, and stand attributes. Forest Ecology and Management. 218(1-3): 193-209. doi:10.1016/j.foreco.2005.07.012

Scott, D.W.; Schmitt, C.L.; Spiegel, L.H. 2002. Factors affecting survival of fire injured trees: a rating system for determining relative probability of survival of conifers in the Blue and Wallowa mountains. BMPMSC-03-01. La Grande, OR: USDA Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest, Blue Mountains Pest Management Service Center. 86 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_025713.pdf

Seager, S.T.; Eisenberg, C.; St. Clair, S.B. 2013. Patterns and consequences of ungulate herbivory on aspen in western North America. Forest Ecology and Management. 299: 81-90. doi:10.1016/j.foreco.2013.02.017

Secretariat of the Convention on Biological Diversity. 2001a. Assessment, conservation and sustainable use of forest biodiversity. Montreal, QB: Secretariat of the Convention on Biological Diversity. 120 p. http://www.cbd.int/doc/publications/cbd-ts-03.pdf

Secretariat of the Convention on Biological Diversity. 2001b. The value of forest ecosystems. Montreal, QB: Secretariat of the Convention on Biological Diversity. 60 p. http://www.cbd.int/doc/publications/cbd-ts-04.pdf

Secretariat of the Convention on Biological Diversity. 2009. Sustainable forest management, biodiversity and livelihoods: A good practice guide. Montreal, QB: Secretariat of the Convention on Biological Diversity. 47 p. https://portals.iucn.org/library/sites/library/files/documents/2009-093.pdf

Sedell, J.R.; Leone, F.N.; Duval, W.S. 1991. Water transportation and storage of logs. In: Meehan, W.R., ed. Influences of forest and rangeland management on salmonid fishes and their habitats. Special Publication 19. Bethesda, MD: American Fisheries Society: 325-368. http://andrewsforest.oregonstate.edu/pubs/pdf/pub3177.pdf

Seidel, K.W. 1971. Growth of young even-aged western larch stands after thinning in eastern Oregon. Res. Note PNW-165. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 12 p. Growth_of_young_even_aged_western_larch

Seidel, K.W. 1975. Response of western larch to changes in stand density and structure. Res. Note PNW-258. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 11 p. https://archive.org/download/responseofwester258seid/responseofwester258seid.pdf

Seidel, K.W. 1977. Levels-of-growing-stock study in thinned western larch pole stands in eastern Oregon. Res. Pap. PNW-221. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 14 p. Levels_of_growing_stock_Study_in_Thinned

Seidel, K.W. 1979a. Regeneration in mixed conifer clearcuts in the Cascade Range and Blue Mountains of eastern Oregon. Res. Pap. PNW-248. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 24 p. https://www.biodiversitylibrary.org/itempdf/172179

Seidel, K.W. 1979b. Regeneration in mixed conifer shelterwood cuttings in the Cascade Range of eastern Oregon. Res. Pap. PNW-264. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 29 p. https://www.fs.usda.gov/treesearch/pubs/26865

Seidel, K.W. 1980. Diameter and height growth of suppressed grand fir saplings after overstory removal. Res. Pap. PNW-275. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 9 p. https://www.fs.usda.gov/treesearch/pubs/21255

Seidel, K.W. 1980. Growth of western larch after thinning from above and below to several density levels: 10-year results. Res. Note PNW-366. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 20 p. https://www.fs.usda.gov/treesearch/pubs/7632

Seidel, K.W. 1982. Growth and yield of western larch: 15-year results of a levels-of-growing-stock study. Res. Note PNW-398. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 14 p. https://www.fs.usda.gov/treesearch/pubs/26763

Seidel, K.W. 1984. A western larch-Engelmann spruce spacing study in eastern Oregon: results after 10 years. Res. Pap. PNW-409. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 6 p. https://www.fs.usda.gov/treesearch/pubs/26776

Seidel, K.W. 1986. Growth and yield of western larch in response to several density levels and two thinning methods: 15-year results. Res. Note PNW-RN-455. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 18 p.

http://www.treesearch.fs.fed.us/pubs/21227

Seidel, K.W. 1986. Tolerance of seedlings of ponderosa pine, Douglas-fir, grand fir, and Engelmann spruce for high temperatures. Northwest Science. 60(1): 1-7. http://hdl.handle.net/2376/1778

Seidel, K.W. 1987. Results after 20 years from a western larch levels-of-growing-stock study. Gen. Tech. Rep. PNW-RP-387. Portland, OR: USDA Forest Service, Pacific Northwest Region. 18 p. https://www.fs.usda.gov/treesearch/pubs/9223

Seidel, K.W.; Cochran, P.H. 1981. Silviculture of mixed conifer forests in eastern Oregon and Washington. Gen. Tech. Rep. PNW-121. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 70 p. https://www.fs.usda.gov/treesearch/pubs/7516

Seidel, K.W.; Head, S.C. 1983. Regeneration in mixed conifer partial cuttings in the Blue Mountains of Oregon and Washington. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 14 p. https://www.fs.usda.gov/treesearch/pubs/25190

Seidel, K.W.; Geist, J.M.; Strickler, G.S. 1990. The influence of cattle grazing and grass seeding on coniferous regeneration after shelterwood cutting in eastern Oregon. Res. Pap. PNW-RP-417. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 32 p. http://www.treesearch.fs.fed.us/pubs/9252

Seidl, R.; Lexer, M.J. 2013. Forest management under climatic and social uncertainty: Trade-offs between reducing climate change impacts and fostering adaptive capacity. Journal of Environmental Management. 114: 461-469. doi:10.1016/j.jenvman.2012.09.028

Seidl, R.; Fernandes, P.M.; Fonseca, T.F.; Gillet, F.; Jönsson, A.M.; Merganicová, K.; Netherer, S.; Arpaci, A.; Bontemps, J.D.; Bugmann, H.; González-Olabarria, J.R.; Lasch, P.; Meredieu, C.; Moreira, F.; Schelhaas, M.-J.; Mohren, F. 2011. Modelling natural disturbances in forest ecosystems: a review. Ecological Modelling. 222(4): 903-924. doi:10.1016/j.ecolmodel.2010.09.040

Seidl, R.; Rammer, W.; Scheller, R.M.; Spies, T.A. 2012. An individual-based process model to simulate landscape-scale forest ecosystem dynamics. Ecological Modelling. 231: 87-100. doi:10.1016/j.ecolmodel.2012.02.015

Seidl, R.; Rammer, W.; Blennow, K. 2014. Simulating wind disturbance impacts on forest landscapes: Tree-level heterogeneity matters. Environmental Modelling & Software. 51: 1-11. doi:10.1016/j.envsoft.2013.09.018

Seidl, R.; Spies, T.A.; Peterson, D.L.; Stephens, S.L.; Hicke, J.A. 2016a. Searching for resilience: Addressing the impacts of changing disturbance regimes on forest ecosystem services. Journal of Applied Ecology. 53(1): 120-129. doi:10.1111/1365-2664.12511

Seidl, R.; Donato, D.C.; Raffa, K.F.; Turner, M.G. 2016b. Spatial variability in tree regeneration after wildfire delays and dampens future bark beetle outbreaks. Proceedings of the National Academy of Sciences. 113(46): 13075-13080. doi:10.1073/pnas.1615263113

Seidl, R.; Thom, D.; Kautz, M.; Martin-Benito, D.; Peltoniemi, M.; Vacchiano, G.; Wild, J.; Ascoli, D.; Petr, M.; Honkaniemi, J.; Lexer, M.J.; Trotsiuk, V.; Mairota, P.; Svoboda, M.; Fabrika, M.; Nagel, T.A.; Reyer, C.P.O. 2017. Forest disturbances under climate change. Nature Climate Change. 7: 395-402. doi:10.1038/nclimate3303

Seidl, R.; Albrich, K.; Thom, D.; Rammer, W. 2018. Harnessing landscape heterogeneity for managing future disturbance risks in forest ecosystems. Journal of Environmental Management. 209: 46-56. doi:10.1016/j.jenvman.2017.12.014

Seidl, R.; Albrich, K.; Erb, K.; Formayer, H.; Leidinger, D.; Leitinger, G.; Tappeiner, U.; Tasser, E.; Rammer, W. 2019. What drives the future supply of regulating ecosystem services in a mountain forest landscape? Forest Ecology and Management. 445: 37-47. doi:10.1016/j.foreco.2019.03.047

Sensenig, T.; Bailey, J.D.; Tappeiner, J.C. 2013. Stand development, fire and growth of old-growth and young forests in southwestern Oregon, USA. Forest Ecology and Management. 291: 96-109. doi:10.1016/j.foreco.2012.11.006

Senyk, J.; Craigdallie, D. 1997. Effects of harvesting methods on soil properties and forest productivity in interior British Columbia. Information Rep. BC-X-365. Victoria, BC: British Columbia, Ministry of Forests. 37 p. https://cfs.nrcan.gc.ca/publications/download-pdf/4774

SERI (Society for Ecological Restoration International). 2004. The SER International primer on ecological restoration; version 2. Tucson, AZ: Society for Ecological Restoration International, Science and Policy Working Group. 13 p. http://www.ser.org/pdf/primer3.pdf

Service, R.F. 2004. As the west goes dry. Science. 303(5661): 1124-1127. doi:10.1126/science.303.5661.1124

Sesnie, S.E.; Dickson, B.G.; Rundall, J.M.; Sisk, T.D. 2009. Preliminary stratification and characterization of the mixed conifer forest type on the Kaibab National Forest, North Kaibab Ranger District. Unpublished final report. Flagstaff, AZ: Northern Arizona University, School of Earth Sciences and Environmental Sustainability. 21 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5120029.pdf

Shafer, S.L.; Bartlein, P.J.; Thompson, R.S. 2001. Potential changes in the distributions of western North America tree and shrub taxa under future climate scenarios. Ecosystems. 4(3): 200-215. doi:10.1007/s10021-001-0004-5

Shakesby, R.A.; Doerr, S.H. 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews. 74(3): 269-307. doi:10.1016/j.earscirev.2005.10.006

Shamoun, S.F.; Ramsfield, T.D.; van der Kamp, B.J. 2003. Biological control approach for management of dwarf mistletoes. New Zealand Journal of Forestry Science. 33(3): 373-384. http://www.scionresearch.com/__data/assets/pdf_file/0015/5334/07_SHAMOUN.pdf

Shapiro-Miller, L.B.; Heyerdahl, E.K.; Morgan, P. 2007. Comparison of fire scars, fire atlases, and satellite data in the northwestern United States. Canadian Journal of Forest Research. 37(10): 1933-1943. doi:10.1139/X07-054

Sharik, T.L.; Adair, W.; Baker, F.A.; Battaglia, M.; Comfort, E.J.; D'Amato, A.W.; Delong, C.; DeRose, R.J.; Ducey, M.J.; Harmon, M.; Levy, L.; Logan, J.A.; O'Brien, J.; Palik, B.J.; Roberts, S.D.; Rogers, P.C.; Shinneman, D.J.; Spies, T.; Taylor, S.L.; Woodall, C.; Youngblood, A. 2010. Emerging themes in the ecology and management of North American forests. International Journal of Forestry Research. 2010: 964260 (11 p). doi:10.1155/2010/964260

Shaw, C.G., III; Kile, G.A. 1991. Armillaria root disease. Agric. Handb. No. 691. Washington, DC: USDA Forest Service. 233 p.

Shaw, T.M.; Moore, J.A.; Marshall, J.D. 1998. Root chemistry of Douglas-fir seedlings grown under different nitrogen and potassium regimes. Canadian Journal of Forest Research. 28(10): 1566-1573. doi:10.1139/x98-136

Shaw, D.C.; Oester, P.T.; Filip, G.M. 2009. Managing insects and diseases of Oregon conifers. EM 8980. [Corvallis, OR]: Oregon State University, Extension Service. 98 p. https://catalog.extension.oregonstate.edu/em8980

Shaw, M.R.; Pendleton, L.; Cameron, D.R.; Morris, B.; Bachelet, D.; Klausmeyer, K.; MacKenzie, J.; Conklin, D.R.; Bratman, G.N.; Lenihan, J.; Haunreiter, E.; Daly, C.; Roehrdanz, P.R. 2011. The impact of climate change on California's ecosystem services. Climatic Change. 109(Supplement 1): 465-484. doi:10.1007/s10584-011-0313-4

Shea, K.L. 1985. Demographic aspects of coexistence in Engelmann spruce and subalpine fir. American Journal of Botany. 72(11): 1823-1833. doi:10.1002/j.1537-2197.1985.tb08454.x

Shearer, R.C. 1976. Early establishment of conifers following prescribed broadcast burning in western larch/Douglas-fir forests. In: Proceedings of the annual Tall Timbers fire ecology conference no. 14. Tallahassee, FL: Tall Timbers Research Station: 481-500. http://talltimbers.org/wp-content/uploads/2014/03/Shearer1974_op.pdf

Shearer, R.C. 1980. Regeneration establishment in response to harvesting and residue management in a western larch–Douglas-fir forest. In: USDA Forest Service. Environmental consequences of timber harvesting in Rocky Mountain coniferous forests. Gen. Tech. Rep. INT-90. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 249-269. https://archive.org/download/CAT80751102/CAT80751102.pdf

Shearer, R.C.; Schmidt, W.C. 1987. Cone production and stand density in young Larix occidentalis. Forest Ecology and Management. 19(1-4): 219-226. doi:10.1016/0378-1127(87)90030-2

Shearer, R.C.; Schmidt, J.A. 1999. Natural regeneration after harvest and residue treatment in a mixed conifer forest of northwestern Montana. Canadian Journal of Forest Research. 29(2): 274-279. doi:10.1139/x99-006

Sheehan, K.A. 1996a. Defoliation by western spruce budworm in Oregon and Washington from 1980 through 1994. Tech. Pub. R6-NR-TP-04-96. Portland, OR: USDA Forest Service, Pacific Northwest Region. Irregular pagination. Defoliation_by_Western_Spruce_Budworm

Sheehan, K.A. 1996b. Effects of insecticide treatments on subsequent defoliation by western spruce budworm in Oregon and Washington: 1982-1992. Gen. Tech. Rep. PNW-GTR-367. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 54 p. https://www.fs.usda.gov/treesearch/pubs/5598

Shelby, B.; Thompson, J.; Brunson, M.; Johnson, R. 2003. Changes in scenic quality after harvest: a decade of ratings for six silviculture treatments. Journal of Forestry. 101(2): 30-35. doi:10.1093/jof/101.2.30

Shen, G.; Moore, J.A.; Hatch, C.R. 2000. The effect of habitat type and rock type on individual tree basal area growth response to nitrogen fertilization. Canadian Journal of Forest Research. 30(4): 613-623. doi:10.1139/x99-249

Shen, G.; Moore, J.A.; Hatch, C.R. 2001. The effect of nitrogen fertilization, rock type, and habitat type on individual tree mortality. Forest Science. 47(2): 203-213. doi:10.1093/forestscience/47.2.203

Shepherd, R.F. 1994. Management strategies for forest insect defoliators in British Columbia. Forest Ecology and Management. 68(2-3): 303-324. doi:10.1016/0378-1127(94)90053-1

Sheppard, S.; Picard, P. 2006. Visual-quality impacts of forest pest activity at the landscape level: a synthesis of published knowledge and research needs. Landscape and Urban Planning. 77(4): 321-342.

Shepperd, W.D.; Binkley, D.; Bartos, D.L.; Stohlgren, T.J.; Eskew, L.G., comps. 2001. Sustaining aspen in western landscapes: symposium proceedings. Proceedings RMRS-P-18. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station. 460 p.

Sherriff, R.L.; Veblen, T.T. 2006. Ecological effects of changes in fire regimes in *Pinus ponderosa* ecosystems in the Colorado Front Range. Journal of Vegetation Science. 17(6): 705-718. doi:10.1111/j.1654-1103.2006.tb02494.x

Shiflet, T.N., ed. 1994. Rangeland cover types of the United States. Denver, CO: Society for Range Management. 152 p. isbn:1-884930-01-8

Shifley, S.R. 2006. Sustainable forestry in the balance. Journal of Forestry. 104(4): 187-195.

Shigo, A.L. 1979. Tree decay: an expanded concept. Agric. Infor. Bull. Num. 419. Washington, DC: USDA Forest Service. 73 p.

Shindler, B. 2000. Landscape-level management: It's all about context. Journal of Forestry.

98(12): 10-14. doi:10.1093/jof/98.12.10

Shindler, B.; Cramer, L.A. 1999. Shifting public values for forest management: making sense of wicked problems. Western Journal of Applied Forestry. 14(1): 28-34. doi:10.1093/wjaf/14.1.28

Shinn, D.A. 1980. Historical perspectives on range burning in the inland Pacific Northwest. Journal of Range Management. 33(6): 415-423. https://journals.uair.arizona.edu/index.php/jrm/article/download/7102/6714

Shinneman, D.J.; Baker, W.L.; Rogers, P.C.; Kulakowski, D. 2013. Fire regimes of quaking aspen in the Mountain West. Forest Ecology and Management. 299: 22-34. doi:10.1016/j.foreco.2012.11.032

Shirley, D.M.; Erickson, V. 2001. Aspen restoration in the Blue Mountains of northeast Oregon. In: Shepperd, W.D.; Binkley, D.; Bartos, D.L.; Stohlgren, T.J.; Eskew, L.C., comps. Sustaining aspen in western landscapes. Proceedings RMRS-P-18. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station: 101-115. https://www.fs.usda.gov/treesearch/pubs/35810

Shive, K.L.; Preisler, H.K.; Welch, K.R.; Safford, H.D.; Butz, R.J.; O'Hara, K.L.; Stephens, S.L. 2018. From the stand scale to the landscape scale: predicting the spatial patterns of forest regeneration after disturbance. Ecological Applications. 28(6): 1626-1639. doi:10.1002/eap.1756

Shlisky, A.J. 1994. Multiscale analysis in the Pacific Northwest. Journal of Forestry. 92(8): 32-34. doi:10.1093/jof/92.8.32

Shore, T.L.; Safranyik, L. 1992. Susceptibility and risk rating systems for the mountain pine beetle in lodgepole pine stands. Infor. Rep. BC-X-336. Victoria, BC: Forestry Canada, Pacific and Yukon Region, Pacific Forestry Centre. 12 p. http://cfs.nrcan.gc.ca/pubwarehouse/pdfs/3155.pdf

Shore, T.L.; Safranyik, L.; Riel, W.G.; Ferguson, M.; Castonguay, J. 1999. Evaluation of factors affecting tree and stand susceptibility to the Douglas-fir beetle (Coleoptera:Scolytidae). Canadian Entomologist. 131(6): 831-839. doi:10.4039/Ent131831-6

Shore, T.L.; Brooks, J.E.; Stone, J.E., eds. 2004. Mountain pine beetle symposium: challenges and solutions. Infor. Rep. BC-X-399. Victoria, BC: Natural Resources Canada, Canadian Forest Service, Pacific Forestry Centre. 287 p. https://cfs.nrcan.gc.ca/pubwarehouse/pdfs/25159.pdf

Shrimpton, D.M. 1978. Resistance of lodgepole pine to mountain pine beetle infestation. In: Berryman, A.A.; Amman, G.D.; Stark, R.W., eds. Theory and practice of mountain pine beetle management in lodgepole pine forest. Moscow, ID: University of Idaho, Forest, Wildlife and Range Experiment Station: 64-76.

Shrimpton, D.M.; Thomson, A.J. 1983. Growth characteristics of lodgepole pine associated with the start of mountain pine beetle outbreaks. Canadian Journal of Forest Research. 13(1): 137-144. doi:10.1139/x83-020

Shugart, H.H. 1987. Dynamic ecosystem consequences of tree birth and death patterns. BioScience. 37(8): 596-602. doi:10.2307/1310670

Shugart, H.H., Jr.; West, D.C. 1981. Long-term dynamics of forest ecosystems. American Scientist. 69(6): 647-652. https://www.jstor.org/stable/27850716

Siegel, R.B.; DeSante, D.F. 2003. Bird communities in thinned versus unthinned Sierran mixed conifer stands. Wilson Bulletin. 115(2): 155-165. doi:10.1676/02-103

Sillett, S.C.; Van Pelt, R.; Freund, J.A.; Campbell-Spickler, J.; Carroll, A.L.; Kramer, R.D. 2018. Development and dominance of Douglas-fir in North American rainforests. Forest Ecology and Management. 429: 93-114. doi:10.1016/j.foreco.2018.07.006

Silva, V.; Pereira, J.L.; Campos, I.; Keizer, J.J.; Gonçalves, F.; Abrantes, N. 2015. Toxicity assessment of aqueous extracts of ash from forest fires. Catena. 135: 401-408. doi:10.1016/j.catena.2014.06.021

Simard, S.W.; Hagerman, S.M.; Sachs, D.L.; Heineman, J.L.; Mather, W.J. 2005. Conifer growth, *Armillaria ostoyae* root disease, and plant diversity responses to broadleaf competition reduction in mixed forests of southern interior British Columbia. Canadian Journal of Forest Research. 35(4): 843-859. doi:10.1139/x05-010

Simard, M.; Romme, W.H.; Griffin, J.M.; Turner, M.G. 2011. Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? Ecological Monographs. 81(1): 3-24. doi:10.1890/10-1176.1

Simberloff, D. 1998. Flagships, umbrellas, and keystones: Is single-species management passé in the landscape era? Biological Conservation. 83(3): 247-257. doi:10.1016/S0006-3207(97)00081-5

Simberloff, D. 1999. The role of science in the preservation of forest biodiversity. Forest Ecology and Management. 115(2-3): 101-111. doi:10.1016/S0378-1127(98)00391-0

Simberloff, D. 2000. Measuring diversity of communities and ecosystems with special reference to forests. In: Bunnell, F.L.; Johnson, J.F., eds. Policy and practices for biodiversity in managed forests: the living dance. Vancouver, BC: UBC Press: 66-81. isbn:0-7748-0691-5

Simon, N.P.P.; Stratton, C.B.; Forbes, G.J.; Schwab, F.E. 2002a. Similarity of small mammal abundance in post-fire and clearcut forests. Forest Ecology and Management. 165(1-3): 163-172. doi:10.1016/S0378-1127(01)00613-2

Simon, N.P.P.; Schwab, F.E.; Otto, R.D. 2002b. Songbird abundance in clear-cut and burned stands: a comparison of natural disturbance and forest management. Canadian Journal of Forest Research. 32(8): 1343-1350. doi:10.1139/x02-057

Simonin, K.A. 2000. Vaccinium membranaceum. In: Fire Effects Information System (online). Missoula, MT: USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (producer).

Simpson, D.G. 2000. Water use of interior Douglas-fir. Canadian Journal of Forest Research. 30(4): 534-547. doi:10.1139/x99-233

Sinclair, A.R.E.; Hik, D.S.; Schmitz, O.J.; Scudder, G.G.E.; Turpin, D.H.; Larter, N.C. 1995. Biodiversity and the need for habitat renewal. Ecological Applications. 5(3): 579-587. doi:10.2307/1941968

Six, D.L.; Skov, K. 2009. Response of bark beetles and their natural enemies to fire and fire surrogate treatments in mixed-conifer forests in western Montana. Forest Ecology and Management. 258(5): 761-772. doi:10.1016/j.foreco.2009.05.016

Six, D.; Biber, E.; Long, E. 2014. Management for mountain pine beetle outbreak suppression: Does relevant science support current policy? Forests. 5(1): 103-133. doi:10.3390/f5010103

Skinner, C.N. 2007. Silviculture and forest management under a rapidly changing climate. In: Powers, R.F., tech. ed. Restoring fire-adapted ecosystems: proceedings of the 2005 National Silviculture Workshop. Gen. Tech. Rep. PSW-GTR-203. Albany, CA: USDA Forest Service, Pacific Southwest Research Station: 21-32. http://www.treesearch.fs.fed.us/pubs/25889

Skog, K.E.; Barbour, R.J. 2006. Estimating woody biomass supply from thinning treatments to reduce fire hazard in the U.S. West. In: Andrews, P.L.; Butler, B.W., comps. Fuels management–how to measure success: Conference proceedings. Proceedings RMRS-P-41. Fort Collins, CO: USDA Forest Service, Rocky Mountain Research Station: 657-672. https://www.fs.usda.gov/treesearch/pubs/25986

Skog, K.E.; Nicholson, G.A. 1998. Carbon cycling through wood products: the role of wood and paper products in carbon sequestration. Forest Products Journal. 48(7/8): 75-83. https://www.fpl.fs.fed.us/documnts/pdf1998/skog98a.pdf

Skog, K.E.; Barbour, R.J.; Abt, K.L.; Bilek, E.M.; Burch, F.; Fight, R.D.; Hugget, R.J.; Miles, P.D.; Reinhardt, E.D.; Sheppard, W.D. 2006. Evaluation of silvicultural treatments and biomass use for reducing fire hazard in western states. Res. Pap. FPL-RP-634. Madison, WI: USDA Forest Service, Forest Products Laboratory. 29 p. https://www.fs.usda.gov/treesearch/pubs/24554

Skovlin, J.M.; Thomas, J.W. 1995. Interpreting long-term trends in Blue Mountain ecosystems from repeat photography. Res. Pap. PNW-GTR-315. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 102 p. http://www.treesearch.fs.fed.us/pubs/25667

Skovlin, J.M.; Strickler, G.S.; Peterson, J.L.; Sampson, A.W. 2001. Interpreting landscape change in high mountains of northeastern Oregon from long-term repeat photography. Gen. Tech. Rep. PNW-GTR-505. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 78 p. http://www.treesearch.fs.fed.us/pubs/3079

Sloan, J.P. 1998. Interruption of the natural fire cycle in a grand fir forest of central Idaho: changes in stand structure and composition. In: Proceedings of the annual Tall Timbers fire ecology conference no. 20. Tallahassee, FL: Tall Timbers Research Station: 250-257. https://talltimbers.org/wp-content/uploads/2018/09/250-Sloan1998_op.pdf

Smith, R.E.K. 1912. Report of the present condition of insect infestation on the Whitman National Forest, Oregon. Unpublished typescript report obtained from the National Archives, College Park, MD; record group 95. 13 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5414279.pdf

Smith, G.W. 1982. Habitat use by porcupines in a ponderosa pine/Douglas-fir forest in northeastern Oregon. Northwest Science. 56(3): 236-240.

Smith, T.M.; Smith, R.L. 2006. Elements of ecology, 6th edition. San Francisco, CA: Pearson Education, Inc. 658 p. isbn:0-8053-4830-1

Smith, K.; Weitknecht, R.H. 1915. Windfall damage in selection cuttings in Oregon. Proceedings of the Society of American Foresters. 10: 263-265. https://archive.org/download/vol10proceedings00soci/vol10proceedings00soci.pdf

Smith, D.M.; Larson, B.C.; Kelty, M.J.; Ashton, P.M.S. 1997. The practice of silviculture: applied forest ecology. 9th edition. New York: John Wiley & Sons. 537 p. isbn:0-471-10941-X

Smith, J.E.; McKay, D.; Brenner, G.; McIver, J.; Spatafora, J.W. 2005. Early impacts of forest restoration treatments on the ectomycorrhizal fungal community and fine root biomass in a mixed conifer forest. Journal of Applied Ecology. 42(3): 526-535. doi:10.1111/j.1365-2664.2005.01047.x

Smith, T.F.; Rizzo, D.M.; North, M. 2005b. Patterns of mortality in an old-growth mixed-conifer forest of the southern Sierra Nevada, California. Forest Science. 51(3): 266-275. doi:10.1093/forestscience/51.3.266

Smith, R.J.; Gray, A.N.; Swanson, M.E. 2020. Peak plant diversity during early forest development in the western United States. Forest Ecology and Management. 475: 118410 (11 p). doi:10.1016/j.foreco.2020.118410

Smith-McKenna, E.K.; Malanson, G.P.; Resler, L.M.; Carstensen, L.W.; Prisley, S.P.; Tomback, D.F. 2014. Cascading effects of feedbacks, disease, and climate change on alpine treeline dynamics. Environmental Modelling & Software. 62: 85-96. doi:10.1016/j.envsoft.2014.08.019

Smithers, L.A. 1961. Lodgepole pine in Alberta. Bulletin 127. Ottawa, Ontario, Canada: Canada, Department of Forestry. 153 p. http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.843.6862&rep=rep1&type=pdf

Snider, G.; Daugherty, P.J.; Wood, D. 2006. The irrationality of continued fire suppression:

An avoided cost analysis of fire hazard reduction treatments versus no treatment. Journal of Forestry. 104(8): 431-437. doi:10.1093/jof/104.8.431

Sniezko, R.; Smith, J.; Liu, J.-J.; Hamelin, R. 2014. Genetic resistance to fusiform rust in southern pines and white pine blister rust in white pines—a contrasting tale of two rust pathosystems—current status and future prospects. Forests. 5(9): 2050-2083. doi:10.3390/f5092050

Society for Ecological Restoration International. 2004. The SER international primer on ecological restoration. Tucson, AZ: Society for Ecological Restoration International. 13 p. http://www.ser.org/pdf/primer3.pdf

Sommers, W.T.; Coloff, S.G.; Conard, S.G. 2011. Synthesis of knowledge: fire history and climate change. Final report to the Joint Fire Science Program. Fairfax, VA: George Mason University. 215 p. plus 6 appendices. Deliverable_Gov_Pub_Synthesis

Song, S.J., ed. 2002. Ecological basis for stand management: A synthesis of ecological responses to wildfire and harvesting. Vegreville, AB: Alberta Research Council, Inc. ecological_basis_for_stand_management

Souto, X.C.; Chiapusio, G.; Pellissier, F. 2000. Relationships between phenolics and soil microorganisms in spruce forests: significance for natural regeneration. Journal of Chemical Ecology. 26(9): 2025-2034. doi:10.1023/A:1005504029243

Spies, T. 1997. Forest stand structure, composition, and function. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press: 11-30. isbn:1-55963-399-9

Spies, T.A.; Duncan, S.L. 2009. Old growth in a new world: a Pacific Northwest icon reexamined. Washington, DC: Island Press. 344 p. isbn:978-1-59726-410-5

Spies, T.A.; Hemstrom, M.A.; Youngblood, A.; Hummel, S. 2006. Conserving old-growth forest diversity in disturbance-prone landscapes. Conservation Biology. 20(2): 351-362. doi:10.1111/j.1523-1739.2006.00389.x

Spies, T.A.; Giesen, T.W.; Swanson, F.J.; Franklin, J.F.; Lach, D.; Johnson, K.N. 2010. Climate change adaptation strategies for federal forests of the Pacific Northwest, USA: ecological, policy, and socio-economic perspectives. Landscape Ecology. 25(8): 1185-1199. doi:10.1007/s10980-010-9483-0

Spracklen, D.V.; Mickley, L.J.; Logan, J.A.; Hudman, R.C.; Yevich, R.; Flannigan, M.D.; Westerling, A.L. 2009. Impacts of climate change from 2000 to 2050 on wildfire activity and carbonaceous aerosol concentrations in the western United States. Journal of Geophysical Research: Atmospheres. 114(D20): D20301 (17 p). doi:10.1029/2008JD010966

Sprugel, D.G. 1991. Disturbance, equilibrium, and environmental variability: what is 'natural' vegetation in a changing environment? Biological Conservation. 58(1): 1-18. doi:10.1016/0006-3207(91)90041-7

Spurr, S.H. 1952. Origin of the concept of forest succession. Ecology. 33(3): 426-427. doi:10.2307/1932839

Spurr, S.H. 1981. Clearcutting on national forests. Natural Resources Journal. 21(2): 223-243. http://lawlibrary.unm.edu/nrj/21/2/02_spurr_clearcutting.pdf

Stadler, B.; Solinger, S.; Michalzik, B. 2001. Insect herbivores and the nutrient flow from the canopy to the soil in coniferous and deciduous forests. Oecologia. 126(1): 104-113. doi:10.1007/s004420000514

St. George, R.A. 1930. Drought-affected and injured trees attractive to bark beetles. Journal of Economic Entomology. 23(5): 825-828. doi:10.1093/jee/23.5.825

Stanturf, J.A.; Madsen, P., eds. 2005. Restoration of boreal and temperate forests. Boca Raton, FL: CRC Press. 569 p. isbn:1-56670-635-1

Stanturf, J.A.; Lamb, D.; Madsen, P. 2012. Forest landscape restoration: Integrating natural and social sciences. World Forests XV. New York: Springer. 350 p. isbn:978-94-007-5325-9

Stark, N.M. 1977. Fire and nutrient cycling in a Douglas-fir/larch forest. Ecology. 58(1): 16-30. doi:10.2307/1935105

Stark, R.W. 1987. Impacts of forest insects and diseases: significance and measurement. Critical Reviews in Plant Sciences. 5(2): 161-203. doi:10.1080/07352688709382238

Starker, T.J. 1915. Recommendations for cutting inferior species on the Whitman National Forest, Oregon. Unpublished report obtained from the National Archives, College Park, Maryland; record group 95. 10 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5414184.pdf

Starker, T.J. 1916. Instructions for marking timber in the western yellow pine region, Pacific Northwest District. Unpublished report obtained from the National Archives, College Park, Maryland; record group 95. 11 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5414126.pdf

Starker, T.J. 1934. Fire resistance in the forest. Journal of Forestry. 32(4): 462-467. doi:10.1093/jof/32.4.462

Starns, H.D.; Fuhlendorf, S.D.; Elmore, R.D.; Twidwell, D.; Thacker, E.T.; Hovick, T.J.; Luttbeg, B. 2019. Recoupling fire and grazing reduces wildland fuel loads on rangelands. Ecosphere. 10(1): e02578 (15 p). doi:10.1002/ecs2.2578

Steele, R. 1984. An approach to classifying seral vegetation within habitat types. Northwest Science. 58(1): 29-39. Steele1984

Steele, R. 1994. The role of succession in forest health. Journal of Sustainable Forestry. 2(1-2): 183-190. doi:10.1300/J091v02n01_08

Steele, R.; Pfister, R.D. 1991. Western-montane plant communities and forest ecosystem perspectives. In: Harvey, A.E.; Neuenschwander, L.F., comps. Proceedings–management and productivity of western-montane forest soils. Gen. Tech. Rep. INT-280. Ogden, UT: USDA Forest Service, Intermountain Research Station: 20-28. http://www.treesearch.fs.fed.us/pubs/33908

Steele, R.; Pfister, R.D.; Ryker, R.A.; Kittams, J.A. 1981. Forest habitat types of central Idaho. Gen. Tech. Rep. INT-114. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 138 p. https://www.fs.usda.gov/treesearch/pubs/40120

Steele, R.; Williams, R.E.; Weatherby, J.C.; Reinhardt, E.D.; Hoffman, J.T.; Thier, R.W. 1996. Stand hazard rating for central Idaho forests. Gen. Tech. Rep. INT-GTR-332. Ogden, UT: USDA Forest Service, Intermountain Research Station. 29 p. https://www.fs.usda.gov/treesearch/pubs/23915

Steele, C.L.; Katoh, S.; Bohlmann, J.; Croteau, R. 1998. Regulation of oleoresinosis in grand fir (Abies grandis): differential transcriptional control of monoterpene, sesquiterpene, and diterpene synthase genes in response to wounding. Plant Physiology. 116: 1497-1504. doi:10.1104/pp.116.4.1497

Steele, B.M.; Reddy, S.K.; Keane, R.E. 2006. A methodology for assessing departure of current plant communities from historical conditions over large landscapes. Ecological Modelling. 199(1): 53-63. doi:10.1016/j.ecolmodel.2006.06.016

Steelman, T.A.; Burke, C.A. 2007. Is wildfire policy in the United States sustainable? Journal of Forestry. 105(2): 67-72. doi:10.1093/jof/105.2.67

Steelman, T.A.; McCaffrey, S.M. 2011. What is limiting more flexible fire management—Public or agency pressure? Journal of Forestry. 109(8): 454-461. doi:10.1093/jof/109.8.454

Steen, H.K. 1999. Forest and wildlife science in America: a history. Durham, NC: Forest History Society. 455 p. isbn:0-89030-057-7

Steen, O.A.; Dawson, R.J.; Armleder, H.M. 2008. An old-growth index for Douglas-fir stands in portions of the interior Douglas-fir zone, central British Columbia. BC Journal of Ecosystems and Management. 9(2): 31-47. https://jem-online.org/index.php/jem/article/download/394/309

Stein, S.J. 1988. Explanations of the imbalanced age structure and scattered distribution of ponderosa pine within a high-elevation mixed coniferous forest. Forest Ecology and Management. 25(2): 139-153. doi:10.1016/0378-1127(88)90125-9

Steinberg, P.D. 2001. *Populus balsamifera* subsp. *trichocarpa*. In: Fire Effects Information System [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us /database/feis/plants/tree/popbalt/all.html

Steinberg, P.D. 2002. *Pseudotsuga menziesii* var. *glauca*. In: Fire Effects Information System [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us /database/feis/plants/tree/psemeng/all.html

Stenseth, N.C.; Mysterud, A.; Ottersen, G.; Hurrell, J.W.; Chan, K.-S.; Lima, M. 2002. Ecological effects of climate fluctuations. Science. 297(5585): 1292-1296. doi:10.1126/science.1071281

Stephens, S.L. 1998. Evaluation of the effects of silvicultural and fuels treatments on potential fire behaviour in Sierra Nevada mixed-conifer forests. Forest Ecology and Management. 105(1-3): 21-35. doi:10.1016/S0378-1127(97)00293-4

Stephens, J.L.; Alexander, J.D. 2011. Effects of fuel reduction on bird density and reproductive success in riparian areas of mixed-conifer forest in southwest Oregon. Forest Ecology and Management. 261(1): 43-49. doi:10.1016/j.foreco.2010.09.022

Stephens, S.L.; Collins, B.M. 2004. Fire regimes of mixed conifer forests in the north-central Sierra Nevada at multiple spatial scales. Northwest Science. 78(1): 12-23. Stephens and Collins

Stephens, S.L.; Moghaddas, J.J.; Edminster, C.; Fiedler, C.E.; Haase, S.; Harrington, M.; Keeley, J.E.; Knapp, E.E.; McIver, J.D.; Metlen, K.; Skinner, C.N.; Youngblood, A. 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. Ecological Applications. 19(2): 305-320. doi:10.1890/07-1755.1

Stephens, S.L.; Millar, C.I.; Collins, B.M. 2010. Operational approaches to managing forests of the future in Mediterranean regions within a context of changing climates. Environmental Research Letters. 5(2): 024003 (9 p) doi:10.1088/1748-9326/5/2/024003

Stephens, S.L.; Boerner, R.E.J.; Moghaddas, J.J.; Moghaddas, E.E.Y.; Collins, B.M.; Dow, C.B.; Edminster, C.; Fiedler, C.E.; Fry, D.L.; Hartsough, B.R.; Keeley, J.E.; Knapp, E.E.; McIver, J.D.; Skinner, C.N.; Youngblood, A. 2012a. Fuel treatment impacts on estimated wildfire carbon loss from forests in Montana, Oregon, California, and Arizona. Ecosphere. 3(5): art38. doi:10.1890/es11-00289.1

Stephens, S.L.; Collins, B.M.; Roller, G. 2012b. Fuel treatment longevity in a Sierra Nevada mixed conifer forest. Forest Ecology and Management. 285: 204-212. doi:10.1016/j.foreco.2012.08.030

Stephens, S.L.; McIver, J.D.; Boerner, R.E.J.; Fettig, C.J.; Fontaine, J.B.; Hartsough, B.R.; Kennedy, P.L.; Schwilk, D.W. 2012c. The effects of forest fuel-reduction treatments in the United States. BioScience. 62(6): 549-560. doi:10.1525/bio.2012.62.6.6

Stephens, S.L.; Burrows, N.; Buyantuyev, A.; Gray, R.W.; Keane, R.E.; Kubian, R.; Liu, S.; Seijo, F.; Shu, L.; Tolhurst, K.G.; van Wagtendonk, J.W. 2014. Temperate and boreal forest mega-fires: characteristics and challenges. Frontiers in Ecology and the Environment. 12(2): 115-122. doi:10.1890/120332

Stephens, S.L.; Lydersen, J.M.; Collins, B.M.; Fry, D.L.; Meyer, M.D. 2015. Historical and current landscape-scale ponderosa pine and mixed conifer forest structure in the Southern Sierra Nevada. Ecosphere. 6(5): art79 (63 p). doi:10.1890/es14-00379.1

Stephenson, N.L.; van Mantgem, P.J.; Bunn, A.G.; Bruner, H.; Harmon, M.E.; O'Connell, K.B.; Urban, D.L.; Franklin, J.F. 2011. Causes and implications of the correlation between forest productivity and tree mortality rates. Ecological Monographs. 81(4): 527-555. doi:10.1890/10-1077.1

Stevens, W.R. 1935. Meteorological conditions preceding thunderstorms on the national forests: 2. The Blue Mountains of Washington and Oregon. Monthly Weather Review. 63(6): 183-184. doi:10.1175/1520-0493(1935)63<183:Mcptot>2.0.Co;2

Stevens, W.K. 1990. New eye on nature: the real constant is eternal turmoil. New York: New York Times, Science Column for Tuesday, July 31, 1990. 2 p. new-eye-on-nature-the-real-constant-is-eternal-turmoil

Stevens, M.T.; Turner, M.G.; Tuskan, G.A.; Romme, W.H.; Gunter, L.E.; Waller, D.M. 1999. Genetic variation in postfire aspen seedlings in Yellowstone National Park. Molecular Ecology. 8(11): 1769-1780. doi:10.1046/j.1365-294x.1999.00732.x

Stevens-Rumann, C.S.; Sieg, C.H.; Hunter, M.E. 2012. Ten years after wildfires: How does varying tree mortality impact fire hazard and forest resiliency? Forest Ecology and Management. 267: 199-208. doi:10.1016/j.foreco.2011.12.003

Stewart, O.C. 1951. Burning and natural vegetation in the United States. Geographical Review. 41(2): 317-320. http://www.jstor.org/stable/211026

Stewart, R.E. 1975. Allelopathic potential of western bracken. Journal of Chemical Ecology. 1(2): 161-169. doi:10.1007/BF00987865

Stewart, O.C. 2009. Forgotten fires: Native Americans and the transient wilderness. Norman, OK: University of Oklahoma Press. 364 p. isbn:978-0-8061-4037-7

Stewart, I.T.; Cayan, D.R.; Dettinger, M.D. 2004. Changes in snowmelt runoff timing in western North America under a 'business as usual' climate change scenario. Climatic Change. 62(1): 217-232. doi:10.1023/B:CLIM.0000013702.22656.e8

Stickney, P.F. 1990. Early development of vegetation following holocaustic fire in northern Rocky Mountain forests. Northwest Science. 64(5): 243-246. http://hdl.handle.net/2376/1678

Stidham, M.; Simon-Brown, V. 2011. Stakeholder perspectives on converting forest biomass to energy in Oregon, USA. Biomass and Bioenergy. 35(1): 203-213. doi:10.1016/j.biombioe.2010.08.014

Stiling, P.; Moon, D.; Rossi, A.; Hungate, B.A.; Drake, B.G. 2009. Seeing the forest for the trees: long-term exposure to elevated CO_2 increases some herbivore densities. Global Change Biology. 15(8): 1895-1902. doi:10.1111/j.1365-2486.2009.01902.x

Stockdale, C.A.; McLoughlin, N.; Flannigan, M.; Macdonald, S.E. 2019. Could restoration of a landscape to a pre-European historical vegetation condition reduce burn probability? Ecosphere. 10(2): e02584 (18 p). doi:10.1002/ecs2.2584

Stockmann, K.D.; Hyde, K.D.; Jones, J.G.; Loeffler, D.R.; Silverstein, R.P. 2010. Integrating fuel treatment into ecosystem management: a proposed project planning process. International Journal of Wildland Fire. 19(6): 725-736. doi:10.1071/WF08108

Stoszek, K.J. 1988a. Forests under stress and insect outbreaks. Northwest Environmental Journal. 4: 247-261.

Stoszek, K.J. 1988b. Prospects for management prevention of disturbances in forests of the mountain west. In: Schmidt, W.C., comp. Proceedings–future forests of the mountain west: a stand culture symposium. Gen. Tech. Rep. INT-243. Ogden, UT: USDA Forest Service, Intermountain Research Station: 267-274. https://archive.org/download/CAT88897341/CAT88897341.pdf

Stoszek, K.J.; Mika, P.G. 1984. Application of hazard rating models in Douglas-fir tussock moth and spruce budworm management strategies. In: Baumgartner, D.M.; Mitchell, R., comps. Silvicultural management strategies for pests of the interior Douglas-fir and grand fir forest types. Pullman, WA: Washington State University, Cooperative Extension: 143-151.

Stoszek, K.J.; Mika, P.G.; Moore, J.A.; Osborne, H.L. 1981. Relationships of Douglas-fir tussock moth defoliation to site and stand characteristics in northern Idaho. Forest Science. 27(3): 431-442. doi:10.1093/forestscience/27.3.431

Strauss, D.; Bednar, L.; Mees, R. 1989. Do one percent of forest fires cause ninety-nine percent of the damage? Forest Science. 35(2): 319-328. doi:10.1093/forestscience/35.2.319

Strickler, G.S.; Edgerton, P.J. 1976. Emergent seedlings from coniferous litter and soil in eastern Oregon. Ecology. 57(4): 801-807. doi:10.2307/1936193

Strunk, J.L.; Harrington, C.A.; Brodie, L.C.; Prevéy, J.S. 2020. Seeing the forest below the trees: occurrences of shrubs in the Pacific Northwest. Gen. Tech. Rep. PNW-GTR-980. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 87 p. https://www.fs.usda.gov/treesearch/pubs/59327

Stuart, J.D.; Agee, J.K.; Gara, R.I. 1989. Lodgepole pine regeneration in an old, self-perpetuating forest in south central Oregon. Canadian Journal of Forest Research. 19(9): 1096-1104. doi:10.1139/x89-166

Stump, L.M.; Binkley, D. 1993. Relationships between litter quality and nitrogen availability in Rocky Mountain forests. Canadian Journal of Forest Research. 23(3): 492-502. doi:10.1139/x93-067

Sturrock, R.N.; Frankel, S.J.; Brown, A.V.; Hennon, P.E.; Kliejunas, J.T.; Lewis, K.J.; Worrall, J.J.; Woods, A.J. 2011. Climate change and forest diseases. Plant Pathology. 60(1): 133-149. doi:10.1111/j.1365-3059.2010.02406.x

Sturtevant, B.R.; Scheller, R.M.; Miranda, B.R.; Shinneman, D.; Syphard, A. 2009. Simulating dynamic and mixed-severity fire regimes: A process-based fire extension for LANDIS-II. Ecological Modelling. 220(23): 3380-3393. doi:10.1016/j.ecolmodel.2009.07.030

Suding, K.N.; Gross, K.L.; Houseman, G.R. 2004. Alternative states and positive feedbacks in restoration ecology. Trends in Ecology and Evolution. 19(1): 46-53. doi:10.1016/j.tree.2003.10.005

Sudworth, G.B. 1916. The spruce and balsam fir trees of the Rocky Mountain region. Bull. No. 327. Washington, DC: U.S. Department of Agriculture. 43 p. https://www.biodiversitylibrary.org/itempdf/190231

Suffling, R.; Perera, A.H. 2004. Characterizing natural forest disturbance regimes: concepts and approaches. In: Perera, A.H.; Buse, L.J.; Weber, M.G., eds. Emulating natural forest landscape disturbances: concepts and applications. New York: Columbia University Press: 43-54. isbn:0-231-12916-5

Sutherland, S.; Nelson, C.R. 2010. Nonnative plant response to silvicultural treatments: A model based on disturbance, propagule pressure, and competitive abilities. Western Journal of Applied Forestry. 25(1): 27-33. doi:10.1093/wjaf/25.1.27

Swanson, D.K. 2007 (October 5). Site visit to old clearcuts on the Brock Allotment (Fry Meadow area), Walla Walla Ranger District, Umatilla National Forest; file designation 2060 memorandum to District Ranger. Baker City, OR: USDA Forest Service, Malheur, Umatilla, and Wallowa-Whitman National Forests, Area Ecology. 6 p.

Swanson, F.J.; Jones, J.A.; Wallin, D.O.; Cissel, J.H. 1994. Natural variability–implications for ecosystem management. In: Jensen, M.E.; Bourgeron, P.S., eds. Volume II; Ecosystem management: principles and applications. Gen. Tech. Rep. PNW-GTR-318. Portland, OR: USDA Forest Service, Pacific Northwest Research Station: 80-94. https://www.fs.usda.gov/treesearch/pubs/6223

Swanson, F.J.; Jones, J.A.; Grant, G.E. 1997. The physical environment as a basis for managing ecosystems. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century: the science of ecosystem management. Washington, DC: Island Press: 229-238.

isbn:1-55963-399-9

Swanson, D.K.; Schmitt, C.L.; Shirley, D.M.; Erickson, V.; Schuetz, K.J.; Tatum, M.L.; Powell, D.C. 2010. Aspen biology, community classification, and management in the Blue Mountains. Gen. Tech. Rep. PNW-GTR-806. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 117 p. http://www.treesearch.fs.fed.us/pubs/35257

Swanson, M.E.; Franklin, J.F.; Beschta, R.L.; Crisafulli, C.M.; DellaSala, D.A.; Hutto, R.L.; Lindenmayer, D.B.; Swanson, F.J. 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment. 9(2): 117-125. doi:10.1890/090157

Swetnam, T.W.; Lynch, A.M. 1993. Multicentury, regional-scale patterns of western spruce budworm outbreaks. Ecological Monographs. 63(4): 399-424. doi:10.2307/2937153

Swetnam, T.W.; Wickman, B.E.; Paul, H.G.; Baisan, C. 1995. Historical patterns of western spruce budworm and Douglas-fir tussock moth outbreaks in the northern Blue Mountains, Oregon, since A.D. 1700. Res. Pap. PNW-RP-484. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 27 p. http://www.treesearch.fs.fed.us/pubs/20594

Swetnam, T.; Falk, D.A.; Hessl, A.E.; Farris, C. 2011. Reconstructing landscape pattern of historical fires and fire regimes. In: McKenzie, D.; Miller, C.; Falk, D.A., eds. The landscape ecology of fire. New York: Springer: 165-192. isbn:978-94-007-0300-1

Symstad, A.J.; Tilman, D.; Peters, D.P.C.; Wall, D.H.; Chapin, F.S.; Mittelbach, G.G.; Gross, K.L.; Huenneke, L.F. 2003. Long-term and large-scale perspectives on the relationship between biodiversity and ecosystem functioning. BioScience. 53(1): 89-98. doi:10.1641/0006-3568(2003)053[0089:ltalsp]2.0.co;2

Szaro, R.C.; Johnson, N.C.; Sexton, W.T.; Malk, A.J., eds. 1999. Ecological stewardship: a common reference for ecosystem management. Volume II. Oxford, UK: Elsevier Science, Ltd. 741 p. isbn:0-08-043206-9

Szaro, R.C.; Boyce, D.A.; Puchlerz, T. 2005. The challenges associated with developing science-based landscape scale management plans. Landscape and Urban Planning. 72(1): 3-12. doi:10.1016/j.landurbplan.2004.09.011

Taeroe, A.; de Koning, J.H.C.; Löf, M.; Tolvanen, A.; Heiðarsson, L.; Raulund-Rasmussen, K. 2019. Recovery of temperate and boreal forests after windthrow and the impacts of salvage logging. A quantitative review. Forest Ecology and Management. 446: 304-316. doi:10.1016/j.foreco.2019.03.048

Tainter, F.H.; Baker, F.A. 1996. Principles of forest pathology. New York: John Wiley & Sons, Inc. 805 p. isbn:0-471-12952-6

Takekawa, J.Y.; Garton, E.O. 1984. How much is an evening grosbeak worth? Journal of Forestry. 82(7): 426-428. doi:10.1093/jof/82.7.426

Tanaka, J.A.; Starr, G.L.; Quigley, T.M. 1995. Strategies and recommendations for addressing forest health issues in the Blue Mountains of Oregon and Washington. Gen. Tech. Rep. PNW-GTR-350. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 18 p. http://www.treesearch.fs.fed.us/pubs/8970

Tappeiner, J.C. II; Maguire, D.A.; Harrington, T.B. 2007. Silviculture and ecology of western U.S. forests. Corvallis, OR: Oregon State University Press. 440 p. isbn:978-0-87071-187-9

Taylor, W.P. 1934. Significance of extreme or intermittent conditions in distribution of species and management of natural resources, with a restatement of Liebig's law of minimum. Ecology. 15(4): 374-379. doi:10.2307/1932352

Taylor, A.H. 2000. Fire regimes and forest changes in mid and upper montane forests of the southern Cascades, Lassen Volcanic National Park, California, U.S.A. Journal of Biogeography. 27(1): 87-104. doi:10.1046/j.1365-2699.2000.00353.x

Taylor, A.H.; Beaty, R.M. 2005. Climatic influences on fire regimes in the northern Sierra Nevada mountains, Lake Tahoe Basin, Nevada, USA. Journal of Biogeography. 32(3): 425-438. doi:10.1111/j.1365-2699.2004.01208.x

Taylor, A.H.; Skinner, C.N. 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. Forest Ecology and Management. 111(2-3): 285-301. doi:10.1016/S0378-1127(98)00342-9

Taylor, A.H.; Skinner, C.N. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications. 13(3): 704-719. doi:10.1890/1051-0761(2003)013[0704:Spacoh]2.0.Co;2

Taylor, B.R.; Prescott, C.E.; Parsons, W.J.F.; Parkinson, D. 1991. Substrate control of litter decomposition in four Rocky Mountain coniferous forests. Canadian Journal of Botany. 69(10): 2242-2250. doi:10.1139/b91-281

Taylor, P.D.; Fahrig, L.; Henein, K.; Merriam, G. 1993. Connectivity is a vital element of landscape structure. Oikos. 68(3): 571-573. doi:10.2307/3544927

Taylor, L.; Zelnik, J.; Cadwallader, S.; Hughes, B., eds. 2005. Mixed severity fire regimes: ecology and management. AFE MISC03. Pullman, WA: Washington State University Extension. 278 p.

Tecle, A.; Duckstein, L.; Covington, W.W.; Wood, D.B. 1989. Uncertainty and risk in forest management. In: Tecle, A.; Covington, W.W.; Hamre, R.H., tech. coords. Multiresource management of ponderosa pine forests. Gen. Tech. Rep. RM-185. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station: 208-212. https://archive.org/download/CAT10400419/CAT10400419.pdf

Temesgen, H.; Mitchell, S.J. 2005. An individual-tree mortality model for complex stands of southeastern British Columbia. Western Journal of Applied Forestry. 20(2): 101-109. doi:10.1093/wjaf/20.2.101

Temperli, C.; Hart, S.J.; Veblen, T.T.; Kulakowski, D.; Hicks, J.J.; Andrus, R. 2014. Are density reduction treatments effective at managing for resistance or resilience to spruce beetle disturbance in the southern Rocky Mountains? Forest Ecology and Management. 334: 53-63. doi:10.1016/j.foreco.2014.08.028

Tepley, A.J.; Swanson, F.J.; Spies, T.A. 2013. Fire-mediated pathways of stand development in Douglas-fir/western hemlock forests of the Pacific Northwest, USA. Ecology. 94(8): 1729-1743. doi:10.1890/12-1506.1

Tesch, S.D. 1981. Comparative stand development in an old-growth Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) forest in western Montana. Canadian Journal of Forest Research. 11(1): 82-89. doi:10.1139/x81-011

Tesch, S.D.; Korpela, E.J. 1993. Douglas-fir and white fir advance regeneration for renewal of mixed-conifer forests. Canadian Journal of Forest Research. 23(7): 1427-1437. doi:10.1139/x93-180

Teske, C.C.; Seielstad, C.A.; Queen, L.P. 2012. Characterizing fire-on-fire interactions in three large wilderness areas. Fire Ecology. 8(2): 82-106. doi:10.4996/fireecology.0802082

Teste, F.P.; Lieffers, V.J.; Landhäusser, S.M. 2011a. Seed release in serotinous lodgepole pine forests after mountain pine beetle outbreak. Ecological Applications. 21(1): 150-162. doi:10.1890/09-1881.1

Teste, F.P.; Lieffers, V.J.; Landhäusser, S.M. 2011b. Viability of forest floor and canopy seed banks in *Pinus contorta* var. *latifolia* (Pinaceae) forests after a mountain pine beetle outbreak. American Journal of Botany. 98(4): 630-637. doi:10.3732/ajb.1000252

Thies, W.G. 2001. Root diseases in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 38-45. http://hdl.handle.net/2376/979

Thies, W.G.; Sturrock, R.N. 1995. Laminated root rot in western North America. Res. Bull.

PNW-GTR-349. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 32 p. https://www.fs.usda.gov/treesearch/pubs/3065

Thies, W.G.; Westlind, D.J. 2005. Stump removal and fertilization of five *Phellinus weirii*-infested stands in Washington and Oregon affect mortality and growth of planted Douglas-fir 25 years after treatment. Forest Ecology and Management. 219(2–3): 242-258. doi:10.1016/j.foreco.2005.08.050

Thies, W.G.; Westlind, D.J. 2012. Validating the Malheur model for predicting ponderosa pine post-fire mortality using 24 fires in the Pacific Northwest, USA. International Journal of Wildland Fire. 21(5): 572-582. doi:10.1071/WF10091

Thies, W.G.; Westlind, D.J.; Loewen, M. 2005. Season of prescribed burn in ponderosa pine forests in eastern Oregon: impact on pine mortality. International Journal of Wildland Fire. 14(3): 223-231. doi:10.1071/WF04051

Thies, W.G.; Westlind, D.J.; Loewen, M.; Brenner, G. 2006. Prediction of delayed mortality of fire-damaged ponderosa pine following prescribed fires in eastern Oregon, USA. International Journal of Wildland Fire. 15(1): 19-29. doi:10.1071/WF05025

Thies, W.G.; Westlind, D.J.; Loewen, M.; Brenner, G. 2008. A field guide to predict delayed mortality of fire-damaged ponderosa pine: Application and validation of the Malheur model. Gen. Tech. Rep. PNW-GTR-769. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 16 p. http://www.treesearch.fs.fed.us/pubs/31092

Thiffault, E.; Bélanger, N.; Paré, D.; Munson, A.D. 2007. How do forest harvesting methods compare with wildfire? A case study of soil chemistry and tree nutrition in the boreal forest. Canadian Journal of Forest Research. 37(9): 1658-1668. doi:10.1139/X07-031

Thiffault, E.; Hannam, K.; Quideau, S.; Paré, D.; Bélanger, N.; Oh, S.-W.; Munson, A. 2008. Chemical composition of forest floor and consequences for nutrient availability after wildfire and harvesting in the boreal forest. Plant and Soil. 308(1): 37-53. doi:10.1007/s11104-008-9604-6

Thomas, J.W., tech. ed. 1979. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Agric. Handb. No. 553. Washington, DC: USDA Forest Service. 512 p. https://www.fs.usda.gov/treesearch/pubs/6630

Thomas, J.W. 1995. The Forest Service ethics and course to the future. Landscape and Urban Planning. 32(3): 157-159. doi:10.1016/0169-2046(94)00196-A

Thomas, K.D.; Prescott, C.E. 2000. Nitrogen availability in forest floors of three tree species on the same site: the role of litter quality. Canadian Journal of Forest Research. 30(11): 1698-1706. doi:10.1139/x00-101

Thomas, J.W.; Black, H., Jr.; Scherzinger, R.J.; Pedersen, R.J. 1979a. Deer and elk. In: Thomas, J.W., tech. ed. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Agric. Handb. No. 553. Washington, DC: USDA Forest Service: 104-127. https://www.fs.usda.gov/treesearch/pubs/6630

Thomas, J.W.; Maser, C.; Rodiek, J.E. 1979b. Edges. In: Thomas, J.W., tech. ed. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Agric. Handb. No. 553. Washington, DC: USDA Forest Service: 48-59. https://www.fs.usda.gov/treesearch/pubs/6630

Thompson, D.C. 1989. The effect of stand structure and stand density on the leaf area–sapwood area relationship of lodgepole pine. Canadian Journal of Forest Research. 19(3): 392-396. doi:10.1139/x89-061

Thompson, I.D. 2002. Ecological basis for emulating natural disturbances in forest management. In: Buse, L.J.; Perera, A.H., comps. Emulating natural forest landscape disturbances: concepts and applications. Popular summaries. For. Res. Infor. Pap. No. 149. Sault Ste. Marie, ON: Ontario Ministry of Natural Resources, Ontario Forest Research Institute: 43-45. http://www.ontla.on.ca/library/repository/mon/30010/225303.pdf

Thompson, I.; Christophersen, T. 2008. Cross-sectoral toolkit for the conservation and sustainable management of forest biodiversity. Technical Series No. 39. Montreal, QB: Secretariat of the Convention on Biological Diversity. 53 p. http://www.cbd.int/doc/publications/cbd-ts-39-en.pdf

Thompson, I.D.; Harestad, A.S. 2004. The ecological and genetic basis for emulating natural disturbance in forest management. In: Perera, A.H.; Buse, L.J.; Weber, M.G., eds. Emulating natural forest landscape disturbances: concepts and applications. New York: Columbia University Press: 29-42. isbn:0-231-12916-5

Thompson, J.R.; Spies, T.A. 2009. Vegetation and weather explain variation in crown damage within a large mixed-severity wildfire. Forest Ecology and Management. 258(7): 1684-1694. doi:10.1016/j.foreco.2009.07.031

Thompson, R.S.; Hostetler, S.W.; Bartlein, P.J.; Anderson, K.H. 1998. A strategy for assessing potential future changes in climate, hydrology, and vegetation in the western United States. Circular 1153. Denver, CO: U.S. Department of the Interior, U.S. Geological Survey. 20 p. https://pubs.er.usgs.gov/publication/cir1153

Thompson, J.R.; Spies, T.A.; Ganio, L.M. 2007. Reburn severity in managed and unmanaged vegetation in a large wildfire. Proceedings of the National Academy of Sciences. 104(25): 10743-10748. doi:10.1073/pnas.0700229104

Thompson, J.R.; Duncan, S.L.; Johnson, K.N. 2009. Is there potential for the historical range of variability to guide conservation given the social range of variability? Ecology and Society. 14(1): article 18 (14 p). http://www.ecologyandsociety.org/vol14/iss1/art18/

Thompson, J.R.; Spies, T.A.; Olsen, K.A. 2011. Canopy damage to conifer plantations within a large mixed-severity wildfire varies with stand age. Forest Ecology and Management. 262(3): 355-360. doi:10.1016/j.foreco.2011.04.001

Tidwell, T. 2012. US Forest Service land management: challenges and opportunities for achieving healthier national forests. Statement before the United States House of Representatives, Committee on Agriculture, Subcommittee on Conservation, Energy, and Forestry. 6 p. https://www.govinfo.gov/content/pkg/CHRG-112hhrg73659/html/CHRG-112hhrg73659.htm

Tiedemann, A.R.; Anderson, T.D. 1980. Combustion losses of sulfur from native plant materials and forest litter. In: Martin, R.E.; Edmonds, R.L.; Faulkner, D.A.; Harrington, J.B.; Fuquay, D.M.; Stocks, B.J.; Barr, S., eds. Proceedings: sixth conference on fire and forest meteorology. Washington, DC: Society of American Foresters: 220-227.

Tilman, D.; Knops, J.; Wedin, D.; Reich, P.; Ritchie, M.; Siemann, E. 1997. The influence of functional diversity and composition on ecosystem processes. Science. 277(5330): 1300-1302. doi:10.1126/science.277.5330.1300

Tingley, M.W.; Ruiz-Gutiérrez, V.; Wilkerson, R.L.; Howell, C.A.; Siegel, R.B. 2016. Pyrodiversity promotes avian diversity over the decade following forest fire. Proceedings of the Royal Society B: Biological Sciences. 283(1840): 20161703 (9 p). doi:10.1098/rspb.2016.1703

Tinnin, R.O.; Kirkpatrick, L.A. 1985. The allelopathic influence of broadleaf trees and shrubs on seedlings of Douglas-fir. Forest Science. 31(4): 945-952. doi:10.1093/forestscience/31.4.945

Tinnin, R.O.; Hawksworth, F.G.; Knutson, D.M. 1982. Witches' broom formation in conifers infected by *Arceuthobium* spp.: an example of parasitic impact upon community dynamics. American Midland Naturalist. 107(2): 351-359. doi:10.2307/2425385

Tinnin, R.O.; Parks, C.G.; Knutson, D.M. 1999. Effects of Douglas-fir dwarf mistletoe on trees in thinned stands in the Pacific Northwest. Forest Science. 45(3): 359-365. doi:10.1093/forestscience/45.3.359

Tittler, R.; Messier, C.; Fall, A. 2012. Concentrating anthropogenic disturbance to balance ecological and economic values: applications to forest management. Ecological Applications. 22(4): 1268-1277. doi:10.1890/11-1680.1

Tomback, D.F.; Achuff, P. 2010. Blister rust and western forest biodiversity: ecology, values and outlook for white pines. Forest Pathology. 40(3-4): 186-225. doi:10.1111/j.1439-0329.2010.00655.x

Tomback, D.F.; Arno, S.F.; Keane, R.E., eds. 2001. Whitebark pine communities, ecology and restoration. Washington, DC: Island Press. 440 p. isbn:1-55963-718-8

Torgersen, T.R. 2001. Defoliators in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 11-20. http://hdl.handle.net/2376/976

Torgersen, T.R.; Bull, E.L. 1995. Down logs as habitat for forest-dwelling ants–the primary prey of pileated woodpeckers in northeastern Oregon. Northwest Science. 69(4): 294-302.

Torgersen, T.R.; Campbell, R.W. 1982. Some effects of avian predators on the western spruce budworm in north central Washington. Environmental Entomology. 11(2): 429-431. doi:10.1093/ee/11.2.429

Torgersen, T.R.; Mason, R.R.; Campbell, R.W. 1990. Predation by birds and ants on two forest insect pests in the Pacific Northwest. Studies in Avian Biology. 13: 14-19. PredationbyBirdsandAntsonTwoForestInsectPests

Torgersen, T.R.; Powell, D.C.; Hosman, K.P.; Schmidt, F.H. 1995. No long-term impact of carbaryl treatment on western spruce budworm populations and host trees in the Malheur National Forest, Oregon. Forest Science. 41(4): 851-863. doi:10.1093/forestscience/41.4.851

Trappe, J.M. 1959. Lodgepole pine clearcuts in northeastern Oregon. Journal of Forestry. 57(6): 420-423. doi:10.1093/jof/57.6.420

Trappe, J.M.; Harris, R.W. 1958. Lodgepole pine in the Blue Mountains of northeastern Oregon. Res. Pap. 30. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 22 p. https://www.fs.usda.gov/treesearch/pubs/26056

Trappe, J.M.; Molina, R.; Castellano, M. 1984. Reactions of mycorrhizal fungi and mycorrhiza formation to pesticides. Annual Review of Phytopathology. 22(1): 331-359. doi:10.1146/annurev.py.22.090184.001555

Trombulak, S.C.; Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology. 14(1): 18-30. doi:10.1046/j.1523-1739.2000.99084.x

Trouet, V.; Taylor, A.H.; Carleton, A.M.; Skinner, C.N. 2006. Fire-climate interactions in forests of the American Pacific coast. Geophysical Research Letters. 33: L18704 (5 p). doi:10.1029/2006GL027502

Trouet, V.; Taylor, A.H.; Wahl, E.R.; Skinner, C.N.; Stephens, S.L. 2010. Fire-climate interactions in the American West since 1400 CE. Geophysical Research Letters. 37(4): L04702 (5 p). doi:10.1029/2009gl041695

Trzcinski, M.K.; Reid, M.L. 2008. Effect of management on the spatial spread of mountain pine beetle (*Dendroctonus ponderosae*) in Banff National Park. Forest Ecology and Management. 256(6): 1418-1426. doi:10.1016/j.foreco.2008.07.003

Tucker, G.J. 1940. History of the northern Blue Mountains. Unpub. Rep. [Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest.] 170 p. On file with: USDA Forest Service, Umatilla National Forest, Supervisor's Office, Pendleton, OR. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev7_015563.pdf

Tunnock, S.; Ryan, R.B. 1985. Larch casebearer in western larch. Forest Insect & Disease Leaflet 96. [Place of publication unknown]: USDA Forest Service. 8 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_042976.pdf

Turchi, G.M.; Kennedy, P.L.; Urban, D.; Hein, D. 1995. Bird species richness in relation to isolation of aspen habitats. Wilson Bulletin. 107(3): 463-474. https://www.jstor.org/stable/4163570

Turner, D.P. 1985. Successional relationships and a comparison of biological characteristics among six northwestern conifers. Bulletin of the Torrey Botanical Club. 112(4): 421-428. doi:10.2307/2996045

Turner, M.G. 1998. Landscape ecology: living in a mosaic. In: Dodson, S.I.; Allen, T.F.H.; Carpenter, S.R.; Ives, A.R.; Jeanne, R.L.; Kitchell, J.F.; Langston, N.E.; Turner, M.G. Ecology. New York: Oxford University Press: 77-122. isbn:0-19-512079-5

Turner, N.J. 1999. "Time to burn." Traditional use of fire to enhance resource production by aboriginal peoples in British Columbia. In: Boyd, R., ed. Indians, fire, and the land in the Pacific Northwest. Corvallis, OR: Oregon State University Press: 185-218. isbn:0-87071-459-7

Turner, M.G. 2005. Landscape ecology: What is the state of the science? Annual Review of Ecology, Evolution, and Systematics. 36(1): 319-344. doi:10.1146/annurev.ecolsys.36.102003.152614

Turner, M.G.; Dale, V.H. 1998. Comparing large, infrequent disturbances: What have we learned? Ecosystems. 1(6): 493-496. doi:10.1007/s100219900045

Turner, M.G.; Romme, W.H. 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology. 9(1): 59-77. doi:10.1007/BF00135079

Turner, M.G.; Gardner, R.H.; Dale, V.H.; O'Neill, R.V. 1989. Predicting the spread of disturbance across heterogeneous landscapes. Oikos. 55(1): 121-129. doi:10.2307/3565881

Turner, N.J.; Thompson, L.C.; Thompson, M.T.; York, A.Z. 1990. Thompson ethnobotany: knowledge and usage of plants by the Thompson Indians of British Columbia. Memoir No. 3. Victoria, BC: Royal British Columbia Museum. 335 p. isbn:978-0771889165

Turner, M.G.; Hargrove, W.W.; Gardner, R.H.; Romme, W.H. 1994. Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming. Journal of Vegetation Science. 5(5): 731-742. doi:10.2307/3235886

Turner, M.G.; Dale, V.H.; Everham, E.H., III. 1997. Fires, hurricanes, and volcanoes: comparing large disturbances. BioScience. 47(11): 758-768. doi:10.2307/1313098

Turner, M.G.; Baker, W.L.; Peterson, C.J.; Peet, R.K. 1998. Factors influencing succession: lessons from large, infrequent natural disturbances. Ecosystems. 1(6): 511-523. doi:10.1007/s100219900047

Turner, M.G.; Gardner, R.H.; O'Neill, R.V. 2001. Landscape ecology in theory and practice: pattern and process. New York: Springer-Verlag. 401 p. isbn:0-387-95123-7

Turner, B.L.; Kasperson, R.E.; Matson, P.A.; McCarthy, J.J.; Corell, R.W.; Christensen, L.; Eckley, N.; Kasperson, J.X.; Luers, A.; Martello, M.L.; Polsky, C.; Pulsipher, A.; Schiller, A. 2003. A framework for vulnerability analysis in sustainability science. Proceedings of the National Academy of Sciences. 100(14): 8074-8079. doi:10.1073/pnas.1231335100

Turner, N.J.; Davidson-Hunt, I.J.; O'Flaherty, M. 2003. Living on the edge: ecological and cultural edges as sources of diversity for social–ecological resilience. Human Ecology. 31(3): 439-461. doi:10.1023/a:1025023906459

Turner, M.G.; Tinker, D.B.; Romme, W.H.; Kashian, D.M.; Litton, C.M. 2004. Landscape patterns of sapling density, leaf area, and aboveground net primary production in postfire lodgepole pine forests, Yellowstone National Park (USA). Ecosystems. 7(7): 751-775. doi:10.1007/s10021-004-0011-4

Turner, M.G.; Smithwick, E.A.H.; Tinker, D.B.; Romme, W.H. 2009. Variation in foliar nitrogen and aboveground net primary production in young postfire lodgepole pine. Canadian Journal of Forest Research. 39(5): 1024-1035. doi:10.1139/X09-029

Turner, D.P.; Ritts, W.D.; Kennedy, R.E.; Gray, A.N.; Yang, Z. 2016. Regional carbon cycle responses to 25 years of variation in climate and disturbance in the US Pacific Northwest. Regional Environmental Change. 16(8): 2345-2355. doi:10.1007/s10113-016-0956-9

Tuskan, G.A.; Francis, K.E.; Russ, S.L.; Romme, W.H.; Turner, M.G. 1996. RAPD markers reveal diversity within and among clonal and seedling stands of aspen in Yellowstone National Park, U.S.A. Canadian Journal of Forest Research. 26(12): 2088-2098. doi:10.1139/x26-237

Twardus, D.B.; Filip, G.M.; Jaszkowski, R. 1984. Minimizing pest incidence in the grand fir type within the Blue Mountains of eastern Oregon. In: Baumgartner, D.M.; Mitchell, R., comps. Silvicultural management strategies for pests of the interior Douglas-fir and grand fir forest types. Pullman, WA: Washington State University, Cooperative Extension: 171-177.

Uchytil, R.J. 1991. *Abies lasiocarpa*. In: Fire Effects Information System [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us/database/feis/plants/tree/abilas/all.html

Uchytil, R.J. 1991. *Picea engelmannii*. In: Fire Effects Information System [Online]. USDA Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). https://www.fs.fed.us/database/feis/plants/tree/piceng/all.html

Urban, D.L.; O'Neill, R.V.; Shugart, H.H., Jr. 1987. Landscape ecology. BioScience. 37(2): 119-127. doi:10.2307/1310366

U.S. Congress. 2004. Statements and testimony before the House of Representatives, Committee on Resources, Subcommittee on Forests and Forest Health, concerning "In the aftermath of catastrophic events: restoring and protecting communities, water, wildlife and forests." Washington, DC: United States Congress.

USDA Forest Service. 1980. Environmental consequences of timber harvesting in Rocky Mountain coniferous forests. Gen. Tech. Rep. INT-90. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 526 p. https://archive.org/download/CAT80751102/CAT80751102.pdf

USDA Forest Service. 1990. Land and resource management plan: Umatilla National Forest. Portland, OR: USDA Forest Service, Pacific Northwest Region. Irregular pagination. http://www.fs.usda.gov/main/umatilla/landmanagement/planning

USDA Forest Service. 1993a. Region 6 interim old growth definition for Douglas-fir series, grand fir/white fir series, interior Douglas-fir series, lodgepole pine series, Pacific silver fir series, ponderosa pine series, Port-Orford-cedar and tanoak (redwood) series, subalpine fir series, and western hemlock series. Portland, OR: USDA Forest Service, Pacific Northwest Region. Irregular pagination.

USDA Forest Service. 1993b. National hierarchical framework of ecological units. Washington, DC: USDA Forest Service. 21 p.

USDA Forest Service. 1994. The Forest Service ethics and course to the future. FS 567. Washington, DC: USDA Forest Service. 9 p.

USDA Forest Service. 1996. Status of the interior Columbia basin: summary of scientific findings. Gen. Tech. Rep. PNW-GTR-385. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 144 p. http://www.treesearch.fs.fed.us/pubs/25385

USDA Forest Service. 1997. Considering all things: summary of the draft environmental impact statements. R6-P&EA-UP-007-97. [Place of publication unknown]: USDA Forest Service; USDI Bureau of Land Management. 57 p. https://archive.org/download/consideringallth00unit/consideringallth00unit.pdf

USDA Forest Service. 1999. Desolation ecosystem analysis. Pendleton, OR: USDA Forest Service, Pacific Northwest Region, Umatilla National Forest. 206 p. http://hdl.handle.net/1794/6649

USDA Forest Service. 2012a. 36 CFR Part 219. National forest system land management planning. Federal Register. 77(68): 21162-21276.

USDA Forest Service. 2012b. Future of America's forest and rangelands: Forest Service 2010 Resources Planning Act assessment. Washington, DC: USDA Forest Service, Washington Office. 198 p. http://www.treesearch.fs.fed.us/pubs/41976

USDA Forest Service. 2012c. Increasing the pace of restoration and job creation on our national forests. Washington, DC: USDA Forest Service. 8 p. http://www.fs.fed.us/publications/restoration/restoration.pdf

USDA Forest Service. 2015. Baseline estimates of carbon stocks in forests and harvested wood products for National Forest System units; Pacific Northwest Region. Washington, DC: USDA Forest Service, Office of the Chief, Climate Change Advisor's Office. 48 p. PacificNorthwestRegionCarbonAssessment

USDA Forest Service. 2015a. From accelerating restoration to creating and maintaining resilient landscapes and communities across the nation: Update on progress from 2012. FS-1069. Washington, DC: USDA Forest Service. 26 p. accelerating-restoration-update-2015

USDA Forest Service. 2015b. The rising cost of wildfire operations: Effects on the Forest Service's non-fire work. Washington, DC: USDA Forest Service. 16 p. http://tinyurl.com/zt4elp8

USDA Forest Service. 2018a. Potential drought impacts in the Pacific Northwest. Washington, DC: USDA Forest Service, Washington Office, Office of Sustainability and Climate. 12 p. https://www.fs.fed.us/sites/default/files/r6-droughtfactsheet.pdf

USDA Forest Service. 2018b. Toward shared stewardship across landscapes: an outcome-based investment strategy. FS-118. Washington, DC: USDA Forest Service. 24 p. https://www.fs.fed.us/sites/default/files/toward-shared-stewardship.pdf

Vale, T.R. 1988. Clearcut logging, vegetation dynamics, and human wisdom. Geographical Review. 78(4): 375-386. doi:10.2307/215089

Vale, T.R., ed. 2002. Fire, native peoples, and the natural landscape. Washington, DC: Island Press. 315 p. isbn:1-55963-889-3

van der Kamp, B.J. 1991. Pathogens as agents of diversity in forested landscapes. Forestry Chronicle. 67(4): 353-354. doi:10.5558/tfc67353-4

van der Plas, F.; Manning, P.; Soliveres, S.; Allan, E.; Scherer-Lorenzen, M.; Verheyen, K.; Wirth, C.; Zavala, M.A.; Ampoorter, E.; Baeten, L.; Barbaro, L.; Bauhus, J.; Benavides, R.; Benneter, A.; Bonal, D.; Bouriaud, O.; Bruelheide, H.; Bussotti, F.; Carnol, M.; Castagneyrol, B.; Charbonnier, Y.; Coomes, D.A.; Coppi, A.; Bastias, C.C.; Dawud, S.M.; De Wandeler, H.; Domisch, T.; Finér, L.; Gessler, A.; Granier, A.; Grossiord, C.; Guyot, V.; Hättenschwiler, S.; Jactel, H.; Jaroszewicz, B.; Joly, F.-x.; Jucker, T.; Koricheva, J.; Milligan, H.; Mueller, S.; Muys, B.; Nguyen, D.; Pollastrini, M.; Ratcliffe, S.; Raulund-Rasmussen, K.; Selvi, F.; Stenlid, J.; Valladares, F.; Vesterdal, L.; Zielínski, D.; Fischer, M. 2016. Biotic homogenization can decrease landscape-scale forest multifunctionality. Proceedings of the National Academy of Sciences. 113(13): 3557-3562. doi:10.1073/pnas.1517903113

van der Putten, W.H. 2000. Pathogen-driven forest diversity. Nature. 404(6775): 232-233. doi:10.1038/35005188

van der Werf, G.R.; Randerson, J.T.; Giglio, L.; Collatz, G.J.; Kasibhatla, P.S.; Arellano, A.F., Jr. 2006. Interannual variability in global biomass burning emissions from 1997 to 2004. Atmospheric Chemistry and Physics. 6(11): 3423-3441. doi:10.5194/acp-6-3423-2006

Van Gunst, K.J.; Weisberg, P.J.; Yang, J.; Fan, Y. 2016. Do denser forests have greater risk of tree mortality: A remote sensing analysis of density-dependent forest mortality. Forest Ecology and Management. 359: 19-32. doi:10.1016/j.foreco.2015.09.032

van Mantgem, P.J.; Stephenson, N.L. 2007. Apparent climatically induced increase of tree mortality rates in a temperate forest. Ecology Letters. 10(10): 909-916.

doi:10.1111/j.1461-0248.2007.01080.x

van Mantgem, P.J.; Stephenson, N.L.; Byrne, J.C.; Daniels, L.D.; Franklin, J.F.; Fulé, P.Z.; Harmon, M.E.; Larson, A.J.; Smith, J.M.; Taylor, A.H.; Veblen, T.T. 2009. Widespread increase of tree mortality rates in the western United States. Science. 323(5913): 521-524. doi:10.1126/science.1165000

van Mantgem, P.J.; Nesmith, J.C.B.; Keifer, M.; Knapp, E.E.; Flint, A.; Flint, L. 2013. Climatic stress increases forest fire severity across the western United States. Ecology Letters. 16(9): 1151-1156. doi:10.1111/ele.12151

Van Wagner, C.E. 1977. Conditions for the start and spread of crown fire. Canadian Journal of Forest Research. 7(1): 23-34. doi:10.1139/x77-004

Van Wagner, C.E. 1978. Age-class distribution and the forest fire cycle. Canadian Journal of Forest Research. 8(2): 220-227. doi:10.1139/x78-034

Van Wagner, C.E. 1985. Does nature really care who starts the fire? In: Lotan, J.E.; Kilgore, B.M.; Fischer, W.C.; Mutch, R.W., tech. coords. Proceedings–symposium and workshop on wilderness fire. Gen. Tech. Rep. INT-182. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station: 98-100. https://archive.org/download/CAT31118982/CAT31118982.pdf

Vance, E.D. 2018. Conclusions and caveats from studies of managed forest carbon budgets. Forest Ecology and Management. 427: 350-354. doi:10.1016/j.foreco.2018.06.021

Vandygriff, J.C.; Hansen, E.M.; Bentz, B.J.; Allen, K.K.; Amman, G.D.; Rasmussen, L.A. 2015. Long-term efficacy of diameter-limit cutting to reduce mountain pine beetle-caused tree mortality in a lodgepole pine forest. Forestry Chronicle. 91(4): 444-456. doi:10.5558/tfc2015-074

Vasechko, G.I. 1983. An ecological approach to forest protection. Forest Ecology and Management. 5(2): 133-168. doi:10.1016/0378-1127(83)90063-4

Vavra, M.; Parks, C.G.; Wisdom, M.J. 2007. Biodiversity, exotic plant species, and herbivory: the good, the bad, and the ungulate. Forest Ecology and Management. 246(1): 66-72. doi:10.1016/j.foreco.2007.03.051

Veblen, T.T. 1986. Treefalls and the coexistence of conifers in subalpine forests of the central Rockies. Ecology. 67(3): 644-649. doi:10.2307/1937688

Veblen, T.T. 2003. Historic range of variability of mountain forest ecosystems: concepts and applications. Forestry Chronicle. 79(2): 223-226. doi:10.5558/tfc79223-2

Veblen, T.T.; Lorenz, D.C. 1986. Anthropogenic disturbance and recovery patterns in montane forests, Colorado Front Range. Physical Geography. 7(1): 1-24. doi:10.1080/02723646.1986.10642278

Veblen, T.T.; Hadley, K.S.; Reid, M.S.; Rebertus, A.J. 1989. Blowdown and stand development in a Colorado subalpine forest. Canadian Journal of Forest Research. 19(10): 1218-1225. doi:10.1139/x89-186

Veblen, T.T.; Hadley, K.S.; Reid, M.S.; Rebertus, A.J. 1991a. Methods of detecting past spruce beetle outbreaks in Rocky Mountain subalpine forests. Canadian Journal of Forest Research. 21(2): 242-254. doi:10.1139/x91-030

Veblen, T.T.; Hadley, K.S.; Reid, M.S.; Rebertus, A.J. 1991b. The response of subalpine forests to spruce beetle outbreak in Colorado. Ecology. 72(1): 213-231. doi:10.2307/1938916

Veblen, T.T.; Hadley, K.S.; Nel, E.M.; Kitzberger, T.; Reid, M.; Villalba, R. 1994. Disturbance regime and disturbance interactions in a Rocky Mountain subalpine forest. Journal of Ecology. 82(1): 125-135. doi:10.2307/2261392

Vera, F.W.M. 2004. Grazing ecology and forest history. Cambridge, MA: CABI Publishing. 506 p. isbn:0-85199-442-3

Verkaar, H.J. 1988. Are defoliators beneficial for their host plants in terrestrial ecosystems–

a review. Acta Botanica Neerlandica. 37(2): 137-152. doi:10.1111/j.1438-8677.1988.tb02122.x

Vicente-Serrano, S.M.; Beguería, S.; López-Moreno, J.I. 2010. A multiscalar drought index sensitive to global warming: The standardized precipitation evapotranspiration index. Journal of Climate. 23(7): 1696-1718. doi:10.1175/2009jcli2909.1

Vieira, D.C.S.; Fernández, C.; Vega, J.A.; Keizer, J.J. 2015. Does soil burn severity affect the post-fire runoff and interrill erosion response? A review based on meta-analysis of field rainfall simulation data. Journal of Hydrology. 523: 452-464. doi:10.1016/j.jhydrol.2015.01.071

Visser, S.; Parkinson, D. 1999. Wildfire vs. clearcutting: impacts on ectomycorrhizal and decomposer fungi. In: Meurisse, R.T.; Ypsilantis, W.G.; Seybold, C., tech. eds. Proceedings: Pacific Northwest forest and rangeland soil organism symposium. Gen. Tech. Rep. PNW-GTR-461. Portland, OR: USDA Forest Service, Pacific Northwest Research Station: 114-123. https://www.fs.usda.gov/treesearch/pubs/7603

Vitousek, P.M.; Mooney, H.A.; Lubchenco, J.; Melillo, J.M. 1997. Human domination of earth’s ecosystems. Science. 277(5325): 494-499. doi:10.1126/science.277.5325.494

Voller, J.; Harrison, S., eds. 1998. Conservation biology principles for forested landscapes. Vancouver, BC: UBC Press. 243 p. isbn:0-7748-0630-3

Volney, W.J.A.; Hirsch, K.G. 2005. Disturbing forest disturbances. Forestry Chronicle. 81(5): 662-668. doi:10.5558/tfc81662-5

Vose, J.M.; Peterson, D.L.; Patel-Weynand, T. 2012. Effects of climatic variability and change on forest ecosystems: a comprehensive science synthesis for the U.S. forest sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 265 p. http://www.treesearch.fs.fed.us/pubs/42610

Vose, J.M.; Miniat, C.F.; Luce, C.H.; Asbjornsen, H.; Caldwell, P.V.; Campbell, J.L.; Grant, G.E.; Isaak, D.J.; Loheide Ii, S.P.; Sun, G. 2016. Ecohydrological implications of drought for forests in the United States. Forest Ecology and Management. 380: 335-345. doi:10.1016/j.foreco.2016.03.025

Vose, J.M.; Peterson, D.L.; Luce, C.H.; Patel-Weynand, T. 2019. Effects of drought on forests and rangelands in the United States: translating science into management responses. Washington, DC: USDA Forest Service, Washington Office. 227 p. https://www.fs.usda.gov/treesearch/pubs/59158

Wadleigh, L.; Jenkins, M.J. 1996. Fire frequency and the vegetative mosaic of a spruce-fir forest in northern Utah. Great Basin Naturalist. 56(1): 28-37. https://www.jstor.org/stable/41712923

Wagner, M.R.; Clancy, K.M.; Tinus, R.W. 1989. Maturational variation in needle essential oils from *Pseudotsuga menziesii*, *Abies concolor* and *Picea engelmannii*. Phytochemistry. 28(3): 765-770. doi:10.1016/0031-9422(89)80111-6

Waide, R.B.; Willig, M.R.; Steiner, C.F.; Mittelbach, G.; Gough, L.; Dodson, S.I.; Juday, G.P.; Parmenter, R. 1999. The relationship between productivity and species richness. Annual Review of Ecology and Systematics. 30: 257-300. doi:10.1146/annurev.ecolsys.30.1.257

Waldron, K.; Ruel, J.-C.; Gauthier, S.; Goulet, P. 2014. Comparisons of spatial patterns between windthrow and logging at two spatial scales. Canadian Journal of Forest Research. 44(7): 740-749. doi:10.1139/cjfr-2013-0402

Wallin, D.O.; Swanson, F.J.; Marks, B.; Cissel, J.H.; Kertis, J. 1996. Comparison of managed and pre-settlement landscape dynamics in forests of the Pacific Northwest, USA. Forest Ecology and Management. 85(1-3): 291-309. doi:10.1016/S0378-1127(96)03765-6

Walstad, J.D.; Radosevich, S.R.; Sandberg, D.V., eds. 1990. Natural and prescribed fire in Pacific Northwest forests. Corvallis, OR: Oregon State University Press. 317 p. isbn:0-87071-359-0

Walter, J.A.; Platt, R.V. 2013. Multi-temporal analysis reveals that predictors of mountain pine beetle infestation change during outbreak cycles. Forest Ecology and Management. 302: 308-318. doi:10.1016/j.foreco.2013.03.038

Walters, C.J. 1986. Adaptive management of renewable resources. New York: Macmillan Publishing Company. isbn:0-02-947970-3

Walther, S.C.; Roering, J.J.; Almond, P.C.; Hughes, M.W. 2009. Long-term biogenic soil mixing and transport in a hilly, loess-mantled landscape: Blue Mountains of southeastern Washington. Catena. 79(2): 170-178. doi:10.1016/j.catena.2009.08.003

Wambolt, C.L. 1973. Conifer water potential as influenced by stand density and environmental factors. Canadian Journal of Botany. 51(12): 2333-2337. doi:10.1139/b73-301

Wang, G.G. 2000. Use of understory vegetation in classifying soil moisture and nutrient regimes. Forest Ecology and Management. 129(1-3): 93-100. doi:10.1016/S0378-1127(99)00142-5

Wang, S. 2002. Wicked problems and metaforestry: Is the era of management over? Forestry Chronicle. 78(4): 505-510. doi:10.5558/tfc78505-4

Wang, Y.; Titus, S.J.; LeMay, V.M. 1998. Relationships between tree slenderness coefficients and tree or stand characteristics for major species in boreal mixedwood forests. Canadian Journal of Forest Research. 28(8): 1171-1183. doi:10.1139/x98-092

Wang, Y.; Wei, X.; del Campo, A.D.; Winkler, R.; Wu, J.; Li, Q.; Liu, W. 2019. Juvenile thinning can effectively mitigate the effects of drought on tree growth and water consumption in a young *Pinus contorta* stand in the interior of British Columbia, Canada. Forest Ecology and Management. 454: 117667. doi:10.1016/j.foreco.2019.117667

Wanner, J.L.; Tinnin, R.O. 1989. Some effects of infection by *Arceuthobium americanum* on the population dynamics of *Pinus contorta* in Oregon. Canadian Journal of Forest Research. 19(6): 736-742. doi:10.1139/x89-113

Waples, R.S.; Beechie, T.; Pess, G.R. 2009. Evolutionary history, habitat disturbance regimes, and anthropogenic changes: What do these mean for resilience of Pacific salmon populations? Ecology and Society. 14(1): 3 (18 p). doi:10.5751/ES-02626-140103

Wardle, D.A.; Nilsson, M-C.; Gallet, C.; Zackrisson, O. 2000. An ecosystem-level perspective of allelopathy. Biological Reviews. 73(3): 305-319. doi:10.1111/j.1469-185X.1998.tb00033.x

Ware, G.W. 1989. The pesticide book. 3rd edition. Fresno, CA: Thomson Publications. 336 p. isbn:0-913702-52-8

Wargo, P.M. 1981. Defoliation and secondary-action organism attack: with emphasis on *Armillaria mellea*. Journal of Arboriculture 7(3): 64-69. http://joa.isa-arbor.com/articles.asp?JournalID=1&VolumeID=7&IssueID=3

Wargo, P.M. 1995. Disturbance in forest ecosystems caused by pathogens and insects. In: Eskew, L.G., comp. Forest health through silviculture: proceedings of the 1995 National Silviculture Workshop. Gen. Tech. Rep. RM-GTR-267. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station: 20-25. https://www.fs.usda.gov/treesearch/pubs/22037

Waring, R.H. 1983. Estimating forest growth and efficiency in relation to canopy leaf area. Advances in Ecological Research. 13: 327-354. doi:10.1016/S0065-2504(08)60111-7

Waring, R.H. 1985. Imbalanced forest ecosystems: assessments and consequences. Forest Ecology and Management. 12(2): 93-112. doi:10.1016/0378-1127(85)90078-7

Waring, R.H. 1987. Characteristics of trees predisposed to die. BioScience. 37(8): 569-574. doi:10.2307/1310667

Waring, K.M.; O'Hara, K.L. 2005. Silvicultural strategies in forest ecosystems affected by introduced pests. Forest Ecology and Management. 209(1-2): 27-41.

doi:10.1016/j.foreco.2005.01.008

Waring, R.H.; Pitman, G.B. 1983. Physiological stress in lodgepole pine as a precursor for mountain pine beetle attack. Journal of Applied Entomology. 96(1-5): 265-270. doi:10.1111/j.1439-0418.1983.tb03668.x

Waring, R.H.; Pitman, G.B. 1985. Modifying lodgepole pine stands to change susceptibility to mountain pine beetle attack. Ecology. 66(3): 889-897. doi:10.2307/1940551

Waring, R.H.; Running, S.W. 1998. Forest ecosystems: analysis at multiple scales. 2nd edition. San Diego, CA: Academic Press. 370 p. isbn:0-12-735443-3

Waring, R.H.; Thies, W.G.; Muscato, D. 1980. Stem growth per unit of leaf area: a measure of tree vigor. Forest Science. 26(1): 112-117. doi:10.1093/forestscience/26.1.112

Waring, R.H.; Cromack, K.; Matson, P.A., Jr.; Boone, R.D.; Stafford, S.G. 1987. Responses to pathogen-induced disturbance: decomposition, nutrient availability, and tree vigour. Forestry. 60(2): 219-227. doi:10.1093/forestry/60.2.219

Warren, W.A. 2007. What is a healthy forest?: definitions, rationales, and the lifeworld. Society and Natural Resources. 20(2): 99-117. doi:10.1080/08941920600901890

Waters, W.E. 1978. Integrating dwarf mistletoe control with forest management. In: Scharpf, R.F.; Parmeter, Jr., J.R., tech. coords. Proceedings of the symposium on dwarf mistletoe control through forest management. Berkeley, CA: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station: 156-161. https://www.fs.usda.gov/treesearch/pubs/24083

Waters, W.E.; Stark, R.W. 1980. Forest pest management: concept and reality. Annual Review of Entomology. 25(1): 479-509. doi:10.1146/annurev.en.25.010180.002403

Watrud, E.; Zensen, F.; Darbyshire, R. 2012. Laws affecting reforestation on USDA Forest Service lands. Tree Planters' Notes. 55(2): 39-42. http://www.rngr.net/publications/tpn/55-2/laws-affecting-reforestation-on-usda-forest-service-lands

Watt, A.S. 1947. Pattern and process in the plant community. Journal of Ecology. 35(1/2): 1-22. doi:10.2307/2256497

Wayman, R.B.; North, M. 2007. Initial response of a mixed-conifer understory plant community to burning and thinning restoration treatments. Forest Ecology and Management. 239(1-3): 32-44. doi:10.1016/j.foreco.2006.11.011

Weaver, T.; Gustafson, D.; Lichthardt, J. 2001. Exotic plants in early and late seral vegetation of fifteen northern Rocky Mountain environments (HTs). Western North American Naturalist. 61(4): 417-427. https://scholarsarchive.byu.edu/wnan/vol61/iss4/5

Webb, W.L.; Lauenroth, W.K.; Szarek, S.R.; Kinerson, R.S. 1983. Primary production and abiotic controls in forests, grasslands, and desert ecosystems in the United States. Ecology. 64(1): 134-151. doi:10.2307/1937336

Wei, Y. 2012. Optimize landscape fuel treatment locations to create control opportunities for future fires. Canadian Journal of Forest Research. 42(6): 1002-1014. doi:10.1139/x2012-051

Wei, X.; Liu, W.; Waterhouse, J.; Armleder, M. 2000. Simulations on impacts of different management strategies on long-term site productivity in lodgepole pine forests of the central interior of British Columbia. Forest Ecology and Management. 133(3): 217-229. doi:10.1016/S0378-1127(99)00234-0

Weigand, J.F.; Haynes, R.W.; Tiedemann, A.R.; Riggs, R.A.; Quigley, T.M. 1993. Economic assessment of ungulate herbivory in commercial forests of eastern Oregon and Washington, USA. Forest Ecology and Management. 61(1-2): 137-155. doi:10.1016/0378-1127(93)90195-S

Weir, J.R. 1916. Mistletoe injury to conifers in the Northwest. Bull. No. 360. Washington, DC: U.S. Department of Agriculture. 39 p.

https://archive.org/download/mistletoeinjuryt360weir/mistletoeinjuryt360weir.pdf

Weir, J.R.; Hubert, E.E. 1919. The influence of thinning on western hemlock and grand fir infected with Echinodontium tinctorium. Journal of Forestry. 17(1): 21-35. doi:10.1093/jof/17.1.21

Welch, K.R.; Safford, H.D.; Young, T.P. 2016. Predicting conifer establishment post wildfire in mixed conifer forests of the North American Mediterranean-climate zone. Ecosphere. 7(12): e01609 (29 p). doi:10.1002/ecs2.1609

Wellner, C.A. 1970. Fire history in the northern Rocky Mountains. In: The role of fire in the intermountain west; proceedings. Missoula, MT: University of Montana, School of Forestry: 42-64.

Wells, A.F. 2006. Deep canyon and subalpine riparian and wetland plant associations of the Malheur, Umatilla, and Wallowa-Whitman National Forests. Gen. Tech. Rep. GTR-PNW-682. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 277 p. http://www.treesearch.fs.fed.us/pubs/24936

West, J.; Julius, S.; Kareiva, P.; Enquist, C.; Lawler, J.; Petersen, B.; Johnson, A.; Shaw, M. 2009. U.S. natural resources and climate change: concepts and approaches for management adaptation. Environmental Management. 44(6): 1001-1021. doi:10.1007/s00267-009-9345-1

Westerling, A.L.; Bryant, B.P. 2008. Climate change and wildfire in California. Climatic Change. 87(1): 231-249. doi:10.1007/s10584-007-9363-z

Westerling, A.L.; Gershunov, A.; Brown, T.J.; Cayan, D.R.; Dettinger, M.D. 2003. Climate and wildfire in the western United States. Bulletin of the American Meteorological Society. 84(5): 595-604. doi:10.1175/BAMS-84-5-595

Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; Swetnam, T.W. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science. 313(5789): 940-943. doi:10.1126/science.1128834

Westoby, M. 1984. The self-thinning rule. Advances in Ecological Research. 14: 167-225. doi:10.1016/S0065-2504(08)60171-3

Westveld, M. 1951. Vegetation mapping as a guide to better silviculture. Ecology. 32(3): 508-517. doi:10.2307/1931727

White, P.S. 1979. Pattern, process, and natural disturbance in vegetation. Botanical Review. 45(3): 229-299. doi:10.1007/BF02860857

White, P.S. 1987. Natural disturbance, patch dynamics, and landscape pattern in natural areas. Natural Areas Journal. 7(1): 14-22. https://www.jstor.org/stable/43910915

White, P.S.; Jentsch, A. 2001. The search for generality in studies of disturbance and ecosystem dynamics. Progress in Botany. 62: 399-449. doi:10.1007/978-3-642-56849-7_17

White, M.A.; Mladenoff, D.J. 1994. Old-growth forest landscape transitions from pre-European settlement to present. Landscape Ecology. 9(3): 191-205. doi:10.1007/BF00134747

White, D.E.; Atzet, T.; Greenup, M. 1992. Predicting planting success and natural regeneration potential using plant associations. Grants Pass, OR: USDA Forest Service, Pacific Northwest Region, Siskiyou National Forest. 31 p.

White, P.S.; Harrod, J.; Romme, W.H.; Betancourt, J. 1999. Disturbance and temporal dynamics. In: Szaro, R.C.; Johnson, N.C.; Sexton, W.T.; Malk, A.J., eds. Ecological stewardship: a common reference for ecosystem management. Volume II. Oxford, UK: Elsevier Science, Ltd: 281-312. isbn:0-08-043206-9

White, K.P.; Coleman, M.; Page-Dumroese, D.S.; Gessler, P.E.; Kimsey, M.; Shaw, T. 2012. Examining soil parent material influence over Douglas-fir stem growth response to fertilization: Taking advantage of information from spatiotemporally distributed experiments. Forest Ecology and Management. 286: 101-107. doi:10.1016/j.foreco.2012.08.043

White, R.M.; Young, J.; Marzano, M.; Leahy, S. 2018. Prioritising stakeholder engagement for forest health, across spatial, temporal and governance scales, in an era of austerity. Forest Ecology and Management. 417: 313-322. doi:10.1016/j.foreco.2018.01.050

White, E.M.; Lindberg, K.; Davis, E.J.; Spies, T.A. 2019. Use of science and modeling by practitioners in landscape-scale management decisions. Journal of Forestry. 117(3): 267-279. doi:10.1093/jofore/fvz007

Whitehead, R.J.; Russo, G.L. 2005. "Beetle-proofed" lodgepole pine stands in interior British Columbia have less damage from mountain pine beetle. Infor. Rep. BC-X-402. Victoria, BC: Natural Resources Canada, Canadian Forest Service, Pacific Forestry Centre. 17 p. http://bookstore.cfs.nrcan.gc.ca/detail_e.php?recid=12584702

Whitehead, R.J.; Safranyik, L.; Russo, G.L.; Shore, T.L.; Carroll, A.L. 2004. Silviculture to reduce landscape and stand susceptibility to the mountain pine beetle. In: Shore, T.L.; Brooks, J.E.; Stone, J.E., eds. Mountain pine beetle symposium: challenges and solutions. Infor. Rep. BC-X-399. Victoria, BC: Natural Resources Canada, Canadian Forest Service, Pacific Forestry Centre: 233-244. https://cfs.nrcan.gc.ca/pubwarehouse/pdfs/25159.pdf

Whiteside, J.M. 1956. Spruce budworm control in Oregon and Washington, 1949-1956. Unnum. Rep. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station, Division of Forest Insect Research. 23 p. https://archive.org/download/CAT10507164/CAT10507164.pdf

Whitlock, C. 1992. Vegetational and climatic history of the Pacific Northwest during the last 20,000 years: implications for understanding present-day biodiversity. Northwest Environmental Journal. 8: 5-28. Whitlock 1992

Whitlock, C.; Knox, M.A. 2002. Prehistoric burning in the Pacific Northwest: human versus climatic influences. Chapter 6. In: Vale, T.R., ed. Fire, native peoples, and the natural landscape. Washington, DC: Island Press: 195-231. isbn:1-55963-889-3

Whitlock, C.; Shafer, S.L.; Marlon, J. 2003. The role of climate and vegetation change in shaping past and future fire regimes in the northwestern US and the implications for ecosystem management. Forest Ecology and Management. 178(1-2): 5-21. doi:10.1016/S0378-1127(03)00051-3

Whitlock, C.; Higuera, P.E.; McWethy, D.B.; Briles, C.E. 2010. Paleoecological perspectives on fire ecology: revisiting the fire-regime concept. Open Ecology Journal. 3(Special Issue 1): 6-23. doi:10.2174/1874213001003020006

Whittaker, R.H. 1953. A consideration of climax theory: the climax as a population and pattern. Ecological Monographs. 23(1): 41-78. doi:10.2307/1943519

Whittaker, J.B. 2001. Insects and plants in a changing atmosphere. Journal of Ecology. 89(4): 507-518. doi:10.1046/j.0022-0477.2001.00582.x

Whittaker, R.J.; Willis, K.J.; Field, R. 2001. Scale and species richness: towards a general, hierarchical theory of species diversity. Journal of Biogeography. 28(4): 453-470. doi:10.1046/j.1365-2699.2001.00563.x

Wicker, E.F.; Leaphart, C.D. 1976. Fire and dwarf mistletoe *(Arceuthobium* spp.) relationships in the northern Rocky Mountains. In: Proceedings of the annual Tall Timbers fire ecology conference number 14. Tallahassee, FL: Tall Timbers Research Station: 279-298. https://talltimbers.org/wp-content/uploads/2018/09/279-WickerandLeaphart1974_op.pdf

Wicklum, D.; Davies, R.W. 1995. Ecosystem health and integrity? Canadian Journal of Botany. 73(7): 997-1000. doi:10.1139/b95-108

Wickman, B.E. 1963. Mortality and growth reduction of white fir following defoliation by the Douglas-fir tussock moth. Res. Pap. PSW-7. Berkeley, CA: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station. 15 p. https://www.fs.usda.gov/treesearch/pubs/28741

Wickman, B. 1977. Observations on spider predation of early instar larvae of Douglas fir tussock moth, *Orgyia pseudotsugata* (McDunnough) (Lepidoptera: Lymantriidae). Pan-Pacific Entomologist. 53: 46.

Wickman, B.E. 1978a. A case study of a Douglas-fir tussock moth outbreak and stand conditions 10 years later. Res. Pap. PNW-244. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 22 p. A_Case_Study_of_a_Douglas_fir_Tussock_Mo

Wickman, B.E. 1978b. Tree mortality and top-kill related to defoliation by the Douglas-fir tussock moth in the Blue Mountains outbreak. Res. Pap. PNW-233. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 47 p. https://archive.org/download/CAT92273162/CAT92273162.pdf

Wickman, B.E. 1980. Increased growth of white fir after Douglas-fir tussock moth outbreak. Journal of Forestry. 78(1): 31-33. doi:10.1093/jof/78.1.31

Wickman, B.E. 1986. Growth of white fir after Douglas-fir tussock moth outbreaks: long-term records in the Sierra Nevada. Res. Note PNW-440. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 8 p. https://www.fs.usda.gov/treesearch/pubs/25059

Wickman, B.E. 1986. Radial growth of grand fir and Douglas-fir 10 years after defoliation by the Douglas-fir tussock moth in the Blue Mountains outbreak. Res. Pap. PNW-367. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 11 p. https://www.fs.usda.gov/treesearch/pubs/8985

Wickman, B.E. 1988. Tree growth in thinned and unthinned white fir stands 20 years after a Douglas-fir tussock moth outbreak. Res. Note PNW-RN-477. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 11 p. https://www.fs.usda.gov/treesearch/pubs/25087

Wickman, B.E. 1990. Old-growth forests and history of insect outbreaks. Northwest Environmental Journal. 6(2): 401-403.

Wickman, B.E. 1992. Forest health in the Blue Mountains: the influence of insects and disease. Gen. Tech. Rep. PNW-GTR-295. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 15 p. http://www.treesearch.fs.fed.us/pubs/9032

Wickman, B.E.; Eaton, C.B. 1962. The effects of sanitation-salvage cutting on insect-caused mortality at Blacks Mountain Experimental Forest, 1938-1959. Berkeley, CA: USDA Forest Service, Pacific Southwest Forest and Range Experiment Station. 39 p. https://archive.org/download/effectsofsanitat66wick/effectsofsanitat66wick.pdf

Wickman, B.E.; Mason, R.R.; Thompson, C.G. 1973. Major outbreaks of the Douglas-fir tussock moth in Oregon and California. Gen. Tech. Rep. PNW-5. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 18 p. https://www.fs.usda.gov/treesearch/pubs/22552

Wickman, B.E.; Henshaw, D.L.; Gollob, S.K. 1980. Radial growth in grand fir and Douglas-fir related to defoliation by the Douglas-fir tussock moth in the Blue Mountains outbreak. Res. Pap. PNW-269. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 23 p. https://www.fs.usda.gov/treesearch/pubs/21251

Wickman, B.E.; Mason, R.R.; Trostle, G.C. 1981. Douglas-fir tussock moth. Forest Insect & Disease Leaflet 86. [Place of publication unknown]: USDA Forest Service. 10 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043384.pdf

Wickman, B.E.; Mason, R.R.; Paul, H.G. 1992. Thinning and nitrogen fertilization in a grand fir stand infested with western spruce budworm; part II, tree growth response. Forest Science. 38(2): 252-264. doi:10.1093/forestscience/38.2.252

Wickman, B.E.; Mason, R.R.; Swetnam, T.W. 1994. Searching for long-term patterns of forest insect outbreaks. In: Leather, S.R.; Walters, K.F.A. Mills, N.J.; Watt, A.D., eds. Individuals, populations and patterns in ecology. Andover, Hampshire, UK: Intercept, Ltd: 251-

261. isbn:0-946707-64-2

Widen, M.J.; Petras O'Neil, M.A.; Dickinson, Y.L.; Webster, C.R. 2018. Rubus persistence within silvicultural openings and its impact on regeneration: The influence of opening size and advance regeneration. Forest Ecology and Management. 427: 162-168. doi:10.1016/j.foreco.2018.05.049

Widenfalk, O.; Weslien, J. 2009. Plant species richness in managed boreal forests–Effects of stand succession and thinning. Forest Ecology and Management. 257(5): 1386-1394. doi:10.1016/j.foreco.2008.12.010

Wiedinmyer, C.; Neff, J.C. 2007. Estimates of CO_2 from fires in the United States: implications for carbon management. Carbon Balance and Management. 2(10): 1-12. doi:10.1186/1750-0680-2-10

Wiens, J.A. 2013. Is landscape sustainability a useful concept in a changing world? Landscape Ecology. 28(6): 1047-1052. doi:10.1007/s10980-012-9801-9

Wiersum, K.F. 1995. 200 years of sustainability in forestry: lessons from history. Environmental Management. 19(3): 321-329. doi:10.1007/BF02471975

Wilkes, C. 1844. Narrative of the United States exploring expedition during the years 1838, 1839, 1840, 1841, 1842. Volume V. Philadelphia, PA: C. Sherman. 591 p.

Williams, C.B., Jr. 1967. Spruce budworm damage symptoms related to radial growth of grand fir, Douglas-fir and Engelmann spruce. Forest Science. 13(3): 274-285. doi:10.1093/forestscience/13.3.274

Williams, J.T. 1978. Management implications from a timber type evaluation of Oregon and Washington tussock moth outbreak areas. M.S. thesis. Seattle, WA: University of Washington. 69 p.

Williams, G.W. 2000. Early fire use in Oregon. Fire Management Today. 60(3): 13-20. https://www.fs.fed.us/file/40943/download?token=E2qLLR6G

Williams, J. 2013. Exploring the onset of high-impact mega-fires through a forest land management prism. Forest Ecology and Management. 294: 4-10. doi:10.1016/j.foreco.2012.06.030

Williams, D.W.; Birdsey, R.A. 2003. Historical patterns of spruce budworm defoliation and bark beetle outbreaks in North American conifer forests: an atlas and description of digital maps. Gen. Tech. Rep. NE-308. Newtown Square, PA: USDA Forest Service, Northeastern Research Station. 33 p. https://www.fs.usda.gov/treesearch/pubs/5521

Williams, D.W.; Liebhold, A.M. 1995. Herbivorous insects and global change: potential changes in the spatial distribution of forest defoliator outbreaks. Journal of Biogeography. 22(4/5): 665-671. doi:10.2307/2845968

Williams, D.W.; Liebhold, A.M. 2002. Climate change and the outbreak ranges of two North American bark beetles. Agricultural and Forest Entomology. 4(2): 87-99. doi:10.1046/j.1461-9563.2002.00124.x

Williams, N.G.; Powers, M.D. 2019. Carbon storage implications of active management in mature Pseudotsuga menziesii forests of western Oregon. Forest Ecology and Management. 432: 761-775. doi:10.1016/j.foreco.2018.10.002

Williams, J.T.; Martin, R.E.; Pickford, S.G. 1980. Silvicultural and fire management implications from a timber type evaluation of tussock moth outbreak areas. In: Martin, R.E.; Edmonds, R.L.; Faulkner, D.A.; Harrington, J.B.; Fuquay, D.M.; Stocks, B.J.; Barr, S., eds. Proceedings: sixth conference on fire and forest meteorology. Washington, DC: Society of American Foresters: 191-196.

Williams, R.E.; Shaw, C.G., III; Wargo, P.M.; Sites, W.H. 1986. Armillaria root disease. Forest Insect & Disease Leaflet 78. [Place of publication unknown]: USDA Forest Service. 8 p. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev2_043192.pdf

Williams, C.K.; Kelley, B.F.; Smith, B.G.; Lillybridge, T.R. 1995. Forested plant associations of the Colville National Forest. Gen. Tech. Rep. PNW-GTR-360. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 375 p. https://www.fs.usda.gov/treesearch/pubs/26243

Williams, A.P.; Allen, C.D.; Macalady, A.K.; Griffin, D.; Woodhouse, C.A.; Meko, D.M.; Swetnam, T.W.; Rauscher, S.A.; Seager, R.; Grissino-Mayer, H.D.; Dean, J.S.; Cook, E.R.; Gangodagamage, C.; Cai, M.; McDowell, N.G. 2013. Temperature as a potent driver of regional forest drought stress and tree mortality. Nature Climate Change. 3(3): 292-297. doi:10.1038/nclimate1693

Williams, A.P.; Cook, E.R.; Smerdon, J.E.; Cook, B.I.; Abatzoglou, J.T.; Bolles, K.; Baek, S.H.; Badger, A.M.; Livneh, B. 2020. Large contribution from anthropogenic warming to an emerging North American megadrought. Science. 368(6488): 314-318. doi:10.1126/science.aaz9600

Williamson, N.M. 1999. Crown fuel characteristics, stand structure, and fire hazard in riparian forests of the Blue Mountains, Oregon. M.S. thesis. Seattle, WA: University of Washington, College of Forest Resources. 98 p.

Williamson, N.M.; Agee, J.K. 2002. Heat content variation of interior Pacific Northwest conifer foliage. International Journal of Wildland Fire. 11(1): 91-94. doi:10.1071/WF01046

Williamson, J.R.; Neilsen, W.A. 2000. The influence of forest site on rate and extent of soil compaction and profile disturbance of skid trails during ground-based harvesting. Canadian Journal of Forest Research. 30(8): 1196-1205. doi:10.1139/x00-041

Willis, K.J.; Birks, H.J.B. 2006. What is natural? The need for a long-term perspective in biodiversity conservation. Science. 314(5803): 1261-1265. doi:10.1126/science.1122667

Willis, K.J.; Whittaker, R.J. 2002. Species diversity – scale matters. Science. 295(5558): 1245-1248. doi:10.1126/science.1067335

Wilson, J.S.; Baker, P.J. 1998. Mitigating fire risk to late-successional forest reserves on the east slope of the Washington Cascade Range, USA. Forest Ecology and Management. 110(1-3): 59-75. doi:10.1016/S0378-1127(98)00274-6

Wilson, J.S.; McGaughey, R.J. 2000. Presenting landscape-scale forest information: What is sufficient and what is appropriate? Journal of Forestry. 98(12): 21-27. doi:10.1093/jof/98.12.21

Wilson, K.; Morren, G.E.B., Jr. 1990. Systems approaches for improvement in agriculture and resource management. New York: Macmillan Publishing Company. 361 p. isbn:0-02-428411-4

Wilson, J.S.; Isaac, E.S.; Gara, R.I. 1998. Impacts of mountain pine beetle *(Dendroctonus ponderosae)* (Col., Scolytidae) infestation on future landscape susceptibility to the western spruce budworm *(Choristoneura occidentalis)* (Lep., Tortricidae) in north central Washington. Journal of Applied Entomology. 122(1-5): 239-245. doi:10.1111/j.1439-0418.1998.tb01490.x

Wilson, C.; Kampf, S.K.; Wagenbrenner, J.W.; MacDonald, L.H. 2018. Rainfall thresholds for post-fire runoff and sediment delivery from plot to watershed scales. Forest Ecology and Management. 430: 346-356. doi:10.1016/j.foreco.2018.08.025

Wimberly, M.C.; Kennedy, R.S.H. 2008. Spatially explicit modeling of mixed-severity fire regimes and landscape dynamics. Forest Ecology and Management. 254(3): 511-523. doi:10.1016/j.foreco.2007.06.044

Wingfield, D.W. 1955. Trees versus water and grass. Journal of Range Management. 8(4): 149-150. doi:10.2307/3894213

Winnett, S.M. 1998. Potential effects of climate change on U.S. forests: a review. Climate Research. 11(1): 39-49. doi:10.3354/cr011039

Wisdom, M.J.; Bate, L.J. 2008. Snag density varies with intensity of timber harvest and human access. Forest Ecology and Management. 255(7): 2085-2093. doi:10.1016/j.foreco.2007.12.027

Wisdom, M.J.; Holthausen, R.S.; Wales, B.C.; Hargis, C.D.; Saab, V.A.; Lee, D.C.; Hann, W.J.; Rich, T.D.; Rowland, M.M.; Murphy, W.J.; Eames, M.R. 2000. Source habitats for terrestrial vertebrates of focus in the interior Columbia basin: broad-scale trends and management implications. Gen. Tech. Rep. PNW-GTR-485. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 529 p. http://www.treesearch.fs.fed.us/pubs/3081

Wisdom, M.J.; Vavra, M.; Boyd, J.M.; Hemstrom, M.A.; Ager, A.A.; Johnson, B.K. 2006. Understanding ungulate herbivory–episodic disturbance effects on vegetation dynamics: knowledge gaps and management needs. Wildlife Society Bulletin. 34(2): 283-292. doi:10.2193/0091-7648(2006)34[283:UUHDEO]2.0.CO;2

Wissmar, R.C. 2004. Riparian corridors of Eastern Oregon and Washington: Functions and sustainability along lowland-arid to mountain gradients. Aquatic Sciences. 66(4): 373-387. doi:10.1007/s00027-004-0720-y

Wissmar, R.C.; Smith, J.E.; McIntosh, B.A.; Li, H.W.; Reeves, G.H.; Sedell, J.R. 1994. Ecological health of river basins in forested regions of eastern Washington and Oregon. Gen. Tech. Rep. PNW-GTR-326. Portland, OR: USDA Forest Service, Pacific Northwest Research Station. 65 p. http://www.treesearch.fs.fed.us/pubs/6371

With, K.A.; Crist, T.O. 1995. Critical thresholds in species' responses to landscape structure. Ecology. 76(8): 2446-2459. doi:10.2307/2265819

Witty, J.H.; Graham, R.C.; Hubbert, K.R. [et al.]. 2003. Contributions of water supply from the weathered bedrock zone to forest soil quality. Geoderma. 114(3-4): 389-400. doi:10.1016/S0016-7061(03)00051-X

Wolfslehner, B.; Seidl, R. 2010. Harnessing ecosystem models and multi-criteria decision analysis for the support of forest management. Environmental Management. 46(6): 850-861. doi:10.1007/s00267-009-9414-5

Wong, C.; Dorner, B.; Sandman, H. 2003. Estimating historical variability of natural disturbances in British Columbia. Land Manage. Handb. No. 53. Victoria, BC: British Columbia Ministry of Forests, Research Branch, British Columbia Ministry of Sustainable Resource Management. 140 p. https://www.for.gov.bc.ca/hfd/pubs/Docs/Lmh/Lmh53.pdf

Wonn, H.T.; O'Hara, K.L. 2001. Height:diameter ratios and stability relationships for four northern Rocky Mountain tree species. Western Journal of Applied Forestry. 16(2): 87-94. doi:10.1093/wjaf/16.2.87

Wood, D.L. 1982. The role of pheromones, kairomones, and allomones in the host selection and colonization behavior of bark beetles. Annual Review of Entomology. 27(1): 411-446. doi:10.1146/annurev.en.27.010182.002211

Woodhouse, C.A.; Overpeck, J.T. 1998. 2000 years of drought variability in the central United States. Bulletin of the American Meteorological Society. 79(12): 2693-2714. doi:10.1175/1520-0477(1998)079<2693:YODVIT>2.0.CO;2

Woodmansee, R.G.; Wallach, L.S. 1981. Effects of fire regimes on biogeochemical cycles. In: Mooney, H.A.; Bonnicksen, T.M.; Christensen, N.L.; Lotan, J.E.; Reiners, W.A., tech. coords. Fire regimes and ecosystem properties; proceedings of the conference. Gen. Tech. Rep. WO-26. [Washington, DC]: USDA Forest Service: 379-400. https://archive.org/download/CAT83781017/CAT83781017.pdf

Woods, S.W.; Balfour, V.N. 2010. The effects of soil texture and ash thickness on the post-fire hydrological response from ash-covered soils. Journal of Hydrology. 393(3): 274-286. doi:10.1016/j.jhydrol.2010.08.025

Woods, A.; Coates, K.D. 2013. Are biotic disturbance agents challenging basic tenets of growth and yield and sustainable forest management? Forestry. 86(5): 543-554. doi:10.1093/forestry/cpt026

Woods, S.W.; Ahl, R.; Sappington, J.; McCaughey, W. 2006. Snow accumulation in thinned lodgepole pine stands, Montana, USA. Forest Ecology and Management. 235(1-3): 202-211. doi:10.1016/j.foreco.2006.08.013

Woodward, S.; Stenlid, J.; Karjalainen, R.; Hüttermann, A., eds. 1998. *Heterobasidion annosum*: biology, ecology, impact, and control. New York: CAB International. 589 p. isbn:0-85199-275-7

Woolley, A.W.; Chabris, C.F.; Pentland, A.; Hashmi, N.; Malone, T.W. 2010. Evidence for a collective intelligence factor in the performance of human groups. Science. 330(6004): 686-688. doi:10.1126/science.1193147

Work, T.T.; Spence, J.R.; Volney, W.J.A.; Burton, P.J. 2003. Sustainable forest management as license to think and to try something different. Chapter 23. In: Burton, P.J.; Messier, C.; Smith, D.W.; Adamowicz, W.L., eds. Towards sustainable management of the boreal forest. Ottawa, ON: NRC Research Press: 953-970. isbn:0-660-18762-0

Worrall, J.J.; Rehfeldt, G.E.; Hamann, A.; Hogg, E.H.; Marchetti, S.B.; Michaelian, M.; Gray, L.K. 2013. Recent declines of *Populus tremuloides* in North America linked to climate. Forest Ecology and Management. 299: 35-51. doi:10.1016/j.foreco.2012.12.033

Worrell, R.; Hampson, A. 1997. The influence of some forest operations on the sustainable management of forest soils – a review. Forestry. 70(1): 61-85. doi:10.1093/forestry/70.1.61

Wright, H.E., Jr. 1974. Landscape development, forest fires, and wilderness management. Science. 186(4163): 487-495. doi:10.1126/science.186.4163.487

Wright, C.S.; Agee, J.K. 2004. Fire and vegetation history in the eastern Cascade Mountains, Washington. Ecological Applications. 14(2): 443-459. doi:10.1890/02-5349

Wright, L.C.; Berryman, A.A. 1978. Effect of defoliation by the Douglas-fir tussock moth on moisture stress in grand fir and subsequent attack by the fir engraver beetle (Coleoptera: Scolytidae). Res. Note PNW-323. Portland, OR: USDA Forest Service, Pacific Northwest Forest and Range Experiment Station. 14 p. https://archive.org/download/effectofdefoliat323wrig/effectofdefoliat323wrig.pdf

Wright, J.W.; Kung, F.H.; Read, R.A.; Lemmieu, W.A.; Bright, J.N. 1971. Genetic variaton in Rocky Mountain Douglas-fir. Silvae Genetica. 10(3): 53-100. https://www.thuenen.de/media/institute/fg/PDF/Silvae_Genetica/1971/Vol._20_Heft_3/20_3_54.pdf

Wright, L.C.; Berryman, A.A.; Gurusiddaiah, S. 1979. Host resistance to the fir engraver beetle, *Scolytus ventralis* (Coleoptera: Scolytidae). 4. Effect of defoliation on wound monoterpene and inner bark carbohydrate concentrations. Canadian Entomologist. 111(11): 1255-1262. doi:10.4039/Ent1111255-11

Wright, L.C.; Berryman, A.A.; Wickman, B.E. 1984. Abundance of the fir engraver, *Scolytus ventralis*, and the Douglas-fir beetle, *Dendroctonus pseudotsugae*, following tree defoliation by the Douglas-fir tussock moth, *Orgyia pseudotsugata*. Canadian Entomologist. 116(3): 293-305. doi:10.4039/Ent116293-3

Wright, E.F.; Coates, K.D.; Bartemucci, P. 1998. Regeneration from seed of six tree species in the interior cedar-hemlock forests of British Columbia as affected by substrate and canopy gap position. Canadian Journal of Forest Research. 28(9): 1352-1364. doi:10.1139/x98-117

Wright, E.F.; Canham, C.D.; Coates, K.D. 2000. Effects of suppression and release on sapling growth for 11 tree species of northern, interior British Columbia. Canadian Journal of Forest Research. 30(10): 1571-1580. doi:10.1139/x00-089

Wright, P.; Harmon, M.; Swanson, F. 2002. Assessing the effect of fire regime on coarse woody debris. In: Laudenslayer, W.F., Jr.; Shea, P.J.; Valentine, B.E.; Weatherspoon, C.P.; Lisle, T.E., eds. Proceedings of the symposium on the ecology and management of dead wood in western forests. Gen. Tech. Rep. PSW-GTR-181. Albany, CA: USDA Forest Service, Pacific Southwest Research Station: 621-634. http://www.treesearch.fs.fed.us/pubs/6718

Wu, J.; Loucks, O.L. 1995. From balance of nature to hierarchical patch dynamics: a paradigm shift in ecology. Quarterly Review of Biology. 70(4): 439-466. doi:10.1086/419172

Wuebbles, D.J.; Fahey, D.W.; Hibbard, K.A.; Dokken, D.J.; Stewart, B.C.; Maycock, T.K., eds. 2017. Climate science special report: Fourth National Climate Assessment, Volume I. Washington, DC: U.S. Global Change Research Program. 470 p. https://science2017.globalchange.gov/downloads/CSSR2017_FullReport.pdf

Wulder, M.A.; Franklin, S.E., eds. 2007. Understanding forest disturbance and spatial pattern. Boca Raton, FL: CRC Press. 246 p. isbn:0-8493-3425-X

Wurtzebach, Z.; Schultz, C.; Waltz, A.E.M.; Esch, B.E.; Wasserman, T.N. 2019. Broader-scale monitoring for federal forest planning: Challenges and opportunities. Journal of Forestry. 117(3): 244-255. doi:10.1093/jofore/fvz009

Wurtzebach, Z.; DeRose, R.J.; Bush, R.R.; Goeking, S.A.; Healey, S.; Menlove, J.; Pelz, K.A.; Schultz, C.; Shaw, J.D.; Witt, C. 2020. Supporting national forest system planning with Forest Inventory and Analysis data. Journal of Forestry. 118(3): 289-306. doi:10.1093/jofore/fvz061

York, R.A.; Battles, J.J.; Heald, R.C. 2003. Edge effects in mixed conifer group selection openings: tree height response to resource gradients. Forest Ecology and Management. 179(1-3): 107-121. doi:10.1016/S0378-1127(02)00487-5

York, R.A.; Heald, R.C.; Battles, J.J.; York, J.D. 2004. Group selection management in conifer forests: relationships between opening size and tree growth. Canadian Journal of Forest Research. 34(3): 630-641. doi:10.1139/x03-222

York, R.A.; Battles, J.J.; Wenk, R.C.; Saah, D. 2012. A gap-based approach for regenerating pine species and reducing surface fuels in multi-aged mixed conifer stands in the Sierra Nevada, California. Forestry. 85(2): 203-213. doi:10.1093/forestry/cpr058

Youngblood, A. 2001. Old-growth forest structure in eastern Oregon and Washington. Northwest Science. 75(Special Issue): 110-118. http://hdl.handle.net/2376/987

Zabowski, D.; Java, B.; Scherer, G.; Everett, R.L.; Ottmar, R. 2000. Timber harvesting residue treatment: Part 1. Responses of conifer seedlings, soils and microclimate. Forest Ecology and Management. 126(1): 25-34. doi:10.1016/S0378-1127(99)00081-X

Zabowski, D.; Chambreau, D.; Rotramel, N.; Thies, W.G. 2008. Long-term effects of stump removal to control root rot on forest soil bulk density, soil carbon and nitrogen content. Forest Ecology and Management. 255(3–4): 720-727. doi:10.1016/j.foreco.2007.09.046

Zachmann, L.J.; Shaw, D.W.H.; Dickson, B.G. 2018. Prescribed fire and natural recovery produce similar long-term patterns of change in forest structure in the Lake Tahoe basin, California. Forest Ecology and Management. 409: 276-287. doi:10.1016/j.foreco.2017.11.013

Zackrisson, O.; Nilsson, M.-C.; Wardle, D.A. 1996. Key ecological function of charcoal from wildfire in the boreal forest. Oikos. 77(1): 10-19. doi:10.2307/3545580

Zald, H.S.J.; Gray, A.N.; North, M.; Kern, R.A. 2008. Initial tree regeneration responses to fire and thinning treatments in a Sierra Nevada mixed-conifer forest, USA. Forest Ecology and Management. 256(1-2): 168-179. doi:10.1016/j.foreco.2008.04.022

Zambino, P.J. 2010. Biology and pathology of *Ribes* and their implications for management of white pine blister rust. Forest Pathology. 40(3-4): 264-291. doi:10.1111/j.1439-0329.2010.00658.x

Zarnoch, S.J.; Blake, J.I.; Parresol, B.R. 2014. Are prescribed fire and thinning dominant processes affecting snag occurrence at a landscape scale? Forest Ecology and Management. 331: 144-152. doi:10.1016/j.foreco.2014.08.007

Zdanowicz, C.M.; Zielinski, G.A.; Germani, M.S. 1999. Mount Mazama eruption: calendrical age verified and atmospheric impact assessed. Geology. 27(7): 621-624. doi:10.1130/0091-7613(1999)027<0621:Mmecav>2.3.Co;2

Zeglen, S.; Pronos, J.; Merler, H. 2010. Silvicultural management of white pines in western North America. Forest Pathology. 40(3-4): 347-368. doi:10.1111/j.1439-0329.2010.00662.x

Zeide, B. 2008. The science of forestry. Journal of Sustainable Forestry. 27(4): 345-473. doi:10.1080/10549810802339225

Zenner, E.K.; Hibbs, D.E. 2000. A new method for modeling the heterogeneity of forest structure. Forest Ecology and Management. 129(1–3): 75-87. doi:10.1016/s0378-1127(99)00140-1

Zhang, J.; Powers, R.F.; Skinner, C.N. 2010. To manage or not to manage: the role of silviculture in sequestering carbon in the specter of climate change. In: Deal, R.L., ed. Integrated restoration of forested ecosystems to achieve multi-resource benefits: proceedings of the 2007 national silviculture workshop. Gen. Tech. Rep. PNW-GTR-733. Portland, OR: USDA Forest Service, Pacific Northwest Research Station: 95-110. https://www.fs.usda.gov/treesearch/pubs/29399

Zhao, S.; Klutsch, J.G.; Cale, J.A.; Erbilgin, N. 2019. Mountain pine beetle outbreak enhanced resin duct-defenses of lodgepole pine trees. Forest Ecology and Management. 441: 271-279. doi:10.1016/j.foreco.2019.03.023

Zhao, D.; Bullock, B.P.; Montes, C.R.; Wang, M. 2020. Rethinking maximum stand basal area and maximum SDI from the aspect of stand dynamics. Forest Ecology and Management. 475: 118462 (10 p). doi:10.1016/j.foreco.2020.118462

Zhong, J.; van der Kamp, B.J. 1999. Pathology of conifer seed and timing of germination in high-elevation subalpine fir and Engelmann spruce forests of the southern interior of British Columbia. Canadian Journal of Forest Research. 29(2): 187-193. doi:10.1139/x98-201

Zimmerman, G.T.; Laven, R.D. 1984. Ecological interrelationships of dwarf mistletoe and fire in lodgepole pine forests. In: Hawksworth, F.G.; Scharpf, R.F., tech coords. Biology of dwarf mistletoes: proceedings of the symposium. Gen. Tech. Rep. RM-111. Fort Collins, CO: USDA Forest Service, Rocky Mountain Forest and Range Experiment Station: 123-131. https://archive.org/download/CAT31299587/CAT31299587.pdf

Zimmerman, G.T.; Omi, P.N. 1998. Fire restoration options in lodgepole pine ecosystems. In: Pruden, T.L.; Brennan, L.A., eds. Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology Conference Proceedings No. 20. Tallahassee, FL: Tall Timbers Research Station: 285-297. https://talltimbers.org/wp-content/uploads/2018/09/285-ZimmermanandOmi1998_op.pdf

Zirlewagen, D.; Raben, G.; Weise, M. 2007. Zoning of forest health conditions based on a set of soil, topographic and vegetation parameters. Forest Ecology and Management. 248(1-2): 43-55. doi:10.1016/j.foreco.2007.02.038

Znerold, R.M. 1979. Western bracken control with asulam. Thesis. Pullman, WA: Washington State University. 36 p.

Zobel, D.B. 1973. Local variation in intergrading Abies grandis-Abies concolor populations in the central Oregon Cascades: Needle morphology and periderm color. Botanical Gazette. 134(3): 209-220. doi:10.2307/2474013

Zon, R. 1915. Native shrubs and herbaceous plants as indicators of planting sites, Ephraim Canyon. Unpublished report obtained from National Archives, College Park, MD; record group 95. 7 p. On file with: USDA Forest Service, Umatilla National Forest, Supervisor's Office, Pendleton, OR.

Zwieniecki, M.A.; Newton, M. 1996. Seasonal pattern of water depletion from soil–rock profiles in a Mediterranean climate in southwestern Oregon. Canadian Journal of Forest Research. 26(8): 1346-1352. doi:10.1139/x26-150

Zwolak, R. 2009. A meta-analysis of the effects of wildfire, clearcutting, and partial harvest on the abundance of North American small mammals. Forest Ecology and Management. 258(5): 539-545. doi:10.1016/j.foreco.2009.05.033

Zwolak, R.; Pearson, D.E.; Ortega, Y.K.; Crone, E.E. 2010. Fire and mice: Seed predation moderates fire's influence on conifer recruitment. Ecology. 91(4): 1124-1131. doi:10.1890/09-0332.1

APPENDIX 3: 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 more than 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, and they continuously evolve as an issue matures, experiencing many iterations (versions) 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 perception 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. Examples 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).

These 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)

Paper #	Title
20	Height-diameter equations for tree species of Blue and Wallowa Mountains
21	Historical fires 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
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

Paper #	**Title**
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

The genesis of this white paper began in 2010. Umatilla National Forest had been receiving scoping comments, appeals, and similar NEPA input regarding moist-forest proposals for Cobbler, Mirage, and Wildcat projects.

As a result of what was perceived as 'building controversy' about moist-forest treatments, the Forest's Leadership Team decided to spend most of a monthly FLT meeting (Jan. 21, 2010) discussing moist-forest management concepts, how (and whether) moist-forest management proposals should be handled for future timber-sale planning efforts (especially for north-end ranger district planning areas, where moist-forest was a predominant land type), and how moist-forest project proposals should be characterized (described) when considering issues and concerns brought forward in recent public NEPA input.

After the January 2010 FLT meeting, I was asked to begin compiling pertinent scientific background information relating to Blue Mountains moist forests and their management (I had already done this for dry forests, so the FLT had a model or template for what they expected for moist forests). This moist-forest white paper was the ultimate result of that request.

Note that this white paper was not the Forest's sole response to what was viewed as growing public concern about moist-forest management. Umatilla NF implemented several other approaches, including a more rigorous process for evaluating science quality (not just for evaluating our own 'best available science' cited during project planning, but also to be used when evaluating science input received from external commenters during the NEPA process), and the Forest submitted a formal request to the Pacific Northwest Region Regional Forester to assemble a science integration team focused on eastside moist forests.

(Note that a science integration team was eventually convened, and it produced the Stine et al. 2014 moist-forest report. Within the Umatilla NF, the Stine report is viewed as effectively satisfying the Forest's request for compilation and synthesis of moist-forest science relating to a Blue Mountains/eastside context.)

March 2014: the first version (328 p.) was released in January 2013; this revision was extensive, adding almost a hundred additional pages of narrative, figures, and references. This revision also incorporates comments and suggested edits from three technical peer reviewers (see Acknowledgments).

December 2019: this revision added additional references, and all existing references were revised (if necessary) to add doi or weblink information to assist white-paper users in obtaining a copy.