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October 18 - 21, 2004 **Klamath Falls, Oregon**

Abstract

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Ponderosa pine is one of the most widely distributed tree species in western North America. It is highly-valued as a source of lumber, but also is key to the health and social value western forests, whether growing in pure stands or in mixture with other conifer and hardwood species. In recent years, management objectives for forests containing this species have shifted from an emphasis on timber production to an emphasis on restoring ecosystem health and reducing the risk of non-characteristic wildfires. The symposium on "Ponderosa Pine: Issues, Trends, and Management" was convened to provide a venue for researchers and managers to explore the current state-of-our-knowledge, including management practices that help managers to adapt to constantly changing constraints and objectives.

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Proceedings of the Symposium on Ponderosa Pine: Issues, Trends, and Management

October 18 - 21, 2004 **Klamath Falls, Oregon**

Technical Coordinators

Martin W. Ritchie, Douglas A. Maguire and Andrew Youngblood

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Preface

As early exploitation of standing timber yielded to intensive reforestation and active stand management, our silvicultural knowledge base has expanded tremendously. Changing public perception of timber harvesting and different societal values of forest resources, coupled with rapid expansion of the wildland/urban interface has compelled foresters to pursue multiple-resource management with less emphasis on timber production on many public and private lands. Conversely, demand for ponderosa pine wood has intensified management on other private lands, and large disturbances from fire, insects, and disease have fueled intense debates about rehabilitation efforts. The result of these debates and the accompanying information need has been a notable increase in our knowledge of ponderosa pine ecosystems and better understanding of human values and perceptions.

The presentations at the symposium were selected to cover a broad spectrum of topics and to update forest landowners, professional foresters, and forest scientists about the current issues, trends and management of ponderosa pine ecosystems. The presentations in this compilation are arranged by broad topical categories.

Two introductory papers serve to set the foundation for science-based management of ponderosa pine ecosystems. Graham and Jain first present characteristics and key functional elements of ponderosa pine systems. Fiske and Tappeiner follow with an overview of important silvicultural research findings over during the last century, findings that will remain important as we manage ponderosa pine forests for the 21st century. Their work focuses on two broad themes: timber stand improvement and stand dynamics.

Silviculturists today often are challenged to manipulate stand structures to meet an increasingly varied set of objectives. In the section dealing with silviculture and ecosystem management, Youngblood presents an overview of silvicultural systems applicable to ponderosa pine. His presentation covers traditional systems designed to produce commodities such as timber, forage, and water, as well as changes in silvicultural practices prompted by forest health and fuel reduction issues. O'Hara describes an approach for promoting and managing multi-cohort stands of ponderosa pine. Various alternative methods have been proposed for stocking control in uneven-age systems, and he compares these to one based on allocating leaf area among cohorts. Productivity is predicted to be comparable to that of even-aged silvicultural systems. Because, the fundamental tool of siliviculture remains the manipulation of stand structure regardless of objectives or issues, and stand density is probably the most commonly manipulated aspect of stand structure. Oliver presents results from 40 years of research on alternative stand density regimes in ponderosa pine, referred to as the West-wide levels-of-growing-stock study. The initial objective was to evaluate the long-term effects of density management on tree growth and mortality, but many insights have also been gained into the range of stand structures obtained by varying stand density regime. Ritchie and Harcksen describe a new large-scale interdisciplinary study designed to accelerate the development of late-successional features in pine-dominated ecosystems. Keyes and Maguire then describe the complex relationships between shrubs and natural seedling recruitment in ponderosa pine, and show that shrubs can enhance early seedling survival.

In the section on soils and productivity, Busse and Riegel describe the response of soil quality to thinning and prescribed fire, and emphasize the role that N-fixing shrubs play in offsetting losses of soil N during prescribed fire. In addressing soil amendments, Johnson and others show that response to fertilization is dependent on the soil and parent material, and note that ponderosa pine often does not show a strong growth response to N fertilization alone. Multi-nutrient fertilization, however, may provide a better growth response.

The value of ponderosa pine forests for wildlife habitat and forage production for domestic livestock is well documented. Our understanding of the role of specific species and the related structural component in these ecosystems is less complete. Vavra and others note that both domestic and native ungulates can have an impact on understory composition and structure in ponderosa pine systems, and that these changes can impact nutrient cycling, energy flow, biodiversity, stand density, and fire regimes. George and others compare species composition of birds in late- and mid-seral ponderosa pine forests. Their results are consistent with other studies suggesting that woodpeckers and bark gleaners are strongly associated with large trees and snags; however, they found no difference in overall species diversity. Snags are considered an important element for habitat suitability for some avian species. Laudenslayer describes a long-term study of snag demography demonstrating that snag longevity varies with tree species, tree size, soil characteristics, and topography. He recommends that these factors be considered when developing snag management guidelines. The means by which snags are created may also be an important factor in determining habitat suitability. Farris and Zack show that snags created by fire or bark beetles decay more rapidly than those resulting from other mortality agents, but also experience the greatest foraging and nesting use by woodpeckers.

Forest disturbance regimes strongly influence the dynamics of ponderosa pine ecosystems. Currently high fuel levels resulting from fire suppression, the attendant high risk of severe fire, and very recent extreme fire events have created an urgent need to re-asses the role fire in ponderosa pine ecosystems. Fitzgerald discusses the importance of understanding historic fire regimes if managers are to develop effective restoration treatments. Disturbances created by disease also contribute significantly to the long-term dynamics of ponderosa pine forests. Filip describes the role of forest diseases in forest succession and in the maintenance of biological diversity, as well as their interaction with insect outbreaks and fire frequency and severity. The risk of insect outbreaks has also heightened with the disruption of historic fire regimes. Fettig describes chemical, silvicultural and semiochemical techniques for minimizing insect-induced mortality in ponderosa pine stands.

Finally, experience gleaned from operational projects and case studies can extremely valuable in developing effective management strategies for ponderosa pine. In recent years, the application of prescribed fire in reducing fuel levels and maintaining healthy, resilient stands has received greater attention. However, the paper by Skinner shows that applying prescribed fire in stands that have had many decades of fire exclusion can be difficult, and that benefits related to reducing fire danger may be very short lived under such conditions. Arena describes three ongoing studies on the Warm Springs Indian Reservation designed to evaluate the feasibility and efficacy of removing small (unmerchantable) material from dense stands, identify appropriate stocking levels for uneven-aged ponderosa pine stands, and assess the utilization potential of material typically removed in fuel reduction treatments but otherwise considered unsuitable for most forest products.. Webster and Fredrickson finish the proceedings with a set of very practical guidelines for establishing and managing young plantations of ponderosa pine.

Ponderosa Pine Ecosystems[1]

Russell T. Graham[2] and Theresa B. Jain[2]

Abstract

Ponderosa pine is a wide-ranging conifer occurring throughout the United States, southern Canada, and northern Mexico. Since the 1800s, ponderosa pine forests have fueled the economies of the West. In western North America, ponderosa pine grows predominantly in the moist and dry forests. In the Black Hills of South Dakota and the southern portion of its range, the species primarily occupies ponderosa pine potential vegetation types (PVTs) but, in the northern portion of its range, it grows on Douglas-fir, grand fir and/or white fir and western redcedar PVTs. Within this wide range of biophysical settings it is often associated with complex vegetation mixes. Non-lethal, mixed, and lethal wildfires historically burned through most ponderosa pine forests leaving in their wake a wide variety of species compositions and vegetative structures arranged in a variety of mosaics. Since the 1800s, fire exclusion, animal grazing, timber harvest, and climate cycles have contributed to changing these forests. As a result, succession accelerated, plant compositions shifted, trees and other biomass accumulated, soil chemical and physical properties changed, non-native plants were introduced, and epidemics of insects and diseases are more common. Together these changes altered fire regimes, displaced native species, and disrupted other ecological processes. Although the extent of wildfires that now burn in these altered forests is not noteworthy, their severity is. Canopy treatments and surface fuel treatments in combination are most likely to reduce the risk of severe and intense wildfires in these forests that mean a great deal to individuals and society.

Introduction

Ponderosa pine (*Pinus ponderosa* P. & C. Lawson), is a wide-ranging conifer occurring throughout the western United States, southern Canada, and northern Mexico (*fig. 1*) (Little 1971). Generally its greatest extent is in the Inland Northwestern United States and in northern California. However, it is a prevalent species in the Black Hills of South Dakota and Wyoming, along the Front Range of the Rocky Mountains in Colorado and along the Mogollon Rim in Arizona, the rugged escarpment that forms the southern limit of the Colorado Plateau. The species occupies sites with elevations ranging from sea level to 3,281 m (10,000 ft.) depending on latitude (Oliver and Ryker 1990). In terms of area occupied, it is only second to Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco var. *glauca* (Beissn.) Franco) (Van Hooser and Keegan 1988). Two geographic varieties are recognized, the Rocky Mountain (*Pinus ponderosa* var s*copulorum* Engelm.) which grows primarily in the Rocky Mountains and the Pacific (*Pinus ponderosa* var *ponderosa* which is widely distributed in the mountains of the Pacific Coast from British Columbia into California and western Nevada (Little 1979). Arizona pine (*Pinus arizonica* Engelm.), once considered to be a variety of ponderosa, grows in southwest

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[1] An abbreviated version of this paper was presented at the symposium on Ponderosa Pine: Issues, Trends and Management, October 18-21, 2004, Klamath Falls, Oregon.

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New Mexico, southeast Arizona, and northern Mexico. Within this wide range, ponderosa pine grows across highly variable biophysical settings (e.g., soils, slopes, aspects, associated vegetation, and fauna). Our objective in this paper is to describe ponderosa pine ecosystems by drawing heavily from our familiarity with ponderosa pine forests within the Rocky Mountains. In addition, we describe briefly the changes that have occurred in ponderosa pine forests over the last 100 years as the result of disturbances or lack-there-of, vegetative succession, fire exclusion, and wildfires (Graham and others 2004). We also provide some insight into how treatments might be used to restore these forests. We suggest even though our paper may be Rocky Mountain centric, the concepts presented most likely have application to any location where ponderosa pine occurs within western North America.

Ponderosa pine is the principle species on over 11 million ha (27 million ac.) and for every 2.8 ha (7 ac.) it dominates, it is present on an additional 1.4 ha (3.5 ac.). Within the western United States, California alone contains the greatest concentrations of ponderosa pine (2.07 million ha (5 million ac.) closely followed by Oregon with 1.9 million ha (4.7 million ac.) and, when combined, Arizona and New Mexico contain an additional 2.5 million ha (6 million ac.) of ponderosa pine (Van Hooser and Keegan 1988). Ponderosa pine fueled the economies of the West beginning in the 1860s when pines were harvested to supply building material to farms, mines, and towns as they developed. With the coming of the railroads in the early 1900s, harvesting increased mostly by clearcutting. However, with the advent of improved roads allowing access by tractors and trucks, partial cutting became the dominant harvesting method. Ponderosa pine's high value, especially the value of mature and old trees, led to efforts in classifying tree vigor and the risk of mortality which was used in selection silvicultural systems (e.g., vigor selection) (Dunning 1928, Keen 1943, Meyer 1934). High risk trees were removed to decrease the incidence of bark beetles and capture the value of imminent mortality (Keen 1936, Barrett 1979). Ponderosa pine forests presented an opportunity for intensive management across large expanses of the West and considerable research and managerial effort was directed towards this end (Pearson 1950).

Ponderosa Pine Characteristics

Ponderosa pine is three-needled, however, fascicles with both two and three needles can be found on the same tree (Harlow and Harrar 1968). Ponderosa pine trees can exceed 120 cm (48 in.) in diameter on good sites in northern Idaho and western Montana and exceed 183 cm (6 ft.) in diameter on sites in California (Van Hooser and Keegan 1988). The ability of the species to survive low severity wildfires is one of its most unique characteristics. At small diameters (e.g., 5 cm, 2 in.), ponderosa trees can withstand heat from most surface fires because of the insulating bark that protects the underlying cambial layers. Large ponderosa pines with yellow bark invoke a sense of a majestic forest and spiritual feeling in people who frequent these forests. These traits are exemplified by the many historical photos depicting people enjoying the presence of large yellow ponderosa pine trees (Grafe and Horsted 2002, Gruell and others 1982, Smith and Arno 1999). Moreover, these

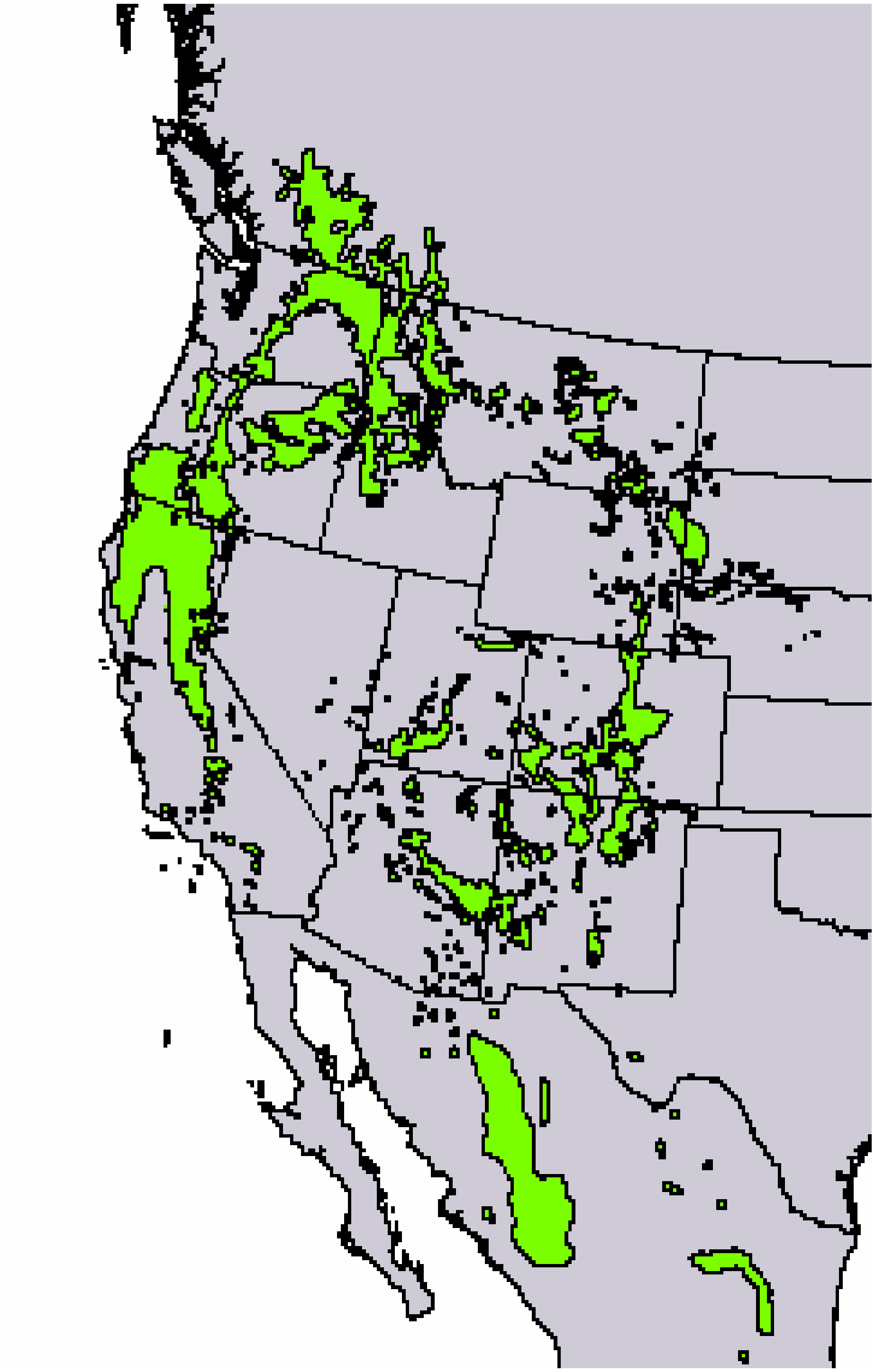


Figure 1—The range of ponderosa pine (Little 1971).

conditions provide valuable wildlife habitat (Long and Smith 2000, Reynolds and others 1992, Thomas 1979) and protect watersheds for the production of domestic water such as those occurring on the Front Range of the Rocky Mountains in Colorado (Robichaud and others 2003).

For trees to survive and develop they must be genetically adapted to the site. For example, the environmental interval in elevation in which ponderosa pine populations show habitat specificity is approximately 453 m (1380 ft.) or 38 frost-free days. In contrast, no habitat specificity in elevation has been found for western white pine (*Pinus monticola* Dougl. ex D. Don), and the habitat specificity interval in frost-free days is 90. The narrowest habitat specificity for any of the associates of ponderosa pine occurs with Douglas-fir which has an environmental interval of 113 m (650 ft.) in elevation or 18 frost-free days (Rehfeldt 1994). Genetically, ponderosa pine has intermediate adaptation to sites compared to Douglas-fir, considered a specialist, and western white pine, considered a generalist. The size of ponderosa pine seed crops in general is smaller than most of its associates and, if it wasn't for western larch (*Larix occidentalis* Nutt.) flowers being frequently damaged by frost, it would also have the most infrequent cone crop of any associated conifer (Minore 1979, Graham and others 1995). Ponderosa pine regenerates readily on both mineral and burned over seed beds, however, it does not establish well on unburned organic surfaces (Haig and others 1941).

Ponderosa Pine Forests

In western North America, ponderosa pine grows within both moist and dry forests but seldom occurs in the cold forests (i.e. subalpine forests). Climate, as well as the associated tree species, distinguishes the two general forest classifications where ponderosa pine can grow (Hann and others 1997). In dry forests, associated species, beginning with the most intolerant to shade and competition, include quaking aspen (*Populus tremuloides* Michx.), western larch, lodgepole pine (*Pinus contorta* Dougl. ex Loud), Douglas-fir, and grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.) or white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.). In moist forests, the species most often grows on south facing aspects, but can occur in small amounts throughout the entire forest type. Early-seral associates include western larch and lodgepole pine while Douglas-fir, western white pine, and grand fir/white fir are more tolerant than ponderosa pine (Minore 1979). The most tolerant tree species associated with ponderosa pine in the moist forests is western redcedar (*Thuja plicata* Donn ex D. Don) (Cooper and others 1991, Daubenmire and Daubenmire 1968). The combination of species and their variable tolerances to competition and shade gives rise to a variety of forest compositions and structures in both forests types.

Succession In Ponderosa Pine Forests

Succession is a term applied to the gradual supplanting of one community of plants by another on a given site through time (Smith and Arno 1999). Vegetative complexes evolve after a disturbance such as a lethal fire (i.e. fires that kill the majority of the dominant and codominant canopy layers) (Hann and others 1997). Early-seral stages often begin with a grass/forb/shrub stage, succeeded by tree seedlings and saplings which grow to young trees, and subsequently develop into the late-seral mature and old vegetative complexes. In some systems, such as those dominated by ponderosa pine, these or similar stages may develop in less than 250 years but in other systems, such as Pacific coastal Douglas-fir (*Psuedotsuga menziesii* (Mirbel) Franco var. *menziesii*), it may take in excess of 1,000 years for the full compliment of structural stages inherent to the system to develop (Franklin and others 2002, Reynolds and others 1992).

A very useful characteristic of vegetative succession is that for a given biophysical environment and species mix, the vegetative development over time from early-seral (pioneer) through late-seral (climax) is predictable. Because of this predictability it can be used to classify sites by the potential vegetation that will occur. These classifications are usually identified by indicator species occurring at the late-seral stage (Daubenmire and Daubenmire 1968, Hann and others 1997, Smith and Arno 1999).

Besides the conceptual endpoint vegetation, and depending on the intensity and severity of disturbances that may occur on a site, there can be many successional and developmental pathways along with many vegetative and structural mixes possible for a given site and species mix (Smith and Fisher 1997). For example, figure 2 shows the successional pathways for a Douglas-fir potential vegetation type (PVT) based on a fire’s severity. Sites exhibiting this succession would be classified as a Douglas-fir potential vegetation type; however, it could be perpetually dominated by ponderosa pine (e.g., D2 – *fig. 2*). Of the PVTs on which ponderosa pine occurs this is one of the simpler in terms of potential species and disturbance interactions, yet a large number of vegetative compositions and structures are possible. This heterogeneity in composition and structure can be arranged in a variety of interspersed mosaics ranging in size from less than 0.1 ha (0.25 ac.) to 100s of hectares (Long and Smith 2000).

Albeit potential vegetation can be a superb classification with excellent interpretative relations, care must be exercised when using such systems. Two sites may have the same potential vegetation classification but their physical locations often reflect a different environment. Ponderosa pine growing on a Douglas-fir/nine bark (*Physocarpus malvaceus* (Greene) Kuntze) PVT in Montana, for example, usually occur on northerly aspects while sites similarly classified in Idaho regularly occur on south facing slopes (Cooper and others 1991, Pfister and others 1977). These differences are reflected in the classification systems used and the different environments expressed by similar species. Nevertheless, even with these differences, potential vegetation is very useful for classifying sites which provide interpretative value for ecological concepts such as successional pathways, fire relations, species mixes, wildlife relations, coarse woody debris relations, site productivity estimates, and vegetation simulations (Bradley and others 1992a, 1992b, Cooper and others 1991, Graham and others 1994, Graham and others 1999b, Pfister and others 1977, Smith and Fischer 1997, Wykoff and others 1982).

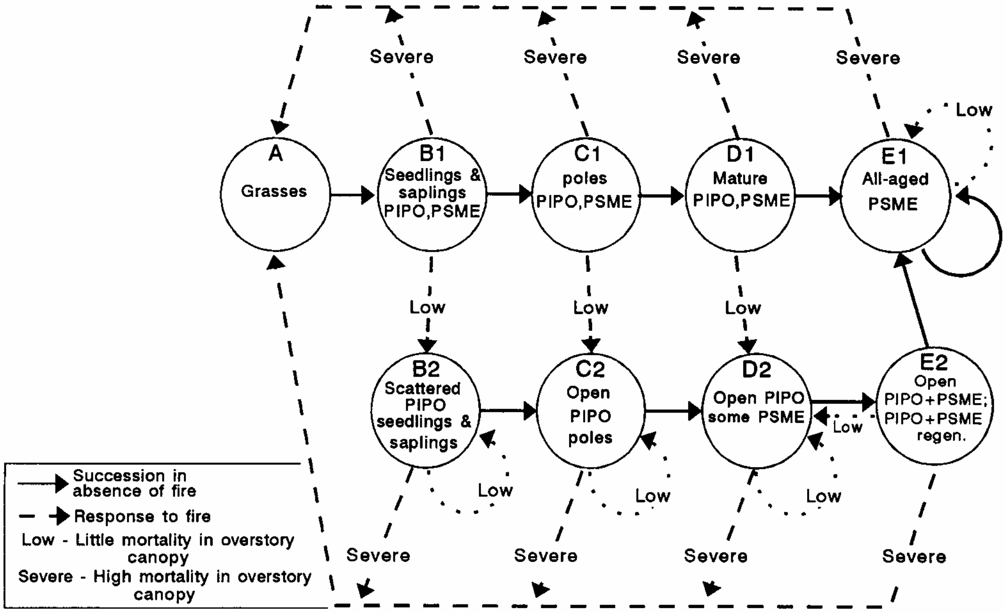


Figure 2—Possible successional pathways for a Douglas-fir potential vegetation type in response to fire as the only disturbance (Smith and Fisher 1997). Note the many different vegetative and structural combinations that can occur on one of the simpler potential vegetation types. PIPO=ponderosa pine, PSME=Douglas-fir.

Vegetative Complexes

In the southern and extreme eastern portion of the range, ponderosa pine grows primarily on ponderosa pine PVTs. On these settings, quaking aspen is the most frequent early-seral tree species (Hoffman and Alexander 1987, Youngblood and Mauk 1985). Ground-level vegetation includes oaks (*Quercus* spp.), grasses (*Festuca* and *Agropyron* spp.), and low shrubs (e.g., snowberry (*Symphorcarpus* spp.) and spirea (*Spirea* spp.). Russet buffaloberry (*Shepherdia canadensis* (L.) Nutt.), a frequent shrub in these forests, stands out for its nitrogen fixing properties which is the process of making elemental nitrogen in the atmosphere available to plants (Jurgensen and others 1991).

With increasing moisture, ponderosa pine occurs as a mid-seral species and Douglas-fir becomes the late-seral species (*fig. 2*). Quaking aspen and lodgepole pine are early-seral associates of ponderosa pine on these Douglas-fir PVTs (Mauk and Henderson 1984). These ponderosa pine forests occur in the Rocky Mountains along the Front Range of Colorado, in Utah, and in southern Idaho. They also occur along the western slopes of the Sierra Nevadas in California and the eastern slopes of the Cascades in Oregon (Franklin and Fites-Kaufman 1996, Hann and others 1997, Steele and others 1983). Ground-level vegetation includes ninebark, elk sedge (*Carex geyeri* Boott), and pine grass (*Calamagrostis rubescens* Buckl.). These species, in particular, exemplify aggressive survivors after disturbance (e.g., fire, mechanical site preparation) and are strong competitors for light and nutrients which compete with ponderosa pine seedlings (Baumgartner and others 1986).

In several locales, dry grand fir and white fir PVTs represent the dry forests (Hann and others 1997). On such settings, ponderosa pine and Douglas-fir occur but are succeeded by late-seral grand fir and/or white fir in the absence of disturbance (Bradley and others 1992b). Additional trees that can occur in such forests include juniper (*Juniperus* spp.), pinyon pine (*Pinus edulis* Engelm.), sugar pine (*Pinus lambertiana* Dougl.), incense-cedar (*Calocedrus decurrens* (Torr.) Florin), western larch, Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.), and lodgepole pine. Pine grass and ninebark are frequent associates but tall shrubs such as Rocky Mountain maple (*Acer glabrum* Torr.) often occur.

The wettest forests where ponderosa pine occurs are the wet grand fir and/or white fir PVTs and the driest western redcedar PVTs. Such forests occur in the interior northwestern United States and in southern British Columbia (Cooper and others 1991, Daubenmire and Daubenmire 1968, Hann and others 1997). The western redcedar PVT is by far the most productive type on which ponderosa pine occurs, and lush and complex vegetation mixes may develop. Western white pine is a frequent associate of ponderosa pine with an occasional paper birch (*Betula papyrifera* Marsh.). A rich understory of shrubs, grasses, and forbs occur in these forests. Early seral-species such as redstem ceanothus (*Ceanothus sanguineus* Pursh), snowbrush ceanothus (*Ceanothus velutinus* Dougl. ex Hook.) and Sitka alder (*Alnus viridis* (Vill.) Lam. & DC. ssp. *sinuata* (Regel A. & D. Löve) rapidly colonize sites after disturbance and are also active nitrogen fixers (Jurgensen and others 1991, Smith and Fischer 1997). Mid-seral shrubs include Rocky Mountain maple which readily survives disturbances and is joined by late-seral species such as huckleberry (*Vaccinium* spp.) and false huckleberry (*Menziesia ferruginea* Smith). The latter readily survives disturbances but is an aggressive colonizer. Probably one of the greatest competitors and survivors after disturbance of any ground-level species occurring with ponderosa pine is pine grass. This ground-level vegetation can play critical roles in forests such as providing wildlife habitat, stabilizing soil, and capturing nutrients after disturbance. Fireweed, (*Chamerion angustifolium* (L.) Holub), for example, rapidly regenerates after fire and captures and recycles nitrogen (Baumgartner and others 1986, Cooper and others 1991, Daubenmire and Daubenmire 1968, Reynolds and others 1992, Smith and Fischer 1997). Because of the range of species that can occur with ponderosa pine and their wide range of tolerance (e.g., shade, competition, fire) along with how they interact with disturbances a plethora of vegetative compositions and structures can occur within ponderosa pine forests arranged and interspersed in a variety of mosaics.

Wildfire and Ponderosa Pine Forests

Before successful fire exclusion, temperature and precipitation patterns combined with natural and human ignitions allowed fires to burn the dry forests at relatively frequent (e.g., < 40 years) intervals (Agee 1993, Hann and others 1997). Cultural burning by Native Americans augmented and even dominated burning in several locations (Barrett and Arno 1982, Stewart 1951). In the northern Rocky Mountains of Idaho and western Montana, dry settings (ponderosa pine and/or Douglas fir PVTs) historically burned by non-lethal (low severity surface fires that did not kill or kill few overstory trees) wildfires at 15 to 23 year mean return intervals

(Smith and Fischer 1997). Mesic forests containing ponderosa pine (grand fir and/or Douglas-fir PVTs) were burned frequently by lethal fires (i.e. fires that kill the majority of the dominant and codominant canopy layers) or mixed fires (a combination of lethal and non-lethal fires), at mean return intervals extending to over 60 years (Smith and Fischer 1997). In the central and southern Rockies (ponderosa pine and/or Douglas-fir PVT's), although non-lethal fires dominated, mixed fires also occurred, especially along the Front Range of the Rocky Mountains in Colorado (Bradley and others 1992a, 1992b, Fulè and others 1997, Kaufmann and others 2001). On the driest settings, (ponderosa pine and/or woodlands), because of discontinuous surface fuels, fires tended to be few (Bradley and others 1992b). In contrast to other locales dominated by late-seral ponderosa pine, the forests of the Black Hills possibly experienced greater extents of lethal fires (Shepperd and Battaglia 2002, Shinnen and Baker 1997). Nevertheless, historical wildfires most likely burned through most ponderosa pine forests leaving in their wake a wide variety of species compositions and vegetative structures.

Other Disturbances

In the western United States domestic livestock grazing and harvesting of ponderosa pine forests was occurring by the mid 1800s (Cooper 1960, Rasmussen 1941). Ponderosa pine was extensively harvested, altering both forest composition and structure (Barrett 1979, Pearson 1950,Van Hooser and Keegan 1988). In mesic forests, grand fir and/or white fir and Douglas-fir rapidly colonized these sites when ponderosa pine was harvested. Especially on the ponderosa pine PVT, grass cover tended to decrease ponderosa pine seedling establishment and survival (Brawn and Balda 1988). However, when heavy livestock grazing ceased in the early 1900s in the southwestern United States, dense stands of ponderosa pine seedlings became established. Because of fire exclusion, climate changes, and other factors these trees readily developed into dense stands (Covington and Moore 1994, Pearson 1950, Stein 1988).

The dense stands that developed increased the abundance of insect and disease epidemics, and when combined with fire exclusion, significantly altered the composition and structure of these forests (Harvey and others 2000). Historically western pine beetle (*Dendroctonus brevicomis* LeConte), pine engraver (*Ips* spp.), fir engraver (*Scolytus ventralis* LeConte), Douglas-fir tussock moth (*Orgyia psudotsugata* McDunnough) were insects associated with regularly burned areas (Hessburg and others 1994). In most years bark beetles occurred at endemic levels in ponderosa pine, Douglas-fir, and grand fir killing large and weakened trees that were struck by lightning, infected by root disease (*Armellaria* spp.), or too old to resist attack (Williams and others 1986, Wu and others 1996). Pine engraver and fir engraver beetles attacked young, densely stocked ponderosa pine or removed trees scorched by low-intensity surface fires and/or trees severely infected with disease. Sometimes disease and insect infestations increased during droughts when trees were stressed.

Since fire exclusion in some settings, these same insects have occurred at epidemic levels (Hedden and others 1981, Gardner and others 1997, Schmid and Mata 1992). Today (2005) ponderosa pine continues to be susceptible to the western pine beetle and mountain pine beetle often kills ponderosa pine on Douglas-fir and grand fir/white fir PVTs. The pine engraver beetle is more abundant and destructive today with some of the severest outbreaks occurring on low-elevation ponderosa pine

PVTs (Hessburg and others 1994). Pandora moth (*Coloradia pondora* Blake) defoliates ponderosa pine and scattered outbreaks have occurred in Arizona, California, Colorado, and Oregon during the 20th century. The larvae prefer loose soils created by weathered granites or pumice where they burrow and pupate (Speer and others 2001). In addition to ponderosa pine being damaged by insects, dense stands of Douglas-fir and grand fir or white fir that developed on many settings are very susceptible to both defoliators and root diseases.

Weather, another formidable disturbance, in the form of snow, ice, or wind often creates variable sized canopy openings ranging to 16 ha (40 ac.) (Boldt and Van Deusen 1974, Shepperd and Battaglia 2002). The crowns of seral species (ponderosa pine, western larch, and western white pine) tend to shed snow thereby minimizing crown breakage. However, as the snow sloughed from tree crowns it could damage trees occurring in the lower layers. Often these mid- and late-seral species (e.g., grand fir, Douglas-fir) would break, creating canopy gaps, decreasing stand densities, and altering species composition (Jain and Graham 2005). In addition to being susceptible to weather damage when occurring in the lower canopy layers these same species when in the overstory are susceptible to the accumulation of snow and ice, which in turn often results in limb and bole damage, which in turn can increase the risk of bark beetle infestations (Jain and Graham 2005).

When combining forest succession, potential vegetation, disturbance, weather, and physical setting, it becomes obvious that ponderosa pine forests can be very complex. Depending on the combination of these components, multiple tree species can occur within stands and across landscapes, as can multiple shrub and forb species (*fig. 2*). This vegetation varies in arrangement, amounts, and juxtapositions that are continually changing, in response to the occurrence, extent, and severity of both natural and human caused disturbances.

Ponderosa Pine Stand Structure

Historically, the development and maintenance of ponderosa pine forests was favored by non-lethal fires ignited by both humans and lightning. The continued fire presence in the drier potential vegetation types (PVTs), limited the regeneration and existence of fire intolerant species. These fires burned heterogeneously and isolated areas existed where ponderosa pine regenerated (i.e., old rotten logs) (Covington and Moore 1994, Everett and others 2000). As a result ponderosa pine (more than any other) trees were often clumped into small groups (*fig. 3*). For example, on the Boise Basin Experimental Forest located in southern Idaho, five to 15 tree clumps per hectare (2 to 6 per ac.) of mature ponderosa pine trees were observed with clumps ranging in size from 0.0004 to 0.02 ha (0.001 to 0.05 ac.) (*fig. 4*) (Graham and Jain 2005). The density of trees within these clumps ranged from 21.6 m^2 ha^{-1} to 430 m^2 ha^{-1} (94 to 1871 ft^2 per ac.) of basal area (*fig. 5*). However, on a stand basis, the basal area averaged 16 m^2 ha^{-1} (70 ft^2 per acre) of basal area with 5.7 m^2 ha^{-1} (25 ft^2 per ac.) occurring within clumps of trees and 10.3 m^2 ha^{-1} (45 ft^2 per ac.) attributed to trees outside of the clumps (*fig. 6*). Pearson (1950) observed a similar clumpy nature on the Fort Valley Experimental Forest in Arizona and Youngblood and others (2004)

Figure 3—Mature ponderosa pine growing in clumps on the Pringle Falls Experimental Forest in central Oregon.

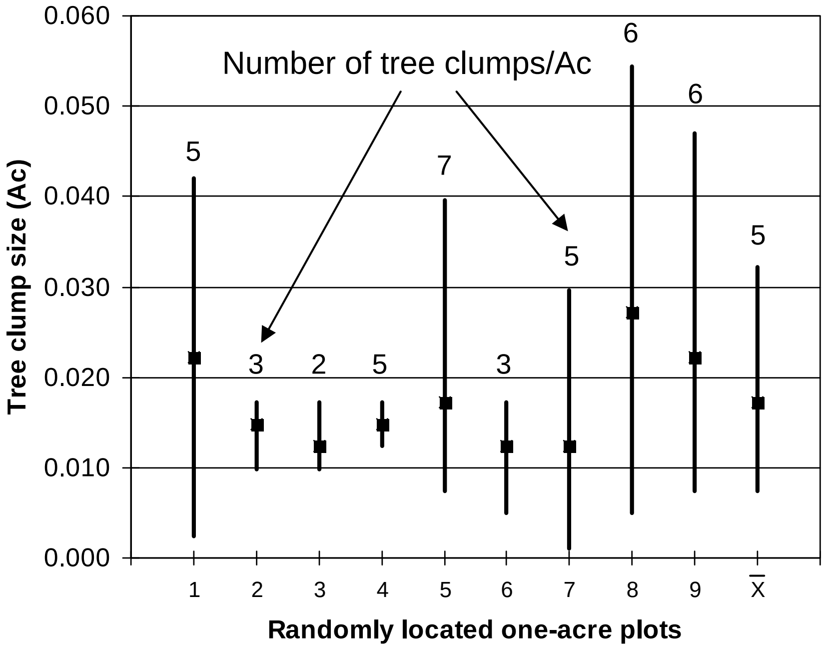


Figure 4—The maximum, mean, and minimum size of ponderosa pine tree clumps and their respective means occurring in a mature stand of ponderosa pine growing on the Boise Basin Experimental Forest in southern Idaho and the number of clumps occurring per acre. These values are displayed for nine randomly located plots as are the means ($\bar{X}$) of the nine plots (Graham and Jain 2005).

observed similar distributions in California and Oregon (also see White 1985). Trees within clumps tend to have fewer limbs and be smaller in diameter while trees on the clump perimeters tend to be larger and lean away from the clump center (*fig. 3*). Even though it appears that space between clumps is not occupied, the root systems of trees within the clump most often occupy this space (Pearson 1950) (*fig. 7*).

Not only do ponderosa pine forests exhibit variation in the standing tree structure, but they also exhibit considerable variation in the amount of coarse woody debris (CWD) that is associated with soil productivity. For example, on ponderosa pine PVTs of Arizona, CWD associated with soil productivity range from 11 to 29 Mg ha^{-1} (5 to 13 tons ac^{-1}). In contrast on Douglas-fir PVTs in Montana, CWD associated with soil productivity ranged from 10 to 55 Mg ha^{-1} (5 to 25 tons ac^{-1}) (Graham and others 1994) (*fig. 8*). In general, ponderosa pine forests in which fires were frequent tended to maintain and use less CWD than those with less frequent fires. This open, clumpy forest structure along with the presence of some CWD benefits many wildlife species and are relatively resistant to stand replacing fire events (Graham and others 2004, Long and Smith 2000, Reynolds and others 1992) (*figs. 4, 5, 6, 7, 8*).

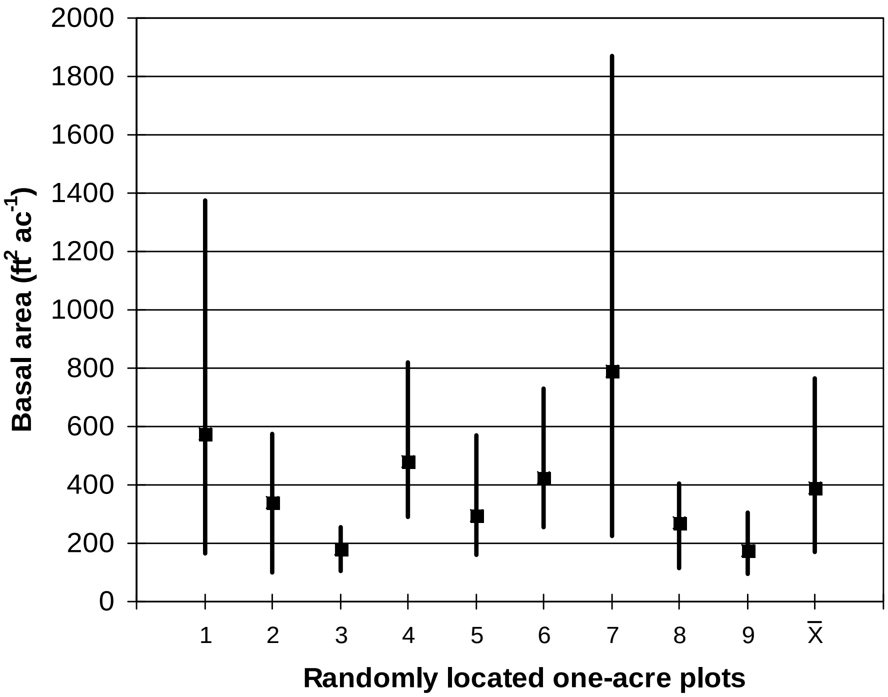


Figure 5—The maximum, mean, and minimum basal area of mature ponderosa pine trees occurring in clumps defined by trees with overlapping crowns growing on the Boise Basin Experimental Forest in Southern Idaho. These values are displayed for nine randomly located plots as are the means ($\bar{X}$) of the nine plots (Graham and Jain 2005).

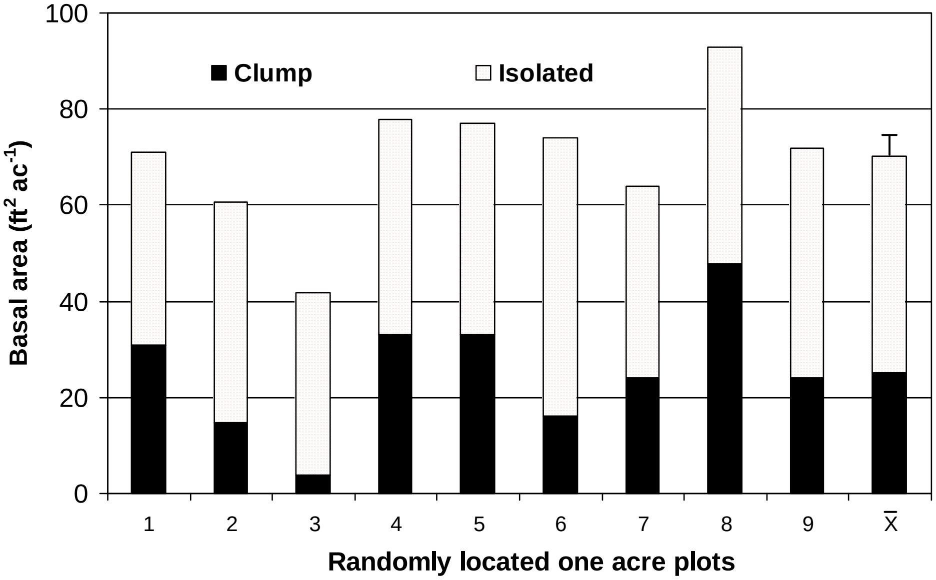
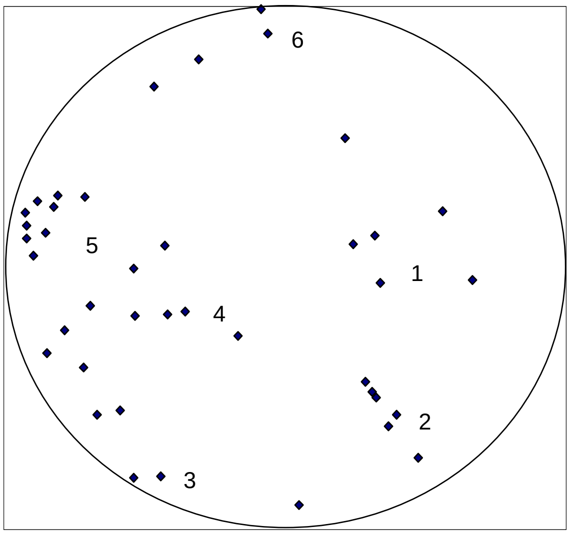


Figure 6—The basal area of mature ponderosa pine occurring within clumps (Clump) defined by trees with overlapping crowns and among trees not associated with clumps (Isolated) growing on the Boise Basin Experimental Forest in southern Idaho. These values are displayed for nine randomly located plots as are the means ($\bar{X}$) of the nine plots. The bar represents the standard error of the mean for the total mean basal area (Graham and Jain 2005).

Ponderosa Pine Forests as Fuel For Wildfires

Another way to characterize forest structure and composition is to describe how different forest components interact and influence fire intensity and severity. For example, forest structure can be divided into fuel bed strata where each level differentially influences combustion, fire propagation and spread, and fire effects (*fig. 9*). Canopy fuels (tree crowns), ladder fuels (midstory trees) and shrub layers contribute to crown fires. Low vegetation (low shrubs and grasses), down woody material (tree branches, boles), and ground fuel (needles, leaves, rotten wood, and litter on the forest floor) contribute to surface fires. Forests composed of mature ponderosa trees are susceptible to the accumulation of large amounts of organic material at the base of trees from both needle fall and bark slough (*fig. 3*). Also, woody material covering the forest floor, which is less than 7.6 cm (3.0 in.) in diameter, is the greatest contributor to the spread of surface fires while large wood (> 7.6 cm, 3.0 in.) and ground fuels tend to favor smoldering fires (Graham and others 2004, Peterson and others 2005). When burning, the intensity (flame length) of these fires determines the potential for the tree canopy to be ignited and most often determines if crown fires would likely be sustained (Graham and others 2004, Peterson and others 2005).



The size of the tree clumps and the basal area within the clumps

Clump	Size (Ac)	Basal Area (Ft^2 Ac^{-1})
1	0.030	406
2	0.032	315
3	0.010	261
4	0.004	366
5	0.034	116
6	0.054	152

Figure 7—A map showing an example of the spatial distribution of mature ponderosa pine growing on the Boise Basin Experimental Forest in southern Idaho (■ = trees in clumps defined by trees with overlapping crowns, ♦ = isolated trees). The circle represents a randomly located one-acre plot (Graham and Jain 2005).

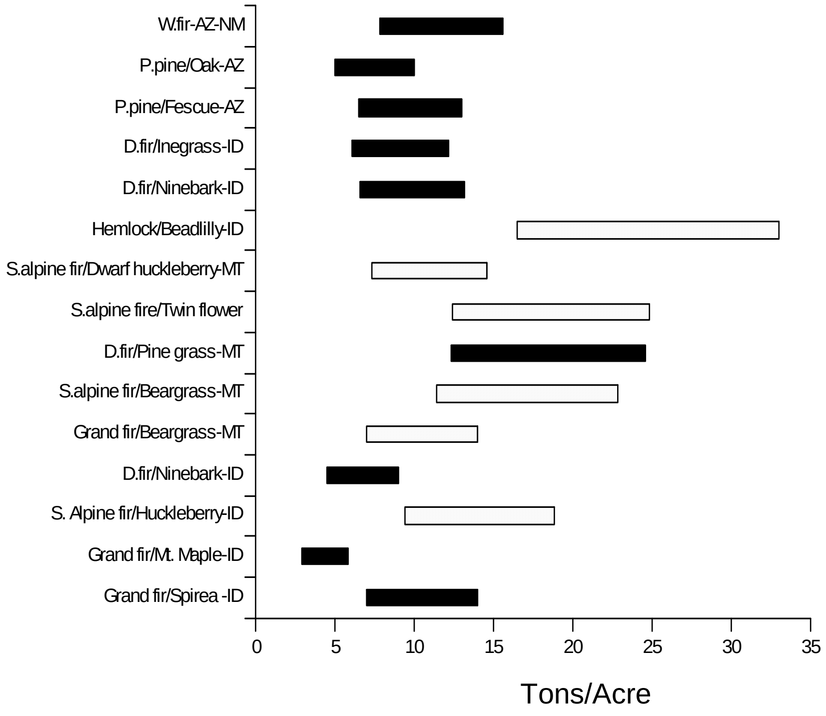


Figure 8—The amount of coarse woody debris recommended by Graham and others (1994) for maintaining soil productivity after disturbance for selected potential vegetation types of the Rocky Mountains. The solid bars represent types in which ponderosa pine is well represented. NM=New Mexico, AZ=Arizona, ID=Idaho, and MT=Montana).

Changes in Ponderosa Pine Forests

With the advent of fire exclusion, animal grazing, timber harvest, and climate cycles on the moist potential vegetation types (e.g. grand/white fir), ponderosa pine is being succeeded by Douglas-fir, grand fir and/or white fir (*figs. 10, 11*) (Graham and others 2004, Gruell and others 1982, Smith and Arno 1999). The accumulation of fire intolerant vegetation, dense forest canopies, with homogenous and continuous horizontal and vertical structures are developing thus creating forests favoring crown fires rather than low intensity surface fires that historically occurred (*fig. 9*) (Arno and Brown 1991, Dodge 1972, Peterson and others 2005, Van Wagner 1977). Within the Inland Northwest, the extent of mid-seral (e.g., Douglas-fir) vegetation has increased by nearly 3.2 million ha (8 million acre) and the extent of single storied mature vegetation (e.g., ponderosa pine) has decreased by over 1.6 million ha (4 million ac.) (Hann and others 1997). Another way to view these changes is that the successional processes in some locations have been compressed by a factor of at least 10. For example, ponderosa pine may or may not be succeeded by Douglas-fir in 300 to 400 years within forests historically burned by frequent fires but in many locations Douglas-fir has succeeded ponderosa pine in less than 50 years (*fig. 2*) (Hann and others 1997, Harvey and others 1999, Smith and Arno 1999).

The structural and compositional changes occurring over the last 100 years in the dry forests are most noticeable within the tree and shrub component or the increase of canopy and ladder fuels (*figs. 9, 10, 11*). In addition; to noticeable changes in plant composition and structure, the soils (surface and mineral) in many settings have also changed considerably during the last century (*fig. 12*). The accumulation of organic materials on the soil surface and the frequent changes in their composition (e.g., ponderosa pine litter to true fir litter) can alter ectomycorrhizae habitat and water holding properties (Harvey and others 1999, Harvey and others 2000). For example, when the species within a forest shifts from being dominated by ponderosa pine to being dominated by Douglas-fir and/or true firs, the vertical distribution of ectomycorrhizal short roots is compressed dramatically (*figs. 10, 11*) (Harvey and others 1986). Ectomycorrhizal activities occur deeper in mineral soil of forests dominated by ponderosa pine compared to sites dominated by true firs; on these sites ectomycorrhizal activities occur in shallow organic horizons. In ponderosa pine dominated forests, ectomycorrhizal-mediated nutrient acquisition and turnover are relatively well protected from wildfire damage. However, when fires burn in forests with uncharacteristically deep organic layers on the surface they can have a detrimental impact on ectomycorrhizal activities and the post-fire acquisition and cycling of nutrients (Harvey and others 1999, Harvey and others 2000, Neary and others 1999) (*figs. 9, 10, 11, 12*). These changes in soil microbial activities may increase the likelihood of uncoupling any continuity between current and preceding vegetative communities (Amaranthus and Perry 1994).

Another soil-based change in pine compared to the fir forests is the type and distribution of substrates important for nutrient storage and cycling. There is a gradual shift in the proportion of soil nitrogen reserves and organic matter from mineral layers in pine forests to surface organic layers in fir forests (*figs*. 10, 11). Accumulation of both above-and below-ground biomass from roots, needles, and boles in fir forests accelerates activities of decomposers by increasing and changing the basic substrate they utilize. Because biological decomposition in ponderosa pine forests is more limited than biological production, accumulation of organic material is inevitable, especially in the absence of fire (*figs. 11B, 12*) (Harvey 1994). The result can be accumulations of materials that differ substantially from those occurring in historical pine forests (*fig. 11*). Associated with these changes in litter type and quantity is a likely change in soil surface chemistry, including the presence of allopathic substances with the potential to alter a variety of microbial activities (Rose and others 1983). Forests dominated by ponderosa pine develop brown cubical rotten wood products that are deposited on the forest floor and subsequently are incorporated into the mineral soil. These products can persist in soils for hundreds of years and during that time provide soil structural and nutritional benefits. In contrast, grand fir/white fir develop white rotten wood products that are dispersed in soil relatively rapidly (decades) shortening their contribution to soil productivity (Harvey and others 1987, Larsen and others 1979).

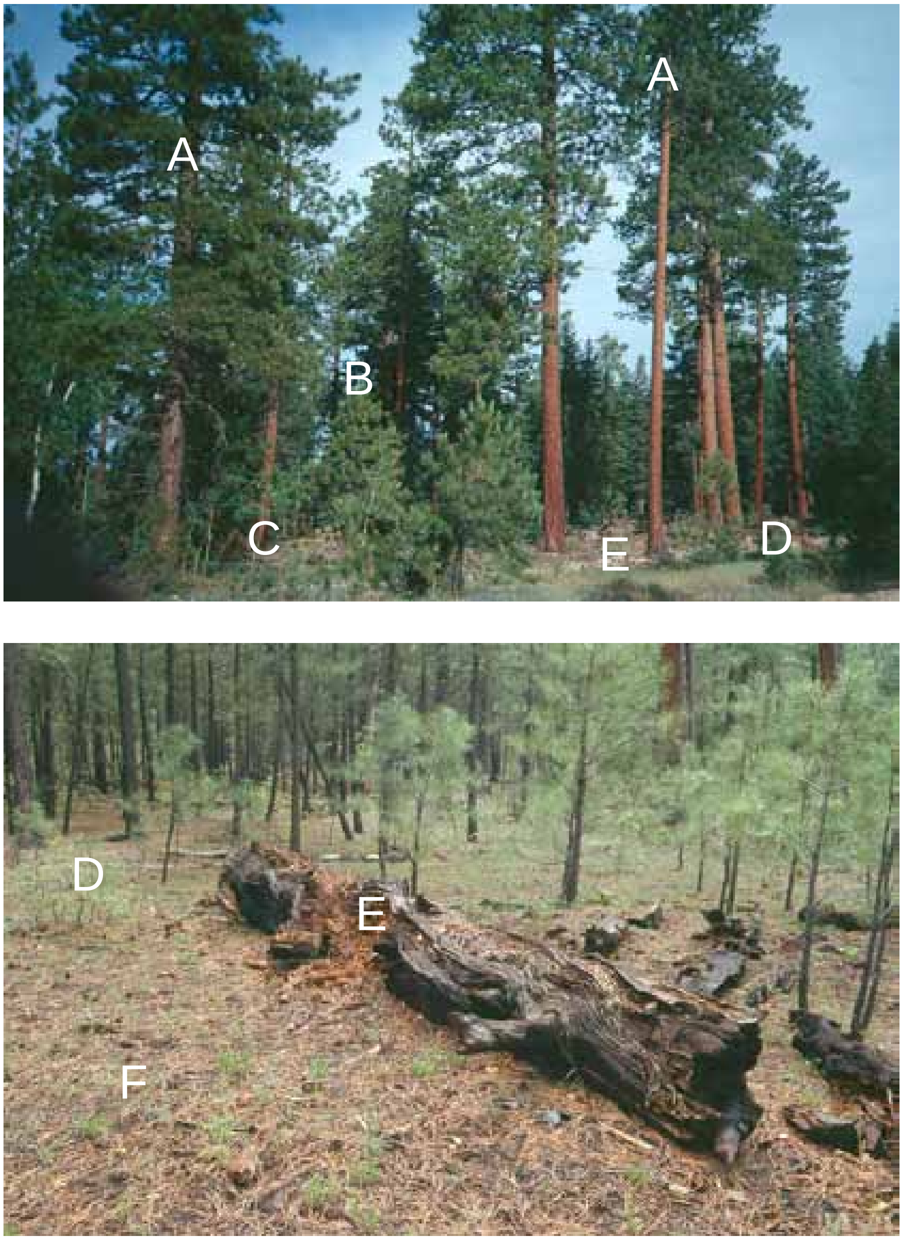
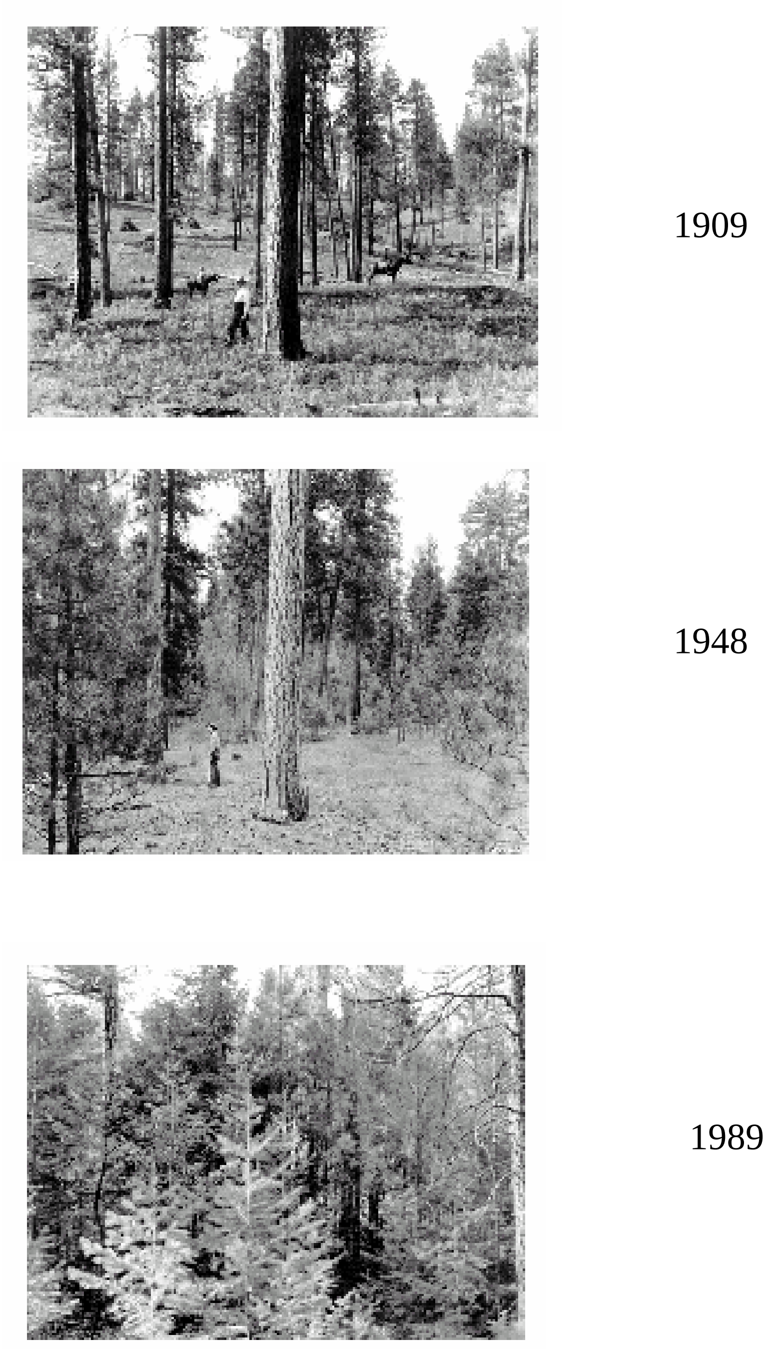


Figure 9—Fuelbed strata have different implications for combustion environment, fire propagation and spread, and fire effects. The canopy (A), ladder fuels (B) and shrub layers (C) contribute to crown fires. Low vegetation (D), woody fuel (E), and ground fuel (F) contribute to surface fires. Woody fuel (E) and ground fuels (F) are most often associated with smoldering fires and residual combustion that can transfer large amounts of heat deep into the soil (Sandburg and others 2001, Graham and others 2004).



1909

1948

1989

Figure 10—Forest development on the Bitterroot National Forest in Montana in a ponderosa pine stand after harvest (1909) in which fire was excluded since 1895. Note the changes in vertical arrangement and horizontal continuity in forest stand structure. In general many of today's ponderosa pine forests contain higher densities of fire-intolerant species and suppressed trees than historical forests (Gruell and others 1982, Smith and Arno 1999, Graham and others 2004).

Figure 11—Photograph A shows the multiple canopy layers developed in a stand in Utah in which harvesting removed much of the ponderosa pine and fire has been excluded allowing abundant white fir trees to develop. Photograph B shows a ponderosa pine stand in which fire has been excluded, the development of multiple canopy layers, and a deep layer of needles and other organic material covering the forest floor (Graham 2003).

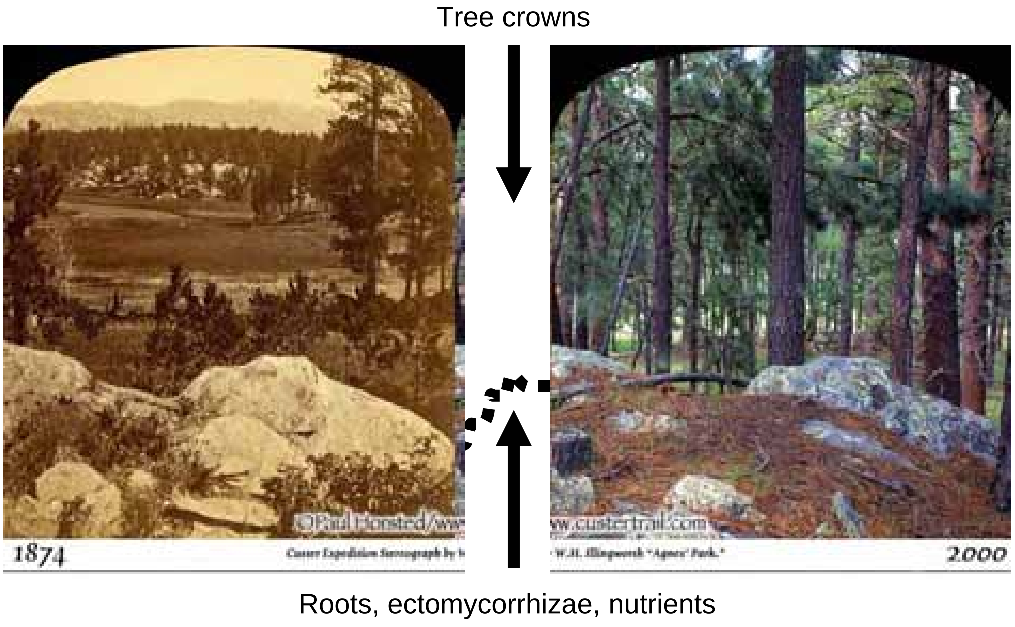


Figure 12—When fires are excluded from ponderosa pine forests organic layers tend to accumulate and tree roots, ectomycorrhizae, and nutrients also tend to concentrate in these layers. Note the contrast between the amount of organic material on the forest floor when General Custer came through the Back Hills in 1874 and the amount that has accumulated around the rocks in the photo in 2000. Photos courtesy of Paul Horsted/custertrail.com (Grafe and Horsted 2002).

Forests dominated by grand fir or white fir tend to concentrate both nitrogen and potassium in their foliage which often extends to the soil surface (*fig. 11*) (Mika and Moore 1990, Moore and others 1991). In general, the combination of a low canopy structure with nutrients and microbial activities concentrated in or near the soil surface make both of these critical ecological resources susceptible to both mechanical and fire destruction (*fig. 11*). Shifting from a few large stems of ponderosa pine to many Douglas-fir stems will maintain brown rotten wood in the system, but these conditions tend to sequester substantial nutrient resources in recalcitrant organic material which is slow to release, thus making these resources susceptible to loss from wildfire or timber harvest activities (Jurgensen and others 1997) (*fig. 10*).

In general, historical ponderosa pine forests were likely well matched to soil resources, relatively resistant to detrimental fire effects, well adapted to wide ranges of site and short-term climate variation, subject to modest (largely beneficial) insect and pathogen mortality, and could be considered long-lived and relatively stable (*fig. 10*). In contrast, forests that were dominated by ponderosa pine and are now dominated by Douglas-fir, grand fir or white fir are probably not well matched to soil resources and are also not likely resistant to the wide range of site and climate variation found within the dry forests (*figs. 10, 11*). In turn, they are often subject to high insect and pathogen mortality and can not be considered either long-lived or stable (Harvey and others 1999).

Exotic Species

The introduction of non-native plant species can cause the decline of native species and pollinators, alter fire regimes, disturb nutrient cycling, and disrupt hydrologic processes (Chong and others 2003, Galley and Wilson 2001). Within the dry forests, some of the more notable invasive species include cheatgrass (*Bromus tectorum* L.), leafy spurge (*Euphorbia esula* L.), Canada thistle (*Cirsium arvense* (L.) Scop.), and spotted knapweed (*Centaurea biebersteinii* DC.) (Hann and others 1997). Cheatgrass aggressively colonizes and invades exposed soil in the dry forests and tends to rapidly develop in the spring and then quickly dry. Annually, the continuous and dense covers of dead grass provide extremely flammable fuels that facilitate more frequent and severe wildfires than those that occurred historically. Because of its interaction with fire, cheatgrass has made some of the successional pathways in the dry forests of the western United States indeterminate (Bradley and others 1992b, Everett 1987, Graham and others 1999b). Cheatgrass is palatable during a portion of its annual development; subsequently it is not always considered a noxious weed. Therefore, even though mulch used in vegetation seeding programs (such as after wildfires) may be weed free, it may contain cheatgrass seed, thereby inadvertently spreading this exotic (Chong and others 2003, Robichaud and others 2003). Exotic plant species have readily spread throughout the Inland West. All 97 counties in the interior Columbia Basin contain cheatgrass and Canada thistle (even some of the most remote locations in central Idaho), 92 counties contain spotted knapweed, and 72 counties contain leafy spurge (Hann and others 1997).

Wildfires in Current Ponderosa Pine Forests

Ponderosa pine forests appear to have evolved with fire and many authors have suggested that their composition and structure are dependent on the vegetation's relations with fire (Covington and Moore 1994, Everett and others 2000, Hann and others 1997, Harvey et al 1999). However, when wildfires burn altered ponderosa pine forests, the extent or area burned is similar to historical times but they tend to burn more severe (Graham 2003, Graham and others 2004, Hann and others 1997, Kaufmann and others 2001). They often kill large continuous expanses of vegetation, consume the forest floor, volatilize nutrients, provide for exotic species introductions, increase soil erosion, and, in general, create forest conditions that may not be favorable to society (*fig. 13*). The long-term consequences are not well understood for issues such as water quality and wildlife habitat. Nor will the sense of place that forests often provide which, in some cases, will not be replaced for many generations (Galliano and Loeffler 1999, Kent and others 2003, Schroeder 2002).

Some of the notable recent wildfires that burned the dry forests include at least 16 fires that burned in the Bitterroot Valley of western Montana in 2000, the Hayman Fire along the Front Range of the Rocky Mountains in Colorado in 2002, the Rodeo-Chediski Fire in Arizona in 2002, and the Biscuit Fire in Oregon in 2002 (USDA 2000, Graham 2003, Graham and others 2004). All of these wildfires exemplified large (≈40,000 to 200,000 ha) and severe events. The Rodeo-Chediski fire in Arizona severely burned over 6,000 ha (15,000 acre) in 15 minutes and moved on a broad front at over 6.4 kph (4 mph). The Hayman Fire severely burned over 19,000 ha (47,000 acre), much of it in one day (*fig. 14*) (Finney and others 2003, Robichaud 2003). Both of these fires burned in ponderosa pine forests that were accustomed to non-lethal or mixed fires. Also, what is striking about all of the fires occurring in the dry forests in recent years (2000-2004), is the large impact they had on real property by burning homes and businesses along with impacting municipal watersheds and the tourism industry (Graham and others 2004, Kent and others 2003, Robichaud and others 2003). The fires occurring in the dry forests and the hazard they pose to human values influenced the development of the Healthy Forests Initiative and the passing of the Healthy Forests Restoration Act (USDA 2004).

Dry Forest Treatments

Dry forests, particularly those containing ponderosa pine, tend to occur at low elevations and along rivers and travel ways, resulting in some of the earliest silvicultural practices being developed and used in these forests (Meyer 1934, Pearson 1950). This long history of treatment application and the knowledge gained through these practices can be applied and adapted to meet present and evolving management objectives. Severe crown fires are often considered the primary threat to these forests (Graham and others 2004). Because of the many changes that have occurred in the dry forests, even low intensity surface fires can now damage soils, weaken or kill overstory trees, and provide an ignition source for homes and other property (Cohen and Stratton 2003, Hungerford and others 1991).

Figure 13—A photograph of the intense and severe Rodeo-Chediski Fire that burned in Arizona in 2002. This ponderosa pine forest was historically burned by low intensity surface fires but singly and in combination fire exclusion, timber harvest, climate change, and livestock stock grazing contributed to forest changes that facilitated this uncharacteristically severe fire.

The current understanding of fire behavior in dry forests indicates that a crown fire begins with a transition from a surface fire to the ignition of the canopy (*figs*. 9, 13, 14) (Finney and others 2003). Therefore, management activities can target specific fuel strata to disrupt the continuity of fuels from the soil surface to the crown and also target the horizontal arrangement of trees to disrupt the progression of fires especially from tree crown to tree crown (Graham and others 1999a, Scott and Reinhardt 2001) (*fig* 9). In the dry forest it appears the most effective strategy for reducing crown fire intensity and severity is (1) reduce surface fuels, (2) increase crown base heights, (3) reduce canopy bulk density, and (4) reduce continuity of the forest canopy (Graham and others 1999a, Cruz and others 2003, Scott and Reinhardt 2001, Van Wagner 1977). Generally, fuel treatments are beneficial for modifying both fire behavior and fire severity under normal weather conditions (Albini 1976). However, under extreme weather conditions (e.g., low humidity, high winds) fires can overwhelm most fuel treatments resulting in intense and severe fires (Finney and others 2003). Also, there is no guarantee that the combination of canopy and surface fuel treatments will reduce the risk of intense wildfires or immunize property losses when a fire does occur. Nevertheless there is the potential that treated areas will experience lower burn severity than untreated areas (Finney and others 2003, Graham and others 2004) (*fig. 14*).

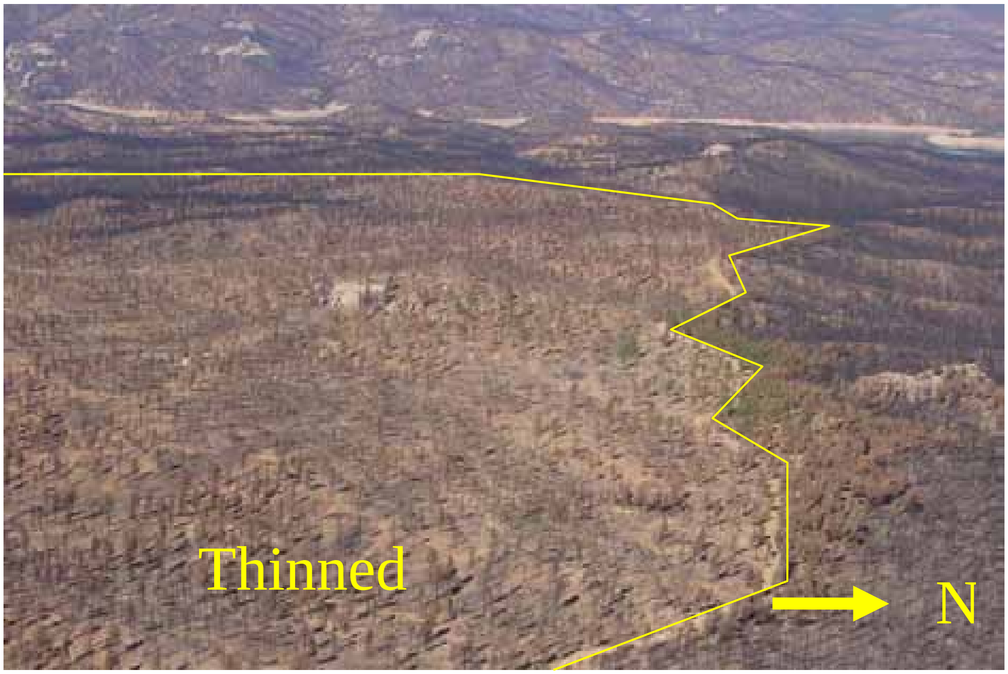


Figure 14—This photograph shows a ponderosa pine forest that was thinned in 2001 reducing the continuity and bulk density of the tree canopy. The surface fuels were in place when the Hayman Fire burned (moving from the left to the right in the photo) the area in on June 9, 2002. Even though the trees were killed the brown needles will mulch the forest floor when they fall reducing soil erosion compared to the areas in which all of the needles were consumed in the fire (Finney and others 2003, Pannkuk and Robichaud 2003).

Prescribed fire has long been used in the dry forests for cleaning the forest floor, increasing canopy base heights, and preparing seed beds for regeneration (Barrett 1979, Pearson 1950, Weaver 1943). Fire can alter multiple fuelbed characteristics by decreasing ground and surface fuels and decreasing ladder fuels (*fig. 9*) (Graham and others 2004, Peterson and others 2005). Mechanically thinning trees can also modify ladder fuels and decrease or alter the continuity of canopy fuels. Mechanical treatments offer more control in creating the desired forest structures than prescribed fire and they are less uncertain because of the inherent risk of prescribed fires escaping (Graham and others 1999a, Scott and Reinhardt 2001, Peterson and others 2005). However, if surface fuels created by mechanical treatments are not removed or treated they can exacerbate both the risk and hazard associated with fuel treatments (Brown and others 1977). This was exemplified on a portion of the Hayman Fire where the canopy bulk density and continuity had been greatly reduced within a treatment area, but because the surface fuels were not treated, the area burned intensely (*fig. 14*), yet the tree severity of the fire was less, as indicated by the presence of brown needles, compared to areas that were not thinned, where all needles were consumed leaving black branches. What is significant about this treatment is that when these brown ponderosa pine needles fall they will mulch the forest floor reducing soil erosion (interrill) by 60 percent compared to bare mineral soil. If they had been Douglas-fir needles, erosion would have been reduced by 80 percent (Pannkuk and Robichaud 2003).

Forests continue to regenerate, develop, and die, meaning that depending on how fast these processes occur, fuel treatments aimed at remediation of wildfire fire hazard need to be continued and/or maintained. For example, surface fuel treatments occurring within a year prior to the Hayman Fire in Colorado had a significant impact on the fire's progression. In contrast, areas with fuel treatments conducted seven years prior were intensely and severely burned (Finney and others 2003). As shown by the large fires that have occurred in the western United States during the last few years, most fuel treatments have local impacts on a fire's intensity and severity but for the most part made little impact on the overall outcome of the fire (Finney and others 2003). Strategically placed fuel treatments may have greater potential for altering the spread of large wildfires than randomly placed fuel treatments. Fuel treatments are most effective if they are encountered by a wildfire early in its development (Finney 2001, 2003, Finney and others 2003).

Conclusions

Because of its wide distribution, ponderosa pine occurs in many diverse forests and it has been studied and managed for over a century (Barrett 1979, Meyer 1934, Pearson 1950, Shepperd and Battaglia 2002). Even though ponderosa pine forests are often considered simple in terms of species and structure they can be rather complex. The species grows on a variety of biophysical settings ranging from those where ponderosa pine is the late seral species and the only conifer species possible to those where western redcedar is the late seral species and over seven conifer species can occur (Cooper and others 1991). With the interaction of wind, snow, ice, diseases, insects, and fire, an abundance of successional pathways are possible (Bradley and others 1992a, Smith and Fischer 1997). The resulting species and structural mosaics can be rather intricate occurring within spatial scales as small as those defined by tree groups to large ones defined by landscapes. Similarly, temporal scales associated with the disturbance, regeneration, and development ponderosa pine forests can range from months to centuries (Hann and others 1997, Long and Smith 2000, Reynolds and others 1992).

Through the advent of fire exclusion, climate cycles, grazing, and harvesting ponderosa pine forests have changed (e.g., species composition, soil characteristics, horizontal and vertical structure) significantly during the last 100 years (Covington and Moore 1994, Hann and others 1997). Nevertheless, even with these changes there are tremendous opportunities for restoring these forests to their structure and composition reminiscent of historical conditions (Graham and Jain 2005, Jain and Graham 2005, Long and Smith 2000). Some key treatments such as reducing the quantity and continuity of surface fuels, increasing crown base heights, and decreasing crown continuity and density can lessen the risk of loss from extreme wildfire events (Graham 2003, Graham and others 2004, Peterson and others 2005 Scott and Reinhardt 2001). Both prescribed fire and mechanical techniques can be used to change the forest structure and encourage the regeneration and development of ponderosa pine, especially as an early seral species, within interspersed mosaics. If the structure were changed, such forests may be resistant and resilient to native insects and diseases, uncharacteristically severe wildfires, and be beneficial to many wildlife species (Graham and others 1999a, Graham and others 1999b, Graham and others 2004, Long and Smith 2000, Reynolds and others 1992).

Ponderosa pine forests extend throughout the western United States and Canada and they contain some of the most prized scenery and special paces in the West. The presence and existence of ponderosa pine reinforces these special qualities (Grafe and Horsted 2002, Gruell and others 1982, Smith and Arno 1999). Forests that contain large trees with yellow bark and emit that unique and pleasant odor on a warm summer day only attributable to ponderosa pine, provide the essence of being in or part of a forest. These attributes and the wildlife they support bring people solace and enjoyment when visiting or living in these settings. Because of these and other values to society, ponderosa pine forests often evoke strong feelings in people as to their management and conservation.

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An Overview of Key Silvicultural Information for Ponderosa Pine[1]

John Fiske[2] and John Tappeiner[3]

Abstract

This paper provides a selected list of classical references for the important silvicultural findings for ponderosa pine, and categorizes some of the key current literature, as well as some of the older, lesser known but important literature. The paper also provides some history of scientific developments, and sources of further information.

Introduction

In this paper we review what we consider to be the key silvicultural information for ponderosa pine (*Pinus ponderosa* P. & C. Lawson). Our experience with ponderosa pine and consequently much of our information comes from Oregon and California. We did not attempt to examine the entire literature on this species, which is enormous! Rather, we focused on what we think is the most important information in fifteen categories. We included older information that is often overlooked, probably because it is not in bibliographic electronic databases. Also, proceedings of at least two symposia that are devoted primarily to ponderosa pine (Baumgartner and Lotan 1988 and Robson and Standiford 1983) and summaries of ponderosa pine silvics (Barrett 1979 and Oliver and Ryker 1990) and silvicultural systems (Schubert 1974, Oliver and others 1983, Ryker and Losensky 1983, and Boldt and others 1983), Ronco and Ready (1983) include very helpful information. Syntheses of regeneration practices are also very important (Schubert and Adams 1971, Cleary and others 1978, Hobbs and others 1992)

Our review is divided into categories under the twin headings Reforestation and Timber Stand Improvement and Stand Growth, and we provide an abstract summarizing the main points in these parts of the literature. The categories overlap. For example, information on bark beetles occurs under stand growth and density as well as under insects, pathogens, animal, and snow damage.

Undoubtedly we omitted important information that ought to be included. Perhaps this review can be considered a work in progress.

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[1] An abbreviated version of this paper was presented at the symposium on Ponderosa Pine: Issues, Trends, and Management, October 18-21, 2004, Klamath Falls, Oregon.

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Regeneration and Timber Stand Improvement

Fully recognizing that regeneration and timber stand improvement (TSI) are best thought of as a complete system, and that failure occurs if any single component fails, for the purposes of this presentation, we divided “regeneration” and “TSI” into five categories: natural regeneration, seed collection and handling, nursery practices, site preparation and release, and precommercial thinning.

Natural Regeneration.

Natural regeneration, with seed-tree or shelterwood systems, has been used effectively in many parts of ponderosa pine’s natural range (for example, Heidmann 1988, McDonald 1976a, 1976b, 1976c, McDonald and Abbot 1994, Muelder and others 1963, Roy 1983, Shearer and Schmidt 1970, 1971), especially if sites are well-prepared, include rodent control, and coincide with good seed crops. Pearson (1923) was the earliest example of the rich scientific literature we found. Advanced regeneration can also be considered as natural regeneration.

Seed Collection and Handling

In contrast to the other major western conifer species, proper collection and seed handling methods for artificial regeneration of ponderosa pine were comparatively easy to determine and are well established (Fowells and Schubert 1956). Early work (in the 1940’s and 1950’s) focused on germination and cold storage. The most significant single development was the establishment of seed zones, elevational bands, and seed transfer guidelines, beginning in the 1940’s in California (Fowells 1946, Buck and others 1970), later elsewhere (for example, starting in the 1950’s in Oregon, Roy 1955).

Nursery Practices

Volumes have been written over the last 5 decades about culturing ponderosa pine seedlings, a subject that now is very well understood and practiced. We focus on just three key developments. The first was the development of fumigation (initially primarily methyl bromide) to control root diseases in bare-root seedlings (Bega and Smith 1960, Smith and Bega, 1966). Although ponderosa pine is somewhat resistant to seedling root diseases compared to most other commercial western conifers, fumigation use significantly improved production efficiencies for ponderosa pine. The second key development was the concept of how nursery practices and seed source (genetics) affect the potential for growing new roots upon out-planting (root growth capacity or root regeneration potential) and lifting windows (Jenkinson 1980, Stone 1955, Stone and Benseler 1962, Stone and Schubert 1959a 1959b, and Stone and others 1963). The third key set of developments was the enormously successful technology of container nurseries, starting in the 1960’s (Tinus and McDonald 1979, Tinus and others 1974). This technology is best summarized in The Container Tree Nursery Manual (Landis and others 1989, 1990a, 1990b, 1992, 1995). Container nursery technology continues to improve in the western United States and Canada, primarily by private nurseries, often in cooperatives.

Site Preparation and Release.

There is considerable literature on the theory and practice of controlling unwanted vegetation (for example the Proceedings of the Annual Forest Vegetation Management Conference, starting in 1981, and Walstad and Kuch 1987). The scientific basis for predicting ponderosa pine responses to effective control of competing vegetation is well established. Practices have undergone continual development, in part to reduce unit costs, and to reduce controversies over herbicide use. Just about every conceivable alternative to herbicide use has been tried somewhere during the last three decades (including dynamite for site preparation!).

We focus on three key aspects: the development and application of herbicides, the USDA Forest Service National Administrative Study, and the scientific understanding of the interaction between site quality and tradeoffs between release vs. precommercial thinning.

Herbicide use started in the 1950's with adaptation of agricultural aerial application techniques of the phenoxy herbicides. Later, appropriate use of a much broader range of herbicides was established on a scientific basis, including human health and ecological risk considerations. Additions to the scientific literature continue, and (in part because of legal challenges) comprehensive risk assessments are scheduled for almost continuous updating. The current risk assessments for hexazinone, sulfometuron methyl (OUST), imazapyr (Arsenal®, Chopper®, and Stalker®[4] formulations), glyphosate, and triclopyr are available online (SERA 1997, 1998a, 1998b, 2003a, 2003b).

The National Administrative Study focused on release in northern California (a 20-year-plus study), and continues to establish much of the long-term scientific basis for herbicide and non-herbicide treatments (Fiddler and McDonald 1984). Comparable long-term study results for site preparation and release are becoming available elsewhere in California and in Oregon.

Our third focus is on the relationship between determining needs for release vs. precommercial thinning, as influenced by site quality. Numerous studies have quantified effects of shrub competition and release treatment on conifer growth (Baron 1962, Crouch 1971, 1979, Doescher and others 1989, Fiddler and McDonald 1984, Lanini and Radosevich 1986, Tappeiner and Radosevich 1982, White and Newton 1989). When investment funds are limited, and only one kind of treatment can be done, which treatment should receive priority, and how does site quality affect the decision? Bill Oliver compared results of both kinds of treatments on a high-quality and a low-quality site in northern California (McDonald and Oliver 1984), and established the important principle that on low-quality sites, inter-tree competition in ponderosa pine plantations is insignificant, compared to competition between the trees and shrubs. That is, precommercial thinning is a wasted investment unless the thinning follows, or is done concurrently with, effective release treatments to control shrubs. On such sites, lack of effective release treatments can result in unacceptably high tree mortality rates. In contrast, on high-quality sites, inter-tree competition can be greater than tree-shrub competition, so a single precommercial thinning treatment can yield a better investment return compared to a single release treatment.

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[4] The use of trade or firm names in this publication is for the reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

Precommercial Thinning

Practical experience with early ponderosa pine plantations (early part of the 20th century) and widely-distributed spacing studies led to commonly-used residual stocking levels of approximately 100 to 250 trees per acre, depending on site quality, management objectives (including wood quality), and other factors. Two significant quantitative models for predicting growth of pre-commercial-sized ponderosa pine in plantations have been developed. Oliver and Powers (1978) developed the first quantitative model for spacings of 6, 8, 10, and 12 feet, respectively, for a range of site qualities in northern California. Powers and others (1989) and Ritchie and Powers (1993) described the first quantitative model (SYSTUM-1) to include the effects of shrub competition (also at different tree spacings), based largely on ponderosa pine and Douglas-fir plantation data from southwestern Oregon and northern California.

Stand Growth

Genetics, Autecology.

Conkle (1973) reported on the ponderosa pine elevational study from the Sierra Nevada. In this study, trees from low-, mid and high-elevation seed sources were reciprocally planted at these elevations. Ponderosa pine height growth varied by elevation. Trees from high elevation seed sources grew slower than those from other elevations, and had lower height-to-diameter ratios. Trees from mid-elevation sources appeared to be most productive and can be moved more readily than those from low and high elevations, but they were susceptible to snow damage when planted at high elevations.

Ponderosa pine has the ability to grow roots into rock fissures in the unweathered soil horizons and extract water stored there. Shrubs that grow on the same sites have the same ability; however, Douglas-fir is much less able to use water stored in rock fissures (Zwieniecki and Newton 1994, 1995, 1996).

Growth and Yield

Much early research on this species was focused on growth and yield, and was presented in tabular form. Examples of published yield tables include: Dunning and Reineke (1933), Meyer (1934, 1938), Oliver and Powers (1978). More recently, growth and yield information usually takes the form of computer simulators (e.g. DeMars and Barrett 1987, Hann 1981, Hann 2003, Ritchie and Powers 1993, Wensel and Koehler 1986).

Site productivity, traditionally reflected in site index curves have been developed for many regions within the range of ponderosa pine (Barrett 1978, Biging 1985, Dunning 1942, Lynch 1958, Oliver 1972, Powers and Oliver 1978, Hann and Scrivani 1987, Verdyla and Fischer 1989).

Tree Growth and Stand Density (tree and shrub)—Thinning

Several studies on thinning response and the growth of ponderosa pine at a range of stand densities have been reported in the last several decades. These studies indicate that young, even-age ponderosa pine stands respond to thinning like most other conifer species. Thinning increases diameter growth, and maintains crown lengths (Barrett 1982, Cochran and Barrett 1993, Fiddler and others 1989, McDonald and others 1992, Oliver 1984, 1997). Heavy thinning tended to decrease volume yield per unit area, but at high densities or with light thinning net volume yield was low because of mortality from bark beetles. Mechanical treatments may impact soil characteristics and site productivity (Busse and others 1996, Helms and Hipkin 1986, Helms and others 1986).

These studies indicate that western pine beetles may determine the upper levels of stand density (Oliver 1995). Also snow breakage is another important cause of mortality at high densities. On dry sites shrubs (Ceanothus and Manzanita sp.), may reduce ponderosa pine growth during stand establishment and reduce or delay it even after the ponderosa pine has overtopped the shrubs (Gordon 1962, McDonald and Oliver 1984, Oliver 1984, 1990, Powers and others 2005, Shainsky and Radosevich 1986). It appears that eventually the pines may shade out the shrubs and increase their growth rates.

Fertilization, Stand Growth Effects of Shrubs

Ponderosa pine responds to fertilizers but only after shrub control. Where shrub density is high, control of shrubs appears necessary to provide the water needed for fertilizers to be effective. Under combined shrub control-fertilization, most growth response resulted from removal of shrubs; the direct effect of fertilizers is secondary (Powers and Jackson 1978, Powers and Ferrell 1996, Powers and others 1988, Walker 1999a, 2002, White and Newton 1989).

Measures of Stand Density

Measures of density for stands in the pine region have been developed around stand density index (Reineke 1933). Stockability, (the potential stand density for a site) can be adjusted for specific sites (Dunning and Reineke 1933, Cochran 1992, Cochran and others 1994; Hall 1983, Peterson and Hibbs 1989).

Uneven Age Management

Techniques for developing stocking guidelines for uneven aged stands are based on distributing the desired residual SDI (possibly half of maximum) throughout several diameter classes, resulting in a large variety of possible stand structures. The resulting distribution can be evaluated by calculating the numbers of trees and basal area by diameter class. Shifting different amounts of SDI into various diameter classes can modify the distribution. Thus the method is quite flexible and a large number of structures are possible (Cochran 1992, McDonald and Abbott 1994, Lillieholm and others 1990, O’Hara 1996, O’Hara and Gersonde 2004, Olson and Helms 1996).

Insects, Pathogens, Animal, and Snow Damage

Drought, diseases, bark beetles, and high stand densities, all influence ponderosa pine mortality (Barrett and Roth 1985). Root diseases (*Heterobasidium*) and mistletoes, and high density weaken trees and make them susceptible to bark beetles

(*Dendroctonus*) especially during periods of drought and also to damage by heavy, wet snow (Hawksworth and Wiens 1996, Filip 1986, Filip and others 1989, 1999, Megahan and Steele 1987, Miller and Keen 1960, Scharpf and Bega 1981, Schmid and others 1994, Smith 1982). Bark beetle attacks may indicate trees under stress (Storm and Halvorson 1967, Stoszek 1973).

Old Stand and Tree Management

Early work has provided tree classification systems for determining the vigor of ponderosa pine and its susceptibility to insects (Dunning 1928 and Keen 1936, 1943). Recent work has shown that older ponderosa pine stands and trees can respond to thinning. For example a sustained 1.5 to 2.0 percent increase in tree basal area growth was common for trees (+200yr) in stands previously thinned 15 to 30 years earlier (Dolph and others 1995). Removal of understory trees established after fire cessation in the early to mid 1900s may improve the vigor of old ponderosa pine as well as protect them from fire (Latham and Tappeiner 2002, McDowell and others 2003, Biondi 1996).

Understory Vegetation and Stand Density

Recent studies have documented the interactions between overstory density and understory development (Riegel and others 1992, 1995). At higher levels of overstory density, understory is “shaded out” and its density is reduced in response to scarce water and light. Understory vegetation, however can increase organic matter and nutrients in the forest floor (Busse and others 1996, Harris and Covington 1982).

Fire History and Use of Prescribed Fire/Thinning

The history of fire in ponderosa pine has been studied extensively, and prescribed fire and thinning are being increasingly used to reduce the risk and intensity of fire in ponderosa pine stands (Covington and Sacket 1984, 1986; Covington and Moore 1994; Harris and Covington 1982, Hall 1976, 1983; McNeil and Zobel 1980, Bork 1985). Some important emerging issues include the effect of fire on mortality of large trees, minimization of slash/dead trees after initial prescribed fire, and the effects of fire in stimulating germination of buried seed and vegetative buds (Mutch and Parsons 1998; Parsons and DeBenedetti 1979, Weaver 1959, 1961, White and others 1973). More research and practical experience is needed on the use of fire, and mechanical methods in commercial and precommercial thinning and combinations of treatments to achieve fuel reduction.

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Silvicultural Systems for Managing Ponderosa Pine[1]

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Abstract

Silviculturists have primarily relied on classical even-aged silvicultural systems (the planned series of treatments for tending, harvesting, and re-establishing a stand) for ponderosa pine, with uneven-aged systems used to a lesser degree. Current management practices involve greater innovation because of conflicting management objectives. Silvicultural systems used in the foreseeable future will likely meld traditional systems with greater reliance on variation across the landscape because of differing values and desired outputs. Significant changes in the management of ponderosa pine are reviewed; issues that likely will affect the future management of ponderosa pine systems are listed; and critical gaps in our understanding of ponderosa pine silviculture that may affect our management in the near term are identified.

Introduction

Ponderosa pine (*Pinus ponderosa* P. & C. Lawson) is one of the most widely distributed pines in western North America (Oliver and Ryker 1990). It occurs from southern British Columbia in Canada into Mexico, and from the Pacific Coast along the California-Oregon border as far east as western Nebraska and Oklahoma. Two subspecies are recognized; Pacific ponderosa pine (var. *ponderosa*) ranges along the flanks of the Cascade and Sierra Nevada Ranges as far south as Riverside County in southern California, while Rocky Mountain ponderosa pine (var. *scopulorum* Engelm.) occurs in Idaho, Montana, Utah, and east of the continental divide (USDA 2004).

Ponderosa pine is a major component of three forest cover types (Eyre 1980). The Interior Ponderosa Pine cover type (Society of American Foresters Type 237), the most widespread pine type in the western United States, is composed of pure or mixed stands east of the Cascade-Sierra Nevada crest in northern California, east of the Cascade Range crest in Oregon and Washington, and eastward into the Plains States: Pacific Ponderosa Pine-Douglas-fir cover type (Society of American Foresters Type 244) is composed of mixed stands on eastern slopes of the Coast Range and western slopes of the Cascades in Oregon, extending southward through the Klamath Mountains into northern California. The Pacific Ponderosa Pine Type (Society of American Foresters Type 245) is essentially pure stands of ponderosa pine in the Klamath Mountains of Oregon and Washington, extending southward along the Sierra Nevada into central California. Ponderosa pine also occurs as a minor component in at least 23 other cover types.

Of all the North American forest types, ponderosa pine forests offer the greatest opportunity for meeting multiple objectives such as water, minerals, timber, forage,

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wildlife habitat, recreation, subsistence or spiritual purposes, firewood, homesites, and scenic beauty (Morgan 1988). Given that ponderosa occurs across diverse landscapes with broad ecological amplitude, and the relative ease in accessibility at lower elevations has led to greater intensity of management activities for multiple objectives compared to cover types occurring at higher elevations, it is not surprising that numerous silvicultural systems for stand management on public lands have been applied. In this paper, the traditional or conventional silvicultural systems recommended for managing ponderosa pine on public lands are summarized, recent changes in ponderosa pine silviculture and the underlying causes for the change are examined, future management actions and their effect on ponderosa pine sites are hypothesized, and the critical gaps in our understanding of ponderosa pine silviculture are identified.

Traditional Silvicultural Systems

Graham and Jain (2004) traced the role of silviculturists from the late 1800s with the transfer of public lands into homesteads, through the rapid expansion westward of railroads, the creation of national forests, and the advent of timber management and fire suppression for the growth and protection of cities and town during in the late 1800s and early 1900s, into the era of multiple use beginning in the 1960s. Ponderosa pine stands figured prominently in these events because they were relatively easy to access and they yielded high quality timber. With passage of the National Forest Management Act of 1976, silvicultural systems that emphasized multiple values of ponderosa pine stand became increasingly important.

Traditional or conventional silvicultural systems (the planned series of treatments for tending, harvesting, and re-establishing a stand) used for managing ponderosa pine stands include clearcutting, seed tree, shelterwood, single tree selection, and group selection. These systems, named after the regeneration method, are fully defined in Helms (1998). Clearcutting is the method of cutting essentially all existing trees, producing a fully exposed microsite for the development of a new age class, with regeneration from either natural seeding, direct seeding, or planted seedlings. A variant is overstory removal, when the presence of advance regeneration is sufficient to warrant no additional action other than the complete harvesting of the overstory. The seed tree method is the cutting of all trees except for a limited number of widely dispersed trees retained for seed production. The shelterwood method is the cutting of most trees, leaving only those needed to produce sufficient shade or insulation from temperature extremes to produce a new age class in a moderated microenvironment. Variants are based on the distribution of the residual trees left for shading. Selection methods involve the removal of some trees in all size classes either singly (single tree selection) or in groups (group selection), with regeneration occurring in the resulting gaps and continued growth of all remaining trees. Clearcutting, seed tree, and shelterwood methods result in the establishment of even-aged stands while selection methods maintain uneven-aged stands. The coppice method may be used for species that sprout from the stump or produce root suckers, thus is not applicable for managing ponderosa pine.

Ponderosa pine stands have been managed to provide a wide range of outputs and values by using these silvicultural systems, including timber, habitat for large game and avian species, protection of wildland resources from damaging insects, disease, or wildfire, quality forage for domestic ungulates, and water. Traditional or conventional silvicultural systems commonly recommended for managing ponderosa pine are summarized (*table 1*) for four regions: 1) northern Rocky Mountains, including western Montana, northern Idaho, and extreme northeastern Washington; 2) Pacific Northwest, including eastside forests of Washington and Oregon; 3) westside California, including southwest Oregon; and 4) eastside California.

In general, the even-aged silvicultural system based on clearcutting was frequently recommended for use across all four regions through the early 1980s. This recommendation likely was in response to a general preference for managing ponderosa pine stands with a primary emphasis on timber production that featured conversion of old stands to more vigorously growing young stands. Other even-aged systems, including seed tree and shelterwood methods, were clearly secondary in preference. A special variant of even-aged management practices in the Blue Mountains of northeastern Oregon and southeastern Washington was either a complete or partial overstory removal of old-growth ponderosa pine followed by management of the advance regeneration that developed after decades of fire exclusion or suppression. This practice had at least three underpinnings: 1) the overstory ponderosa pine was growing slowly and had high susceptibility to bark beetles; 2) the overstory ponderosa pine had very high value and revenue that was generated could be used for other land management projects within the timber sale area; and 3) management of the advanced regeneration avoided the high cost of tree planting, it maintained a green, forested setting thus avoided the visual impacts associated with clearcutting, and it capitalized on the initial growth of the understory trees (Powell 1994). Selection systems were restricted in application to special areas such as high visibility stands surrounding recreation areas.

Clearcutting and other even-aged methods continued to be the preferred means of managing ponderosa pine stands into the early 1990s, except in the Blue Mountains where overstory removals prevailed. Clearcutting and planting was recommended when timber management was the primary objective because it required less sale administration, resulted in lower costs of wood production, and probably had higher wood productivity because of fewer difficulties in controlling tree density compared to uneven-aged management (Helms and Lotan 1988). At the same time, uneven-aged systems became increasingly popular in the northern Rocky Mountains and the Pacific Northwest for protecting visuals, for managing recreation sites, and for providing wildlife habitat. The advisability of uneven-aged systems for ponderosa pine remained in question throughout California.

More recently, uneven-aged silvicultural systems for managing ponderosa pine were frequently suggested, with emphasis on group selection. A further refinement is termed “free selection”, described as the combination of group and single tree selection systems with reserve trees left in all structural stages, and recommended for creating clumpy and irregular stand structure that is preferred for species such as the northern goshawk (*Accipiter gentiles*) (Graham and Jain 2004). Even-aged systems, especially seed tree or shelterwood systems with retention of some overstory into the next rotation, also were recommended because they ensured some high structure, had greater visual appeal, and accommodated the needs of more cavity-dependent avian species.

Table 1—*Recommended silvicultural systems for ponderosa pine forests in the northern Rocky Mountains, Pacific Northwest, eastside California, and westside California.*

Region	Recommended silvicultural system	Source
Northern Rocky Mountains	Clearcutting with planting on dry sites, with limited use of seed tree or shelterwood methods; clearcutting with planting on moist sites, with limited use of group selection or seed tree methods	Foiles and Curtis 1973
Pacific Northwest	Clearcutting, limited use of shelterwood method; selection systems used for special areas	Barrett 1977
Northern Rocky Mountains	Clearcutting or shelterwood methods; selection systems used for special areas	Adams 1980
Pacific Northwest	Overstory removal followed by shelterwood or clearcutting method	Scott 1980
Eastside California	Clearcutting method with planting	Helms 1980
Westside California	Clearcutting or seed tree method, and group selection	Helms 1980
Northern Rocky Mountains	Clearcutting and planting on sites with infections of dwarf mistletoe; single tree or group selection on dry sites where natural regeneration is difficult to obtain; group selection or shelterwood on more moist sites	Ryker and Losensky 1983
Pacific Northwest	Both even-aged and uneven-aged systems; uneven-aged systems sometimes preferred for visuals, in recreations, and for wildlife habitat	Barrett and others 1983
Westside California	Clearcutting, seed tree, or shelterwood methods; uneven-aged systems are impractical	Oliver and others 1983
Westside California	Seed tree, shelterwood, and group selection methods	Minore and Kingsley 1983
Northern Rocky Mountains	Group selection, seed tree, or shelterwood systems with increasing emphasis on green-tree retention	Adams 1994
Pacific Northwest	Multiple-entry management using either long-rotation even-aged systems or uneven-aged system; some type of group selection likely to be most successful	Tesch 1994
Eastside California	Group selection with regeneration best achieved by planting	Helms 1994
Westside California	Clearcutting or seed-tree well-suited where wood production is the primary objective; group selection preferable to single-tree selection because of overstory competition constraints	Helms 1994

Silvicultural Systems Currently In Use

Currently, the full array of silvicultural systems, ranging from clearcutting to single and group selection, are found applied to ponderosa pine management on public lands from the northern Rocky Mountains to California. All the variations applied in specific situations probably are adequately defined and described as deviations within a more formally defined general system.

Perhaps the greatest change in management of ponderosa pine is the prominence of intermediate cuttings which have no expectation or objective of natural regeneration, especially improvement cuttings designed to remove less desirable trees in order to meet stand composition or vertical structure objectives. The emphasis of improvement cutting is on improving the stand structure rather than removal of any particular size stem. Many density-reduction treatments on dry sites in the Blue Mountains are improvement cuttings rather than strict thinning treatments because they have a strong species composition and tree quality component to them.

At least six recent social, political, and legal changes may appreciably influence the application of silvicultural prescriptions designed to implement silvicultural systems for managing ponderosa pine ecosystems:

- The USDA Forest Service adopted in 1992 a new policy limiting the use of clearcutting on national forests. The clearcutting regeneration method currently may be used only under specific circumstances such as to maintain habitat for threatened, endangered, or sensitive species; to enhance wildlife habitat; to rehabilitate lands adversely impacted by natural disturbances, or to rehabilitate lands due to past management practices.
- The Regional Forester amended the forest plans of National Forests in Oregon and Washington east of the crest of the Cascade Range in Oregon and Washington to include three “Eastside Screens”, one of which established a policy restricting the harvesting of trees greater than 53 centimeters (21 inches) in diameter at breast height for areas where stand structural characteristics of late and old successional stages were found to be deficient. This policy currently protects the larger and presumably older ponderosa pine, but it also limits management activities designed to reduce, for example, the spread of western dwarf mistletoe (*Arceuthobium campylopodum* Engelm.), western pine beetle (*Dendroctonus brevicomis* LeConte), or mountain pine beetle (*D. ponderosae* Hopk.).
- Interim Riparian Habitat Conservation Areas on federal lands, established under PACFISH (USDA Forest Service and USDI Bureau of Land Management 1994) and Inland Native Fish (USDA Forest Service 1995), established a buffer on either side of the active stream channel of fish-bearing streams, within which timber harvesting, including fuelwood cutting is prohibited except where silvicultural practices are needed to attain desired vegetation characteristics to meet riparian management objectives.
- Across the four regions, forest density management— silvicultural activities designed to improve stand conditions by concentrating tree growth on fewer stems, reduce fire risk and improve forest health, encourage undergrowth vegetation and wildlife forage, and promote patch- and landscape-scale diversity— has shifted to include a primary focus on reducing the number of small-diameter trees. In the Pacific Northwest Region of the Forest Service,

the forest density management program of work has not kept pace with the programmed need, and the backlog continues to increase at about 200 square kilometers (50,000 acres) each year (Powell and others 2001). The policy restricting the size of trees that are harvested also may already limit the amount of thinning in some ponderosa pine stands, especially stands that have grown near or beyond the 53-centimeter diameter limit. Growth rates in these stands may decline without thinning and the stands will become more susceptible to bark beetle attack.

- A new emphasis on fuel reduction to lessen the risk and severity of wildfire, especially in low elevation forests that have developed under natural disturbance regimes dominated by high frequency, low-severity fires, is focusing much of the silvicultural activity within the wildland/urban interface. The wildland/urban interface in eastern Washington near Spokane, in central Oregon near Bend, and in eastern California in the Tahoe Basin are examples of rapidly expanding wildland/urban interface where fuel reductions treatments are being applied to protect people and homes. Fuel reduction activities commonly used in these areas, such as underburning or mowing, usually result in short term decreases in available browse that may be important for wild ungulates during winter months.
- Implementation of National Fire Plan objectives for reducing fuels is limited because of challenges for the efficient disposal and/or utilization of significant quantities of small trees, especially the large numbers of low-volume stems less than 10 centimeters (4 inches) in diameter. A recent analysis of potential yields from fuel reduction projects in western states indicated the potential size of the manufacturing infrastructure needed to process material from fuel reduction treatments is large (USDA Forest Service 2003). For example, to process the merchantable volume from only the western states where fire regimes have been significantly altered and there is a high risk of losing key ecosystem components in a wildfire (Class 3), fuel reduction treatments will require the capacity of about 75 average-sized conventional sawmills for 30 years. Loss of processing mills, however, has eliminated much of the infrastructure necessary to conduct these fuel reduction treatments across much of the four regions. As a result, many fuel reduction projects now require financial subsidies, resulting in smaller areas being treated. Adding to the problem is the concern that many even-aged stands of ponderosa pine that were precommercially thinned are approaching the target diameter (about 20 centimeters (8 inches)), previously projected for the first commercial thinning. The market and demand for these small trees do not currently exist.

While these changes tend to directly affect management of ponderosa pine ecosystems, they also have an effect on the management of other forest types, especially those forest types at mid elevations that may contain small amounts of ponderosa pine.

Issues Affecting Future Application of Silvicultural Prescriptions

Given the broad array of silvicultural systems that can be use to manage ponderosa pine, it is unlikely that silvicultural activities occurring in the near term (the next 10 years) will not readily fit into the existing framework of planned treatments. What is likely to occur, however, is that increasingly complex resource values associated with ponderosa pine ecosystems will require silviculturists to consider a wider array of treatments within the context of ponderosa pine silviculture. One common context for management is emulation of natural disturbance regimes. Silviculturists likely will be asked to design treatments that are closer analogues or surrogates for a specific disturbance event. Manipulations of stand structure with the objective of reducing fuels will become an increasingly common management action on landscapes supporting ponderosa pine. Fuel reduction treatments involving either mechanical manipulation of fuels, the use of fire, or a combination of the two can result in changes in the horizontal and vertical distribution of trees, their composition, and the number, size, and distribution of gaps or canopy openings. It is likely that federal, state, and private resources devoted to fire suppression will continue to battle wildfires, many of which will occur in ponderosa pine ecosystems. Density management activities will continue to the degree that funding allows projects to be implemented. If these activities continue to be generally restricted to small-diameter trees only, it is likely that little effort will be devoted to creating complex stand structures through spatial and structural diversity. At the same time, fuels will continue to accumulate across landscapes until treated or burned by wildfire. There is a possibility that our collective focus on reducing fire risk will overshadow the need to consider other disturbance factors in the context of long-term forest health. For example, failing to address the increase in dwarf mistletoe in overstory trees now will probably result in greater numbers of infected understory trees in the future. Similarly, failing to consider fully the risk of bark beetle outbreaks may limit our management options and lead to greater reliance on clearcutting in severely infested areas to restore healthy stands.

What are the critical gaps in our understanding of ponderosa pine silviculture that may affect our management in the near-term and beyond? We must first recognize that essentially all of our knowledge base for managing ponderosa pine ecosystems is derived from studies conducted in even-aged stands. While existing work compares the various methods of regeneration after harvests, there are few if any empirical comparisons of even- and uneven-aged systems for the management of ponderosa pine across any full suite of treatments (Helms and Lotan 1988).

Current amounts of old-growth ponderosa pine forest in the northern Rocky Mountains, the Pacific Northwest, and both eastside and westside California are estimated at 3 to 15 percent of pre-Euro-American settlement levels (Bolsinger and Waddell 1993; Beardsley and others 1999). Increasingly, managers are attempting to restore the frequency and intensity of disturbances within existing old-growth stands and thus the resulting periods of stability through various fuel reduction treatments designed to ultimately prevent stand-replacement fires (Conard and others 2001; Oliver 2001). Land managers may lack the knowledge of how disturbance agents, both natural and human-caused, interact with each other and how they interact across multiple scales to cause changes that may affect ecosystem integrity. Methods to protect remnant old-growth ponderosa pine stands and individual trees during implementation of prescribed fire as well as protocols to ensure variability in thinning density to mimic the spatial distribution of stems in old-growth stands (Youngblood and others 2004) are largely unknown or unavailable. At the same time, little is known of methods for developing future old-growth structure in young uneven-aged or even-aged stands, and forecasting the productivity of these multi-aged stands. Tools such as the Ponderosa Pine-Multi-aged Stocking Assessment Model (O'Hara and others 2003) are promising steps toward this goal.

Finally, it is clear that much of the recent changes in the management of ponderosa pine ecosystems on public lands are the result of changes in the way society— either the individual living in the wildland/urban interface, the local community, the special interest groups that represent user groups, or national and international organizations— values these forests. Perhaps our most challenging unknown is how the various components of society will value these forests in the future. If processes were readily available for predicting the range of commodity and amenity values, and thus the structure, that society will desire in ponderosa pine forests of the future, management could begin to develop that structure today. Regardless of these knowledge gaps, the role of the silviculturist in guiding the development of ponderosa pine stands will remain exciting.

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Multiaged Silviculture of Ponderosa Pine[1]

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Abstract

Ponderosa pine (*Pinus ponderosa* P. & C. Lawson) is highly suitable for management using multiaged systems. This suitability is primarily the result of a frequent, low severity disturbance regime, but also because it naturally occurs at low densities and has a long history of management to promote multiple age classes. Several different stocking control tools are available for ponderosa pine including the q-factor, allocation of stand density index, and allocation of growing space represented by leaf area. These methods are applicable to pure or mixed stands. The productivity of multiaged ponderosa pine is shown to be comparable to even-aged stands. Trends in multiaged management have included shorter cutting cycles and lighter harvest treatments. Multiaged silviculture in ponderosa pine is highly suitable for achieving a variety of objectives including timber production, aesthetics, and restoring presettlement stand structures.

Introduction

Ponderosa pine (*Pinus ponderosa* P. & C. Lawson) has a long history of management to retain multiple age classes within a single stand. This is primarily the result of the disturbance regimes that affect regeneration, mortality, stand density, and other ecosystem processes. The result – prior to European settlement – was an abundance of unmanaged stands with multiple age classes of trees. Much of the early management of this forest type perpetuated this structure and many of our multiaged stands today are more the result of previous management than prehistoric disturbance regimes. As a result there is an abundance of stand structures that are multiaged and an increasing desire to maintain this complex structure with future management. This paper will provide an overview of the silviculture of stands with two or more age classes. These stands will be referred to as multiaged in contrast to the term uneven-aged that has traditionally denoted stands with three or more age classes (Helms 1998).

Fire is the primary disturbance agent, but insects, wind, and pathogens are also important. Although fire regimes are highly variable across the entire range of ponderosa pine, a low severity, high frequency regime is common in most areas with pure pine. As species composition becomes more complex at higher elevations or on moister sites, the fire regime becomes one of higher severity and lower frequency. In pure stands, the disturbance regime often results in multiaged stands. These areas represent an ideal situation for multiaged silviculture. In the mixed stands, managing for multiaged ponderosa pine is more difficult because ponderosa pine is usually less

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shade tolerant than its associates. In these situations, the conditions for regeneration and early growth favor other species and ponderosa pine may decline. In this paper, I will generally discuss mixtures and pure ponderosa pine stands separately.

Disturbance Ecology and Regeneration Patterns

Ponderosa pine may have the highest fire frequency of any tree species in western North America. Frequencies vary throughout the range: studies have documented fire return intervals as low as one year in pure stands and 40 or more years in mixtures (Arno 1980, Agee 1993, Weaver 1959). These fires had mixed effects: small trees were very susceptible whereas larger trees were largely unaffected. These events often gave rise to new age classes or cohorts of trees and resulted in highly variable multiaged stands. Patterns of regeneration were variable across the range of ponderosa pine. Some researchers have described groups of distinct age classes (Arno and others 1995, Cooper 1960, West 1969) while others have found heterogeneous patterns of age at the group level (White 1985). The former might resemble stands regenerated with a group selection system, the latter with single tree selection. Both patterns likely occurred at small scales. Generalizations are difficult at larger scales because the great heterogeneity over the entire range of ponderosa pine in soils, topography, elevation, associated species, and genetic variation in ponderosa pine. Another feature of older presettlement ponderosa pine forests was their open, park-like structure as was noted in many early descriptions of these forests. Later descriptions estimated crown closure at less than 30 percent (Covington and Moore 1994) and overstory tree densities of 40 – 60 stems ha^{-1} (Arno and others 1995, Covington and Moore 1994, Youngblood and others 2004).

Historical Management of Multiaged Stands

Much of the early history of timber harvest in ponderosa pine forests consisted of partial harvests that perpetuated multiaged stand structures. Initially these harvests were to support mines, railroads, farms, and towns. Later, as commercial harvests began in the late 1800s, they focused on private lands. These early cuttings were primarily heavy partial cuts that removed 75 to 90 percent of the volume but generally did not have a silvicultural objective (Mowat 1961). Following formation of the national forests, cutting of ponderosa pine on these lands followed similar patterns. One of the first published references to multiaged silviculture in ponderosa pine on national forests was by Clapp (1912). He described systems of very heavy removals of approximately two-thirds of the volume and cutting cycles from 40 to 60 years. Dunning (1928) developed tree classifications to aid in selecting trees that were overmature or low in vigor. Keen (1943) developed a more detailed tree classification system specific to ponderosa pine that became a key component of many multiaged systems for decades. Meyer's (1934) yield tables for selectively cut stands in the Pacific Northwest were one of the first references to quantitative criteria for multiaged management of ponderosa pine. Meyer proposed periodic cuttings of up to about 85 percent of board foot volume at intervals of greater than 40 years. Munger et al. (1936) developed a "maturity selection system" in the interior Pacific Northwest that emphasized the financial maturity of individual trees. This differed from diameter-limit cutting or zero-margin selection cutting (Smith et al. 1997) by also considering the tree's biological maturity in assessing whether to leave or cut. In the southwest, Pearson (1942) developed a procedure called "improvement selection" that attempted to upgrade the amount and quality of the growing stock.

Munger and Pearson debated the relative merits of "maturity selection" and "improvement selection" in the Journal of Forestry (see letters and responses following Pearson 1942). O'Hara (2002) concluded that the differences between the approaches were likely the result of differences in the current state of ponderosa pine forests of the interior Northwest and the southwest US: the forests of the southwest were probably more degraded from heavy cutting, grazing, an abundance of stocking from the 1919 regeneration year, and had a greater need for improvement than the forests of the interior Northwest.

Another trend in these early treatments that has carried forward to more recent multiaged silviculture is the decline in severity of harvest over time. Mowat (1961) noted this trend and described current practices around 1960 as removing 25-65 percent of volume. There are a number of factors that have contributed to this trend including greater stumpage values and the availability of the crawler tractor that made less severe harvest economically and operationally feasible. There was also a developing recognition that lighter cuttings produced greater yields (Brandstrom 1937, Roe 1947) because the stand response to partial harvests was greatest in the first decade or two after cutting (Mowat 1961, Roe 1952). Later descriptions of multiaged silviculture in ponderosa pine described cutting cycles of 20 to 30 years (Alexander and Edminster 1977, Shepperd and Battaglia 2002).

Contemporary Stocking Procedures

Stocking control is central to multiaged systems because through stocking the silviculturist affects stand structure, the potential for regeneration, and the sustainability of the stand. There are a variety of stocking control procedures for multiaged stands managed with single tree selection systems. O'Hara and Gersonde (2004) discussed the development of these systems over time. The most common stocking control procedure for ponderosa pine has been the q-factor approach that uses a reverse-J diameter frequency distribution. The traditional interpretation of the q-factor was as a constant diminution quotient where the number of trees in a diameter class is a constant ratio of the tree number in the next larger class (*fig. 1*). For example, a q-factor of 1.2 applied to a 25-30 cm diameter class with 50 trees ha^{-1} would result in 60 trees ha^{-1} in the 20-25 trees ha^{-1}cm class and 72 trees ha^{-1} in the 15-20 cm class. This diameter distribution may represent the post-harvest stand target. Using this method, a stand is marked to conform to the target diameter distribution at the end of each cutting cycle. During each cutting cycle the stand will experience growth in most diameter classes and the diameter distribution moves away from the target. Alexander (1986) and Fiedler and others (1988) provide more details on this approach. This approach has also been described as the BDq approach where B represents the total basal area, D the maximum diameter class, and q the q-factor of the target structure (see Guldin 1991).

Stand regulation with the q-factor approach is achieved through harvest or thinning treatments that reset the stand to the target structure at the end of each cutting cycle. Stands managed to meet a particular q-factor diameter distribution have been described as "balanced" (Meyer 1943, 1952, Nyland 2002, Smith et al. 1997) because they were assumed to have constant volume production over time and equal space occupancy by each size and age class. The balanced stand therefore resembles a forest under area control regulation but at a smaller scale. O'Hara (1996) questioned these assumptions and concluded that management to maintain these "balanced" stands resulted in lower yields for ponderosa pine and didn't guarantee sustainability. In practice, segmented q-factors can be used so some parts of the diameter distribution have a different q-factor than other parts. This helps avoid the common drawback of this approach where large numbers of small trees occupy a greater share of growing space than necessary for most management objectives.

Stand density index (SDI) can also be allocated among diameter classes or groups of classes as a means of allocating growing space in multiaged stands. Long and Daniel (1990) demonstrated this approach with examples for ponderosa pine (*fig. 1*). SDI can be calculated for groups of diameter classes and then added to obtain a stand-level total. Their assumption was that even-aged density management zones for even-aged stands could also be applied to multiaged stands. Long and Daniel (1990) demonstrated how the approach could be applied to design stands that deviated from the traditional reverse-J diameter distribution with a constant diminution coefficient. This approach has also been applied to ponderosa pine stands with presettlement structures (Cochran 1992) and two-aged stands (Long 1996).

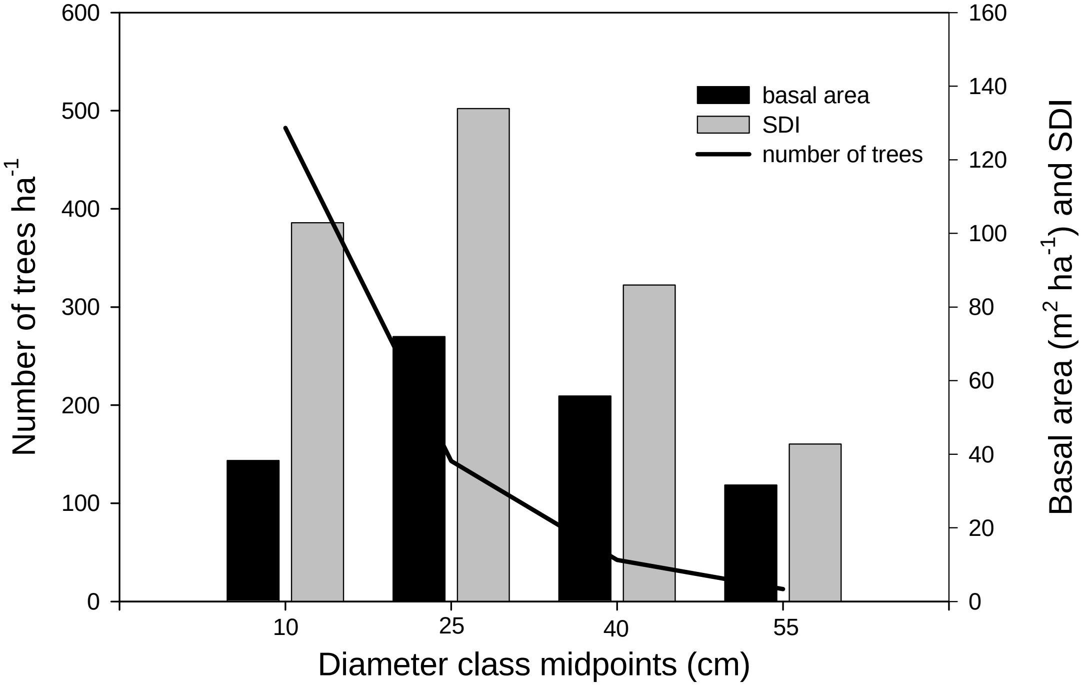

Figure 1— A diameter distribution, stand density index, and basal area for a ponderosa pine stand with a q-factor of 1.5 using metric units. Note the unequal distribution of SDI and basal area on a diameter distribution that was assumed to be "balanced" (Modified from Long and Daniel 1990 and O'Hara and Gersonde 2004).

All stocking control procedures are essentially tools for allocating growing space. This was explicit in the leaf area or growing space allocation tool developed for multiaged ponderosa pine (O'Hara 1996, 1998; O'Hara and others 2003; O'Hara and Nagel 2004). The growing space allocation method is distinguished from other methods because: 1) trees are divided into two to four stand components such as canopy strata or age classes rather than diameter size classes; and 2) growing space occupancy is represented by leaf area index (LAI - ratio of leaf surface area per unit of ground area covered). A user of the system divides growing space among components so as to meet management objectives. The target structure might therefore have the most growing space allocated to larger trees or perhaps to smaller trees. This flexibility is an asset of this method and allows the design of stand structures such as two-aged stands and presettlement structures with large or old trees.

A spreadsheet-based model is used for growing space allocation in this procedure. This model, called PP-MASAM, has been calibrated for pure ponderosa pine stands from three regions: central Oregon, western Montana, and the Black Hills in South Dakota and Wyoming (O’Hara and others 2003; O’Hara and Nagel 2004) and is available online at cnr.berkeley.edu/~ohara/downloads/. *Table 1* shows a three-age ponderosa pine stand using the Montana version of PP-MASAM. A four-age stand is described in *Table 2* using the PP-MASAM from Montana. The model projects volume increment, and calculates estimates of basal area, tree vigor, and SDI. It is flexible for designing stands that are two-aged, stands with high or low stand densities, or designing stand structures where harvest of presettlement trees above a certain diameter is avoided.

Table 1 — *Three-aged ponderosa pine stand designed with the PP-MASAM model from Montana (from O'Hara et al. 2003). Bold text are values provided by the user: the others are model output.*

	USER-SPECIFIED VARIABLES				
TOTAL Leaf Area Index (LAI)	6				
	Cohort 1	Cohort 2	Cohort 3	Cohort 4	TOTAL
Number of Trees/Cohort/Hectare	60	100	140	0	300
Percent of LAI/Cohort	50	35	15	0	100
	DIAGNOSTIC INFORMATION				
	Cohort 1	Cohort 2	Cohort 3	Cohort 4	TOTAL
Leaf Area Index/Cohort ECC	3.0	2.1	0.9	0.0	6.0
Leaf Area Index/Cohort BCC	1.3	0.6	0.0		1.9
Leaf Area/Tree (m^2) ECC	500.0	210.0	64.3	0.0	
BA/Cohort (m^2/ha) ECC	15.0	9.7	3.8	0.0	28.5
BA/Cohort (m^2/ac) BCC	5.8	2.7	0.0		8.5
Avg. Vol. Increment/Tree (m^3/yr) ECC	0.04	0.01	0.01	0.00	
Avg. Vol. Increment/CC (m^3/ha/yr)	1.7	1.0	0.4	0.0	3.1
Quadratic Mean DBH/Cohort (cm) ECC	50.1	31.1	16.5	0.0	
Tree Vigor (cm^3/m^2/yr)	76.988	79.069	91.604	0.000	
Stand Density Index ECC	182.2	141.9	72.3	0.0	396.3
Stand Density Index BCC	85.1	51.6	0.0		136.8

Table 2 — *A four-aged ponderosa pine stand designed with the PP-MASAM model from Montana. Bold text are values provided by the user: the others are model output. The PP-MASAM model is also available in English units (from O'Hara et al. 2003).*

	USER-SPECIFIED VARIABLES				
TOTAL Leaf Area Index (LAI)	**6**				
	Cohort 1	Cohort 2	Cohort 3	Cohort 4	TOTAL
Number of Trees/Cohort/Hectare	**45**	**60**	**74**	**89**	268
Percent of LAI/Cohort	**40**	**30**	**20**	**10**	100
	DIAGNOSTIC INFORMATION				
	Cohort 1	Cohort 2	Cohort 3	Cohort 4	TOTAL
Leaf Area Index/Cohort ECC	2.4	1.8	1.2	0.6	6.0
Leaf Area Index/Cohort BCC	1.4	1.0	0.5		2.8
Leaf Area/Tree (m^2) ECC	533.3	300.0	162.2	67.4	
BA/Cohort (m^2/ha) ECC	12.1	8.3	5.1	2.5	28.1
BA/Cohort (m^2/ac) BCC	6.3	4.2	2.1		12.5
Avg. Vol. Increment/Tree (m^3/yr) ECC	0.05	0.02	0.02	0.00	
Avg. Vol. Increment/CC (m^3/ha/yr)	1.5	1.1	0.7	0.1	3.4
Quadratic Mean DBH/Cohort (cm) ECC	51.8	37.3	26.3	16.8	
Tree Vigor (cm^3/m^2/yr)	77.885	80.591	71.188	46.368	
Stand Density Index ECC	144.4	113.8	80.4	47.2	385.8
Stand Density Index BCC	85.4	65.2	39.3		189.8

Group Selection

Another highly appropriate form of multiaged silviculture for ponderosa pine is group selection. These groups provide openings with sufficient light to allow shade intolerant species to germinate and be competitive in mixed-species stands. Another advantage of group selection is the operational efficiencies that arise from harvesting in larger gaps than single tree selection systems.

For stand regulation, group selection operates on an "area control" basis where openings are moved throughout a stand and each opening cycles back for a second treatment after a period of time analogous to an even-aged rotation. In one of these "rotations", the entire stand would be treated with equal areas harvested in each cutting cycle. Although there is very little experience with more than one "rotation" with group selection systems, a number of areas in the ponderosa pine region are through several cutting cycles. A key planning strategy when first establishing a group selection system is to organize openings so that all are accessible through all cutting cycles. There are also advantages to how openings are oriented with respect to shading each other. For example, there may be significant advantages to having a younger and shorter group of trees on the south side of a new opening to maximize light availability.

Although group selection openings provide open growing conditions for developing trees, this open condition occurs primarily in the center or north side of openings in the ponderosa pine region and declines toward the group edges. In mixed-species forests – that often have greater crown closure and higher LAI – group sizes must therefore be sufficiently large to provide conditions where ponderosa pine has an advantage. York and others (2004) documented the edge effect for six conifer species in the Sierra Nevada where the greatest growth was from seedlings in the center and north of center within group openings. For ponderosa pine, mean height of seedlings after five years was greatest in the largest opening sampled (1 ha). They also measured growth of trees outside but bordering the group opening and found increased growth after group establishment suggesting a positive productivity effect that might compensate for the growth losses of edge seedlings in the opening.

Mixed-Species Applications

Mixed stands that include ponderosa pine also have the potential for management with multiaged systems. However, with the exception of western larch (*Larix occidentalis* Nutt.) in the inland Northwest and the Northern Rocky Mountains, ponderosa pine is less shade tolerant than all of its major competitors in mixed-species stands. This provides a competitive disadvantage for ponderosa pine in multiaged systems in these mixed species types. Without management intervention, the regeneration success of ponderosa pine will decline leading to greater dominance by more shade tolerant species. Whereas these mixed-species stands are desirable for a variety of management objectives, the decline in numbers of shade intolerant species like ponderosa pine often leads to a number of insect and pathogen problems and may leave the stand more susceptible to fire.

Several studies have examined multiaged stand management in the Sierra Nevada. Lilieholm et al. (1990) documented trends in diameter distributions and species composition after several decades of selection harvests in stands that were even-aged but with irregular stocking. They found all five major conifer species – including ponderosa pine – were able to regenerate although the more shade tolerant species comprised the bulk of the regeneration. They also noted that the diameter distributions after 20-28 years of harvest treatments were moving towards the target reverse-J distributions. Guldin (1991) applied the BDq method to the Sierra Nevada mixed-conifer forests. He concluded that a form of group selection was needed to insure regeneration of shade intolerant species and advocated the BDq procedure for these stands. Gersonde and others (2004) used a light model to identify locations where shade intolerant species were most likely to be successful in mixed-conifer stands in the Sierra Nevada.

In other parts of the ponderosa pine range, similar problems exist. For example, interior Douglas-fir (*Pseudotsuga menziesii* var. *glauca* (Mirb.) Franco) and several other species compete with ponderosa pine on many sites in the inland Northwest and northern Rocky Mountains. In these forests, mechanical control of Douglas-fir or use of understory burning are necessary to give ponderosa pine an advantage and maintain its presence.

Other Issues with Multiaged Stands

The Cutting Cycle

In a fully regulated multiaged system, the length of the cutting cycle is a function of the severity of the harvests (O'Hara and Valappil 1999). Heavy harvest treatments would necessitate longer cutting cycles to rebuild pretreatment stocking levels. Lighter harvest treatments would have the opposite effect: shorter cutting cycles. Many of the earliest multiaged systems in ponderosa pine used very heavy removals (Meyer 1934) whereas more recent systems used light cuttings and short cutting cycles (Becker 1995, Shepperd and Battaglia 2002). O'Hara and Valappil (1999) described a tradeoff between the probability of obtaining regeneration and volume increment with shorter cutting cycles/lighter harvests resulting in greater volume increment and a lower probability of obtaining natural regeneration because of heavier competition. The opposite would be true for longer cutting cycles/heavier harvests. The trend to shorter cutting cycles and with lighter harvests in ponderosa pine may favor greater stand growth but less regeneration. In mixed stands, longer cutting cycles would be more likely to produce ponderosa pine regeneration and give this species a competitive advantage.

Theoretically the target multiaged stand is best met at only one point in time during a cutting cycle. Otherwise, the stand is either growing towards or away from the target. Longer cutting cycles would therefore deviate more from the target than shorter cutting cycles. Another consideration is the q-factor and SDI approaches generally design target structures for the beginning of the cutting cycle. The PP-MASAM designs end-of-cutting cycle structures but also provides data on stand characteristics at the beginning of the cutting cycle. When designing a cutting cycle it is more important to recognize the tradeoffs between cutting cycle length and harvest levels and how to achieve the target stand structure for as long as possible.

Productivity

The relative productivity of multiaged stands to other stand structures, particularly even-aged stands, is a point of great interest for most foresters. There is no empirical evidence for ponderosa pine that indicates multiaged stands are more or less productive than other structures. O'Hara (1996) compared increment in even-aged and multiaged ponderosa pine stands in central Oregon and western Montana. Cubic volume increment was slightly higher for multiaged stands in both study areas but differences were not significant (Table 3). Since LAI was slightly higher in even-aged stands, these results suggest higher efficiencies of growing space occupancy in the multiaged stands. Similar results were obtained in the Black Hills although only one even-aged stand was sampled (Table 3 – O'Hara and Nagel 2004). Additional studies found greater water pressure deficits in even-aged stands during the growing season suggesting differences in water availability may contribute to lower productivity in even-aged stands (Nagel and O'Hara 2002). A review on multiaged productivity[3] concluded that any productivity differences that might exist between even-aged and multiaged stands were small and probably less significant than differences in operational effects on growth and volume recovery [(O'Hara and Nagel 2006)].

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[3] O'Hara, K.L. and L.M. Nagel. 2006. A functional comparison of productivity in even-aged and multiaged stands: A synthesis for *Pinus ponderosa*. Forest Science (in press).

Table 3 — *Mean (± SE) volume increment, leaf area index (LAI), and growing space efficiency (GSE) for even-aged and multiaged ponderosa from three study areas. Only the GSE means for central Oregon were significantly different of these results. Data are from O'Hara (1996) and O'Hara and Nagel (2004).*

Location	Age Structure	Vol. Increment ±SE $m^3\ ha^{-1}\ yr^{-1}$	LAI ±SE	Stand GSE ±SE $m^3\ m^{-2}$
Western Montana	Even-aged	5.0 ±0.3	7.2 ±0.47	0.72 ±0.04
Western Montana	Multiaged	5.4 ±0.3	6.8 ±0.31	0.80 ±0.03
Central Oregon	Even-aged	4.8 ±0.4	6.9 ±0.41	0.70 ±0.04
Central Oregon	Multiaged	4.7 ±0.3	6.1 ±0.29	0.81 ±0.03
South Dakota[1]	Even-aged	5.8	7.8	0.73
South Dakota	Multiaged	5.2 ±0.2	6.9 ±0.043	0.77 ±0.04

[1] One even-aged stand was sampled in South Dakota so a comparison of means was not possible for these data.

Slash and Fuel Treatments

The variation in tree sizes in multiaged stands has the potential to form fuel ladders that increase the potential for crown fire development. In pure ponderosa pine, stands can be maintained at low densities so fuels are not continuous. In mixtures, this problem is more significant and may preclude multiaged stands in some locations such as fuel breaks or near structures. Prescribed understory burning may also be more difficult in multiaged stands because some small trees would always be present. Slash disposal may also be problematic as broadcast burning is not possible and residual trees may limit equipment options for piling. These are additional limitations on operations beyond those related to the removal of trees from multiaged stands and may require additional expenses to provide comparable fire protection as in older even-aged stands.

Site Quality

A long-standing shortcoming of multiaged systems is the inability to estimate site quality using site index. Because a requirement of site trees is that they be free growing trees throughout their development, any tree developing in a multiaged system is unsuitable. Managers are therefore forced towards finding similar sites with free-growing even-aged trees, using soil-site relationships, or vegetation analyses such as plant associations. These latter methods often provide only wide ranges of site productivity values and are ultimately tied to site index relationships from even-aged stands. There have been attempts to determine site index from the free-growing period of development for a multiaged tree, but this is an area that could use more research.

Summary

Ponderosa pine has a rich history in both the research and practice of multiaged silviculture. A variety of different methods have been used and many have had some success. Much of this success is because ponderosa pine, particularly when growing in pure stands, is a highly suitable species for multiaged management. This is due to several factors including the low stand densities of many ponderosa pine stands that can permit regeneration and development of several age classes. Another important factor is the disturbance regime dominated by high frequency, low intensity fire in much of the ponderosa pine range. These disturbances often produce new age classes resulting in multiage stands that can be easily emulated by management. On moister sites, where ponderosa pine often grows with more shade tolerant species, multiaged silviculture can be more difficult, particularly on sites where fire exclusion has already altered the species composition of these stands. On these sites, multiaged silviculture can also be successful, but will required additional treatments to control undesirable species and to reduce fire hazards.

One of the truly great assets of the pure ponderosa pine ecosystem is the wide range of management practices that can be successful. This includes multiaged as well as even-aged silviculture. For multiaged systems, land managers can usually choose between regimes with short or long cutting cycles, high or low levels of residual stocking, and many to just a few age classes or cohorts. Flexibility to accommodate a wide variety of regimes is therefore an important component of any management tool. Multiaged silviculture can therefore be used to meet a wide variety of objectives in an ecosystem management context. These objectives can include providing stand structures for wildlife habitat, timber production, enhancing aesthetics, or restoration of presettlement stand structures.

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The West-Wide Ponderosa Pine Levels-of-Growing-Stock Study at Age 40[1]

William W. Oliver[2]

Abstract

In the 1960s a series of levels-of-growing-stock studies was established in young, even-aged stands throughout the range of ponderosa pine in the western United States. Using a common study plan, installations were begun in the Black Hills of South Dakota, eastern and central Oregon, the Coconino Plateau of Arizona and the west slope of the Sierra Nevada in California. Innovative features for the time were tests of a wide range of stand densities from open-grown to densities high enough to jeopardize stand health, and to periodic rethinning of the plots back to the stand density level originally assigned. Perhaps the major reason for the longevity of this west-wide study was the foresight of the original planners in testing a range of stand densities far beyond those practiced at the time. Establishment of this range provided a demonstration of the long-term stand development of even-aged ponderosa pine applicable to many current management objectives.

Introduction

Ponderosa pine (*Pinus ponderosa* P. & C. Lawson) is one of the most widely distributed conifers in North America. It occurs in 15 western states, extending from the western Great Plains to the Pacific Coast and from southern British Columbia, Canada, to Baja California, Mexico. Ponderosa pine is found at elevations ranging from sea level in the northern part of its range to 10,000 ft in the southwestern United States (Oliver and Ryker 1990). Throughout this vast area ponderosa pine is one of the most valued tree species. Recognized initially for the quality of its wood yields and as a major source of forage for cattle, ponderosa pine forests are now recognized as vital wildlife habitat, and they provide abundant recreational opportunities. As a result, ponderosa pine forests have a long history of intensive management.

In the mid 20th century, however, management of young growth ponderosa pine forests was in its infancy. Results from studies conducted in one area (Gaines and Kotok 1954; Mowat 1953; Myers 1958) seemed to have limited use in another area. Dissimilar experimental designs and measurements that were incomplete or based on specific products further restricted comparability. Also, these early studies did not test the low reserve densities that may be desired for today's multi-resource management.

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[1] An abbreviated version of this paper was presented at the symposium on Ponderosa Pine: Issues, Trends and Management, October 18-21, 2004, Klamath Falls, Oregon.

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Dissimilar experimental designs and measurements that were incomplete or based on specific products further restricted comparability. Also, these early studies did not test the low reserve densities that may be desired for today's multi-resource management.

Consequently, growth information was needed over a wide range of stand and site conditions and with a minimum of operational restrictions to provide useful guides for a variety of management objectives. To fill this need three western Research Stations of the U.S. Department of Agriculture, Forest Service, cooperated on a large scale levels-of-growing-stock study through the use of a common study plan (Myers 1967).

Many reports have been published on individual installations (*table 1*) of this west-wide study and two reports combining data from most of the installations were published more than 15 years ago (Oliver and Edminster 1986; Edminster 1988). This paper reports on the present status of the six installations after about 40 years. It describes the development of the stands, their value in demonstrating the influence of low reserve stocking, and the opportunities for satellite studies.

Original Study Plan

Five regions within the range of ponderosa pine in the western United States were arbitrarily selected for study and assigned to the participating Research Stations. These five "provinces" differ in many respects. Physiography ranges from the uplift of the Black Hills of South Dakota to the Coconino Plateau of Arizona, the Blue Mountains and Cascade Range of Oregon, and Sierra Nevada of California. Two varieties of ponderosa pine are recognized (*Pinus ponderosa* P. & C. Lawson var. *ponderosa* and *Pinus ponderosa* P. & C. Lawson var. *scopulorum*) (Conkle and Critchfield 1988). Some provinces are without summer precipitation; other provinces receive most of their annual precipitation during the growing season. Within each province, combinations of four tree-size classes from small saplings to large poles on areas of three site qualities of high, medium, and low were to be sampled, as available, in even-aged stands.

The initial plan specified that each study installation would consist of three replications of plots thinned to five or six different stand densities. Plots were to be at least 0.25 to 0.5 acre in size with 20-ft isolation strips for small saplings, and 0.5 to 1.0 acre with 30-ft isolation strips for the larger size classes. One plot in each replication was to be thinned to the density considered best for that site quality, based on past experience. Two or three plots were to be thinned to lower, and two or three plots to higher stand densities. The highest stand density would be chosen such that the production of merchantable material would be reduced below that of lower densities. And the lowest density level would be chosen such that cubic-foot volume production would be lower than that at higher densities.

Stand densities to be retained after thinning were specified as a series of growing stock levels (GSLs) (Myers 1967). These levels were defined by relationships between basal area and average stand diameter. Numerical designation of the level assigned to a plot was the basal area per acre that would remain after thinning when mean stand diameter was 10 inches or more. Stands with trees smaller than 10 inches when thinned would contain residual basal areas that were less than the designated GSL.

Table 1— *Installations in the west-wide levels-of-growing-stock study fore even-aged ponderosa pine.*

Geomorphic Province	Name & Year	Site Index[1]	Size class	Growing stock levels	Periods analyzed	References
Black Hills South Dakota	Black Hills Saplings 1964	55	large saplings	20,40,60, 80,100, 120	7	Boldt and Van Deusen 1974
Black Hills South Dakota	Black Hills Poles 1964	55	small poles	20,40,60, 80,100, 120	7	Boldt and Van Deusen 1974
Coconino Plateau Arizona	Taylor Woods 1962	62	small poles	30,60,80, 100,120, 150	7	Ronco and others 1982 Schubert 1971
Blue Mts. Oregon	Crawford Creek 1967	60	small poles	30,60,80, 100,120, 140	6	Cochran and Barrett 1995
East-side Cascade Range Oregon	Lookout Mt. 1966	92	large poles	30,60,80, 100,120, 150	7	Barrett 1983 Cochran and Barrett 1999
West-side Sierra Nevada California	Elliot Ranch 1969	140	Small poles	40,70,100, 130,160	6	Oliver 1979 Oliver 1997

[1] feet at 100 yrs (Meyer 1938).

Thinning was to be from below, primarily, with the smallest trees and rough dominants removed first. Each installation was scheduled to run for 20 years, with measurements at 5-year intervals. At the end of 10 years, plots were to be rethinned to the specified GSLs.

Before the growing season following initial thinning, all trees were to be tagged and measured for diameter at breast height (dbh) to the nearest 0.1 inch. Tree damage and diseases were to be noted. Various sampling schemes could be used to select trees for total height and height to the living crown, measured to the nearest 1 ft. Total stem volume was obtained either by the use of local volume tables or by an optical dendrometer. All measurements were to be repeated on the same trees after each 5-year period.

The Installations

The study as envisioned was ambitious, requiring prodigious amounts of land, labor, and money. Cooperators experienced difficulty both in finding sufficient areas of uniform site and stand conditions, and in allocating sufficient resources to sample but a few of the province/size/site combinations. Although many fewer than planned, the five installations established in naturally regenerated stands and the one installation established in a plantation (Elliot Ranch) nearly spanned the geographic range of ponderosa pine (*table 1*).

During the planned 20-year life of the study, the six installations were established and maintained in general conformance with the plan except as follows: (1) two rethinnings were performed at Elliot Ranch at five-year intervals rather than ten-year intervals because of rapid growth (*fig. 1*); (2) only one rethinning was performed in the lowest GSL plots at Lookout Mountain because a second rethinning would have resulted in too few trees for a sufficient sample; (3) only one rethinning has been performed in both of the Black Hills installations; (4) the controlling measure of stand density was changed to stand density index (SDI; Reineke 1933) at Lookout Mountain, Crawford Creek and Elliot Ranch Because basal area (GSL for mean stand diameters above 10 inches) as a measure of stand density is not independent of tree size and age. Remarkably, all installation have been continuously maintained and remeasured for twice as long originally envisioned, albeit not always on a rigid 5-year interval.

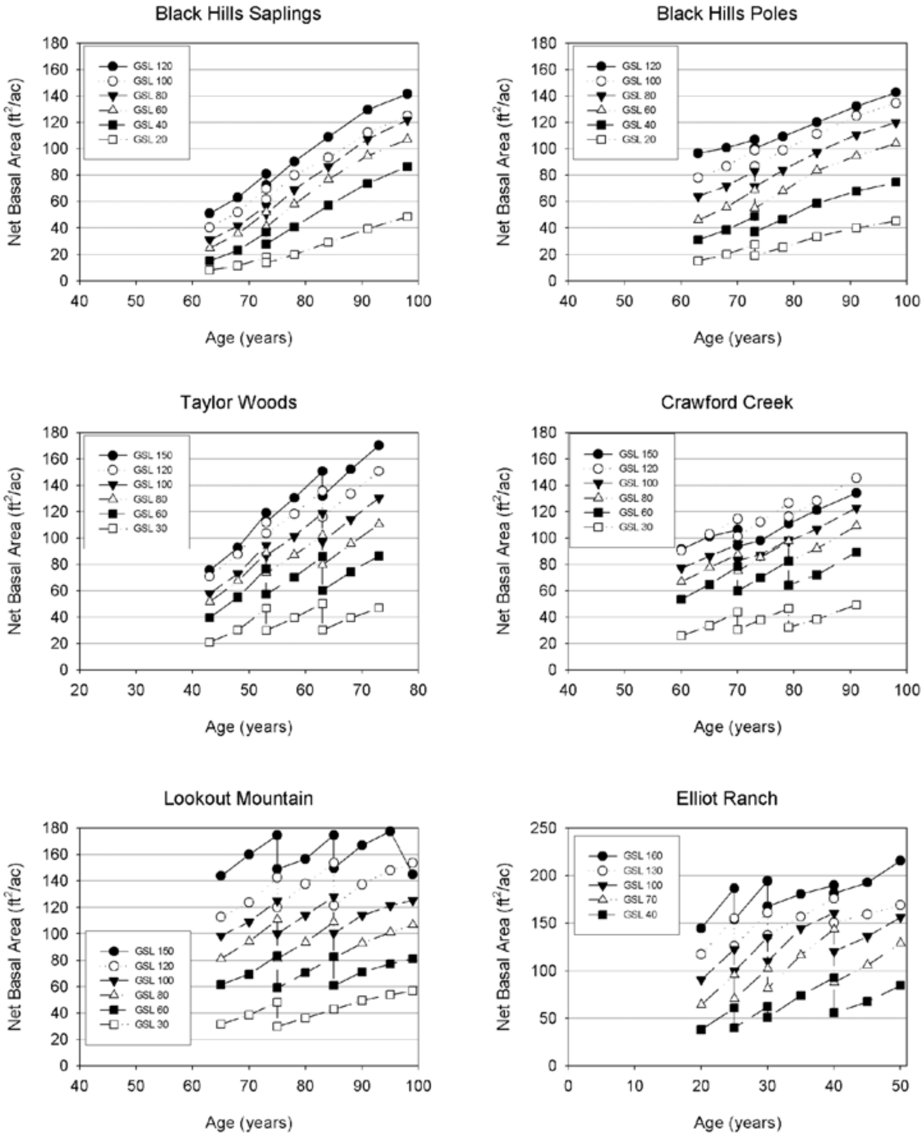

Figure 1—Net basal area development of the Growing Stock Level (GSL) treatments at six installations of the west-wide ponderosa pine levels-of-growing-stock study.

Results and Discussion

Treatment Effects on Stand Growth

An unthinned stand treatment was not included in the original plan. Neither is information available on the stand before thinning. Presuming that the stands before treatment were similar to those of normal density (Meyer 1938), the thinning treatments created a uniform array of stand densities ranging from those containing essentially open-grown trees to those retaining about half of the original basal area after light thinning.

Mortality (primarily in the higher reserve densities) tended to blur this uniform array. Mortality from mountain and western pine beetle was especially prevalent 10 to 20 years after initial thinning in the GSLs above 100 at Elliot Ranch. Mortality from mountain pine beetles was also high at Crawford Creek and breakage from a heavy wet snow was so severe at Taylor Woods that the two highest GSLs at these two installations differed little in actual density for the first 10 years. Mortality by all agents has declined over the years. Except for recent killing by mountain pine beetles at the highest GSL at Lookout Mountain, the treatments are now distinctly different in stand density.

Stand Structure Changes

Thinning, especially heavy thinning, has had a profound influence on the structure of the plots (*fig. 2a, 2b*). In general, maintaining plots at the highest reserve densities, GSLs 120 to 160 has over the period of observation transformed the diameter distribution from one resembling a reverse “J’ shape to one resembling a normal distribution. Heavy thinning, in contrast, immediately created a normal distribution that has been maintained for 40 or more years.

Opportunities for Satellite Studies

Studies continuously maintained for 40 years are rare in forestry. Research interests and priorities can change over such a long time and make such long-term studies obsolete. Nevertheless, silvicultural studies in which vegetation is manipulated in a statistically valid field design and with well-documented treatment responses afford many opportunities for studies never anticipated by the original researchers. Such is the case in this west-wide levels-of-growing-stock study. This study has already answered many of the questions for which it was first designed. The wide range of stand densities created under carefully controlled manipulations, however, provides opportunities for many satellite studies. One example has been a study demonstrating the efficacy of low reserve densities in maintaining stand health. Because most tree mortality, whether it is caused by biotic or abiotic factors, is episodic, evaluations of forest health are meaningful only if reserve stand densities are maintained over a long period of time. Another use of the overall data is in the construction of growth models for even-aged ponderosa pine stands. The database for these models is restricted to remeasured permanent plots, of which the data from the west-wide levels-of-growing-stock study is a critical part. Some examples of these studies follow.

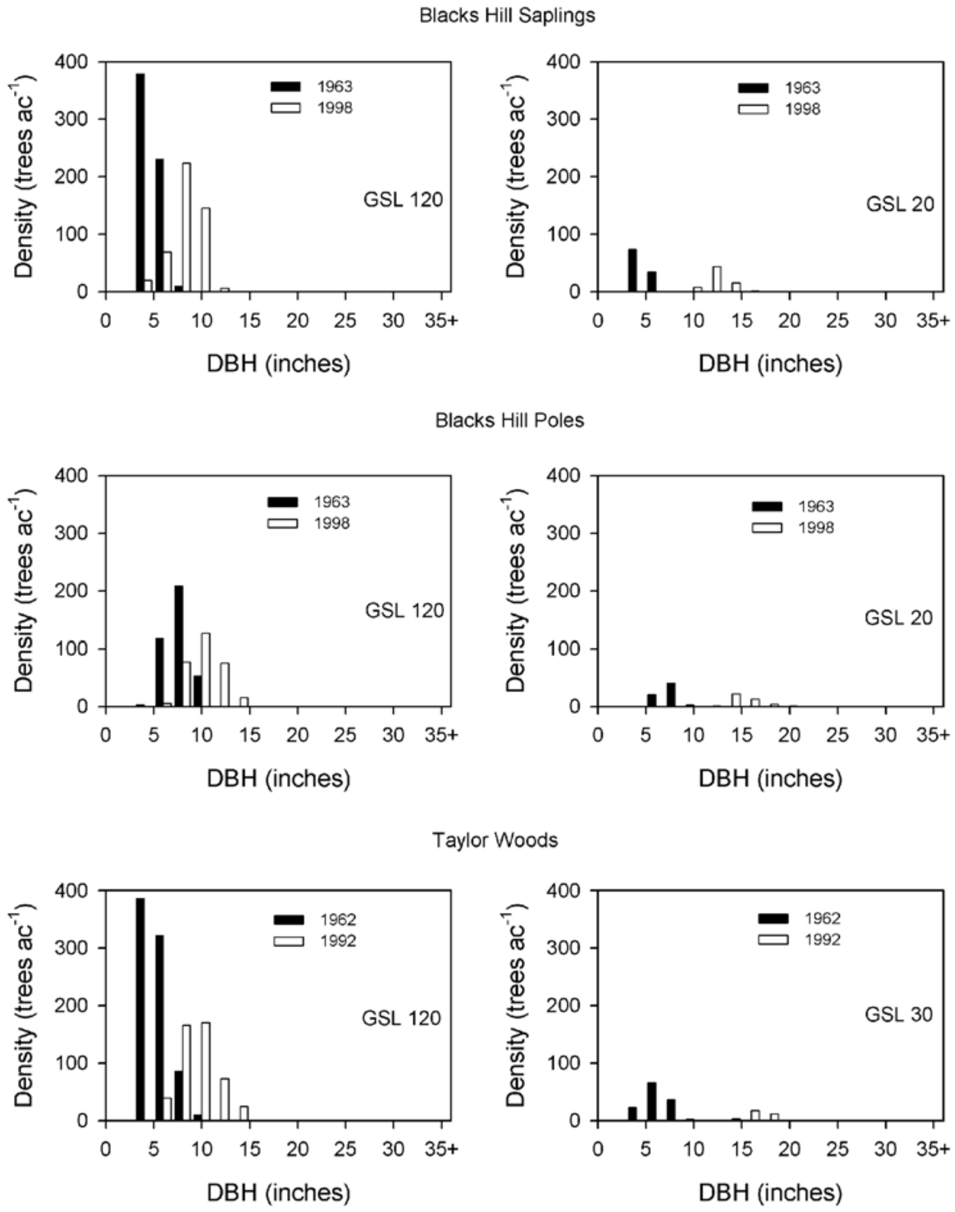

Figure 2a—Diameter distributions for the extreme Growing Stock Level (GSL) treatments after initial thinning and 30 or more years later for three installations (Black Hills Poles, Black Hills Saplings and Taylor Woods) in the west-side ponderosa pine levels-of-growing-stock study.

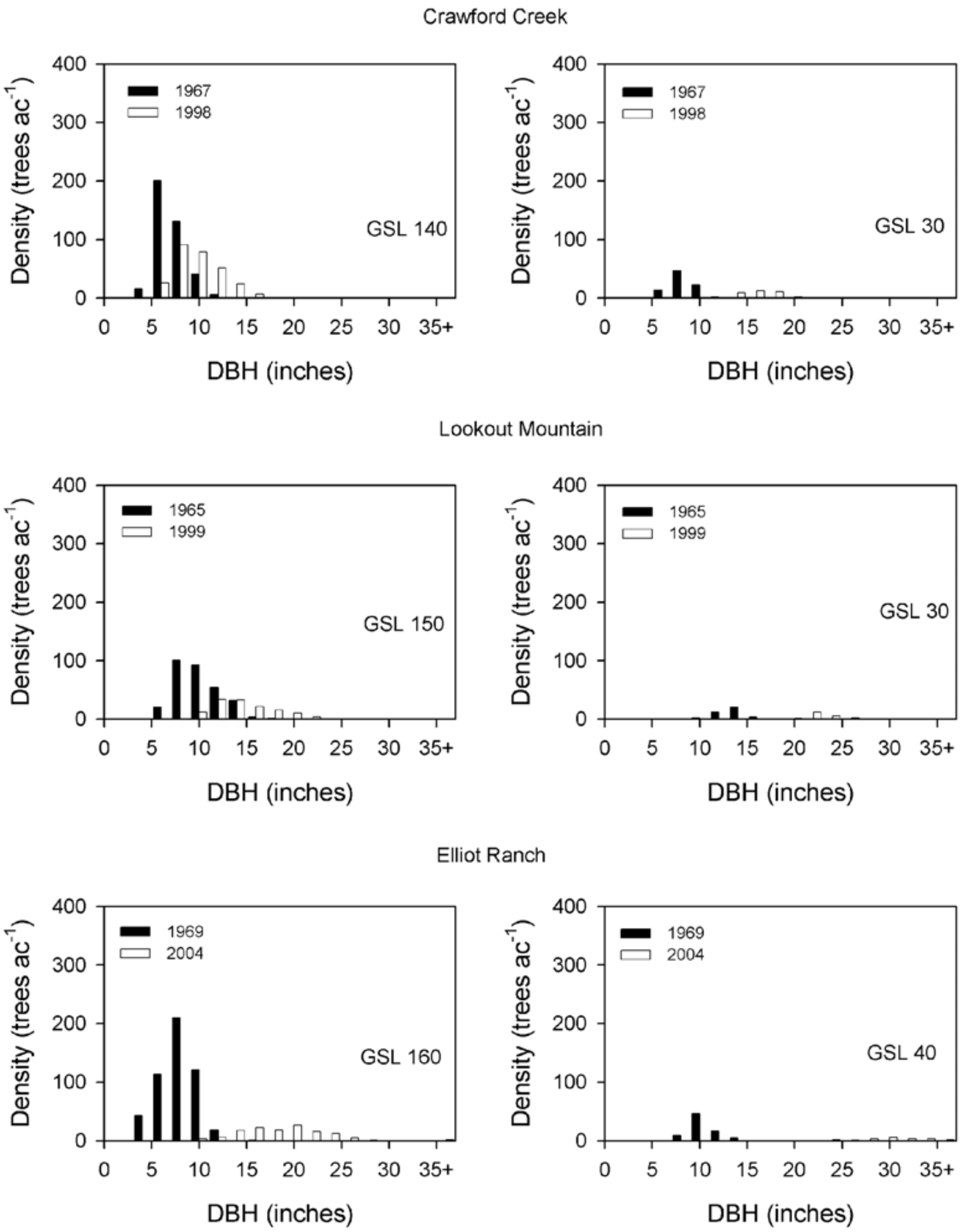

Figure 2b—Diameter distributions for the extreme Growing Stock Level (GSL) treatments after initial thinning and 30 or more years later for three installations (Crawford Creek, Lookout Mountain and Elliot Ranch) in the west-side ponderosa pine levels-of-growing-stock study.

Elliot Ranch

- Influence of overstory stand density on amount and availability of deerbrush (*Ceanothus integerrimus* Hook. & Arn.*)* browse for cattle and deer.
- Influence of overstory shade on growth of five species of conifer seedlings (Oliver and Dolph 1992).
- Influence of *Dendroctonus* bark beetles on maximum stand density for ponderosa pine (Oliver 1995).
- Demonstration of how to create complex species and stand structures in a mono-specific plantation. This has been of particular interest to visitors because of the abundant regeneration of mixed conifers and the interest in speeding development of more "natural" appearing forests. Unfortunately, leaving this advance regeneration unchecked would jeopardize the original study objectives.

Taylor Woods

- Forage production under different overstory stand densities
- Testing differences in canopy cover as measured by spherical densiometer and moosehorn.
- Contribution to database for simulating potential production for various combinations of stand density, site index, age and thinning schedule (Alexander and Edminster 1980).

Black Hills

- Forage production under different overstory stand densities (Severson and Boldt 1977)
- Influence of stand density on wood quality (Markstrom and others 1983)
- Costs and returns from pruning stands thinned to various stand densities (Smith and others 1988)
- Contribution to database for simulating potential production for various combinations of stand density, site index, age and thinning schedule (Myers 1966; Alexander and Edminster 1980).

Conclusions

Although the scientists involved in establishing the installations have retired (four scientists from the Rocky Mountain Research Station have been responsible for Taylor Woods throughout their careers) and the Pacific Northwest Research Station has officially abandoned Lookout Mountain and Crawford Creek, the commitment to maintain the study remains firm. The Rocky Mountain Research Station continues to maintain Taylor Woods and the two Black Hills installations. The Pacific Southwest Research Station has assumed responsibility for Crawford Creek and, with the assistance of Oregon State University, Lookout Mountain.

Early in the life of the study, meetings were held periodically to exchange information and coordinate activities. Several decades have elapsed since the last such meeting. Now may be time for another meeting. It has been evident for some time that another rethinning of the lower reserve densities is no longer possible in most installations. Increasing tree size would result in reducing the number of trees on the half-acre plots below the 12 to 15 considered to be the minimum needed to characterize growth response. A restructuring of the array of treatments might be discussed as well as an exploration of satellite studies that could be undertaken on more than one installation.

Perhaps the major reason for the longevity of this west-wide study is the foresight of the original planners in testing a range of stand densities far beyond those practiced at the time, thus providing a demonstration of the long-term stand development of even-aged ponderosa pine applicable to many current management objectives.

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Accelerating Development of Late-Successional Features in Second-Growth Pine Stands of the Goosenest Adaptive Management Area[1]

Martin W. Ritchie[2] and Kathleen A. Harcksen[3]

Abstract

This paper describes implementation and early results of a large-scale, interdisciplinary experiment in the Goosenest Adaptive Management Area in northeastern California. The study is designed to investigate development of late-successional forest attributes in second-growth ponderosa pine stands. The experiment has four treatments replicated five times and encompasses 1600 hectares, including controls. Complete treatment implementation took five years, including application of prescribed fire. Initial post-treatment measurements were conducted in 2002. Change in quadratic mean diameter averaged 12.5 cm among thinned stands. Initial estimates of post-treatment growth from remeasured diameters indicate little immediate impact of treatments on individual tree growth. However increment cores from dominant trees showed an increase in diameter growth by 11 to 14 percent in the treated plots during the first three years after treatment. Quadratic mean diameter in thinned stands was still well below that reported in reference old-growth stands. Among those stands treated with a targeted change in species composition, the mean treatment effect was an increase of 16 percent in proportion of pine basal area, with a range from 6 to 29 percent. The control treatment and thin from below treatment showed no significant change in species composition. The initial application of prescribed fire resulted in little mortality (less than 1 percent for large trees) and had no immediate impact on the diameter distribution. Logging damage observed on residual trees varied between 2 and 6 percent, depending on treatment and tree size.

Introduction

The Northwest Forest Plan (USDA and USDI 1994a) ushered in a new era for forest management on federal land in the Pacific Northwest. Intended to implement management strategies that would be more favorable to the meet the habitat requirements of the northern spotted owl (*Strix occidentalis caurina*), the Plan changed management emphasis for millions of acres of forests in Washington, Oregon and northern California. One requirement of the Record of Decision (USDA and USDI 1994b) of the Northwest Forest Plan was to establish a network of

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[1] An abbreviated version of this paper was presented at the symposium on Ponderosa Pine: Issues, Trends, and Management, October 18-21, 2004, Klamath Falls, Oregon.

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Adaptive Management Areas, thus providing managers opportunities to develop and test innovative strategies to meet management objectives.

The Goosenest Adaptive Management Area is located in northern California on the Klamath National Forest. One of the primary objectives of the Goosenest AMA is to evaluate "…the development of ecosystem management approaches, including the use of prescribed burning and other silvicultural techniques for management of pine forest including objectives related to forest health, production and maintenance of late-successional forest and riparian habitat, and commercial timber production" (USDA and USDI 1994b; Page D-13).

Harvesting in the early 1930s removed the largest and most valuable pines and most of the larger white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.). The land was transferred to the Klamath National Forest in the 1950s. Subsequently, some of the area received precommercial thinning and sanitation/salvage treatments.

The historic fire return interval in these stands created an environment favoring ponderosa pine (*Pinus ponderosa* P. & C. Lawson), a long-lived species well adapted to frequent low-intensity fires (Skinner and Chang 1996). The harvest of overstory pine trees and the elimination of fire from the ecosystem have significantly changed the size and species distributions of trees throughout the AMA.

Forests in this area are naturally regenerated second-growth stands with ponderosa pine and white fir as the primary tree species. There are lesser amounts of sugar pine (*Pinus lambertiana* Dougl.), incense-cedar (*Calocedrus decurrens* (Torr.) Florin) and red fir (*Abies magnifica* A. Murr.) at the higher elevations. Historically this area featured open park-like stands dominated by large ponderosa pine trees. Measured in 2004, breast-height age of dominant, site quality, pine averages 64 years with a range from 49 to 85. There is a component of 200+ year old trees scattered through the overstory, however the heights indicate that these were subdominant (intermediate crown class) trees in 1930, and released by the initial harvests. These trees have diameters and heights very similar to those of the trees that regenerated after overstory removal.

In 1995, the Klamath National Forest and the Pacific Southwest Research Station entered into a partnership to establish a study on the AMA to address key issues stated in the Record of Decision. An interdisciplinary research team, engaged in work at Blacks Mountain Experimental Forest (Oliver and Powers 1998, Oliver 2000), began working with managers from the Goosenest Ranger District to investigate problems and potential treatments to evaluate on the AMA. The research team considered a range of issues relating to development of late-successional forest conditions. The resulting project is a long-term study geared to evaluating the effects of large-scale treatments designed to accelerate tree growth, create more open stand conditions and re-introduce fire to the system (Ritchie and Harcksen 1999, Ritchie 2005). Our study will provide information to managers and researchers seeking a better understanding of the consequences of various management strategies in stands with a similar treatment history and species composition.

This paper describes treatment implementation and presents some early treatment effects on vegetation and coarse woody debris.

Methods

Study Design

The research team focused on three key factors in designing this study: species composition, distribution of trees by diameter class, and the role of fire in this system. In order to accelerate late successional stand attributes we wish to create stands with (1) a higher proportion of ponderosa pine, (2) more open conditions with larger trees, and (3) fire as a functioning component of the system. The study was designed to evaluate combinations of these three factors.

The AMA is located at 41.5 N latitude and 121.9 W longitude. The elevational range for the study site is 1,480 to 1,780 m (4,860 to 5,840 ft). The forest type (Eyre 1980) is predominantly Interior Ponderosa Pine (Society of American Foresters Type 237). Slopes are gentle, generally with a northwest aspect. Soils in the study area are sandy loams or loams derived from volcanic ash. A 20 to 36 cm (8 to 14 in) pumice overburden is common in the area. Site productivity is difficult to estimate in this area because current site quality trees are from two separate age cohorts, and many have not been truly free-to-grow. Based on the second-generation stand (trees with a breast height age < 70 years) site index (Barrett 1978) is 37 m (120 ft) at a base age of 100. Converted to a base age of 50 years, site index is 25 m (82 ft). This converts to a Forest Service Region 5 site class of 3.

In 1996, a sample of 20 stands in the area was selected to evaluate stand structure, species composition, and appropriateness for inclusion in the study (*table 1*). This represents pre-treatment conditions for stands within the study area. Units with recent sanitation and salvage entries tended to be more open than those without recent treatment history. Reineke's (1933) Stand Density Index (SDI) exceeded 570 (230 trees ac^{-1}), the zone of imminent mortality reported by Oliver (1995), in over half of these stands.

Table 1—*Summaries of pre-treatment quadratic mean diameter (QMD), basal area (BA), stand density index (SDI), stem density (N), and percentage of basal area in pine for stands of the Goosenest Adaptive Management Area. Units are metric with English equivalents in parentheses.*

	QMD	BA	SDI	N	Pine BA
	cm (in)	**m^2 ha^{-1} (ft^2 ac^{-1})**	**trees ha^{-1} (trees ac^{-1})**	**trees ha^{-1} (trees ac^{-1})**	**percent**
Minimum	24.6 (9.7)	26.6 (116)	474 (192)	217 (88)	15
1st Quartile	28.2 (11.1)	30.3 (132)	523 (212)	385 (156)	31
Median	29.0 (11.4)	37.0 (161)	689 (279)	568 (230)	43
3rd Quartile	39.4 (15.5)	42.9 (187)	822 (333)	778 (315)	60
Maximum	41.1 (16.2)	58.3 (254)	1084 (439)	902 (365)	73

There were four treatments: Pine Emphasis, Pine Emphasis with Fire, Large Tree, and a Control treatment. Each treatment was replicated five times in a completely randomized design (*fig. 1*). Each treatment unit was 40 ha (100 ac), with an additional buffer of approximately 100 m (328 ft). This experimental unit size was considered to be sufficient to encompass the range of small mammals and passerine birds (although this aspect of the study is not presented here). The number of replicates and variety of treatments was restricted by our ability to locate units of sufficient size with fairly uniform conditions.

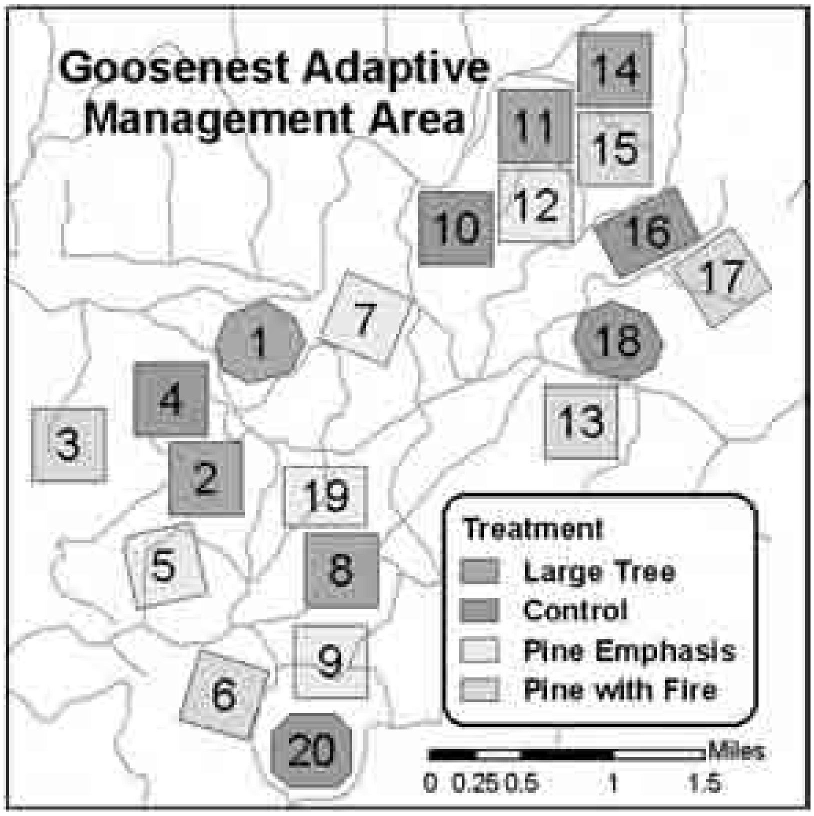

Figure 1—Map of the treatments at the Goosenest Adaptive Management Area.

The Pine Emphasis treatment was intended to accelerate growth of trees and modify species distribution by moving stands toward ponderosa pine dominance. The prescription called for a thinning from below with required retention of all dominant and co-dominant ponderosa pine and sugar pine, regardless of spacing (Ritchie 2005). Spacing for fir and intermediate or suppressed pine was determined by tree size with larger leave trees having a greater spacing than smaller trees. Because of the “must leave” requirement for ponderosa pine and sugar pine dominant and co-dominant trees, the resulting spacing was not uniform. Small aggregates of closely spaced trees were left scattered through the stand. Because artificial regeneration was the only effective means for changing species composition in areas with very dense white fir, small openings, totaling 15 percent of the area, were created in each unit and planted with ponderosa pine seedlings. The openings were deep-tilled to penetrate the pumice overburden and planted with 2-0 ponderosa pine seedlings.

The Pine Emphasis with Fire treatment began with the same thinning prescription and artificial regeneration as the Pine Emphasis. The thinning was then followed with application of prescribed fire. The application of prescribed fire took place after planting for some of the units, however fire did not carry through the plantations because the sub-soiling treatment buried surface fuels and exposed mineral soil.

The prescription called for stands to be burned repeatedly over time in a manner consistent with both the historic fire frequency, and fuel loadings sufficient to carry fire. Stands are to be re-evaluated periodically for the potential to carry fire in subsequent prescribed burns. A study is currently underway to document the historic fire frequency of the Goosenest AMA.

The Large Tree treatment was intended to address only the size distribution of trees in the stand. The prescription was a thinning from below with no regard for species. The intent was to create more large trees as quickly as possible by removing the smaller diameter trees. Since species distribution was not relevant to this treatment, there is no regeneration component. The prescription called for a size-dependent spacing guide (Ritchie 2005). The spacing varied from about 5 m to 9 m (16 to 30 ft), with larger leave trees being more widely spaced. This prescription tended to create a more uniform spacing than the Pine Emphasis treatment because there was no restriction on cutting dominant and co-dominant pine trees.

The Control treatment was consistent with custodial management. No thinning, prescribed burning, or salvage was conducted in the Controls. By comparing our treated units with Control units, we can quantify the impact of our prescriptions on a variety of treatment response variables.

All units have a 100 m (328 ft) grid of permanently monumented sample points; UTM coordinates for these points have an error of less than 15 cm (6 in). All data were spatially referenced to these grid points to facilitate integration of results from different disciplines (Oliver 2000).

Treatment Implementation

The Goosenest Ranger District completed the NEPA process, signed the Finding of No Significant Impact, and offered the timber for bid in 1997. Total revenue for the timber sale was $5.474 million. Scaling was done by weight; removals totaled 92,669 metric tons (green weight) of sawlogs and peelers, and 62,004 metric tons of chips from saplings, limbs, and tops.

To ensure conclusions would be based on the effects of the treatments, not on how the treatments were implemented, the control of all activities within the treatment units was important. To this end, the entire project was offered as one timber sale, knowing full well it would take at least three years to complete. Harvesting started in July 1998 and was completed in October 2000. In the first season, one complete replicate was thinned. In each of the two following years, two replicates were completed. All treatments were “leave-tree” marked, and all landings were located outside the treatment units. Two natural openings became the processing landings” Skid trails were pre-designated at approximately 60 m (200 ft) spacing. Operators were required to place (with feller-bunchers) or fall all trees to lead. Grapple skidders were restricted to the designated skid trails. Operations were authorized only when the soil was adequately drained to prevent compaction, and was neither frozen nor covered with snow.

First, the feller-bunchers cut and placed next to skid trails all trees from 10 cm to 45 cm (4 to 18 in). As the operation progressed across the unit, the skidders followed, dragging the bundles of trees to a “hot landing.” A heal-boom loader was used to sort the trees by size and species and load them onto a hay-rack (*fig. 2*). All limbs and tops accumulating at the landing were redistributed along skid trails by the grapple skidders. Finally, the fallers hand-felled trees greater than 45 cm (18 in) *DBH*. These trees were limbed, topped and skidded to the hot landing where they were loaded onto the hay-rack and hauled to the processing landing.

At the processing landings, trees were sorted by size and species. Ponderosa pine was processed for saw logs and white fir was processed as peelers. Trees larger than 20 cm (8 in) were limbed, topped, and cut to length by a stroke delimber. Saplings between 10 and 20 cm (4 and 8 in) were processed as clean chips for paper. Remaining saplings, tops and limbs were then processed for hog fuel.

After harvesting, the openings in the Pine Emphasis treatment units were deep-tilled and then planted.

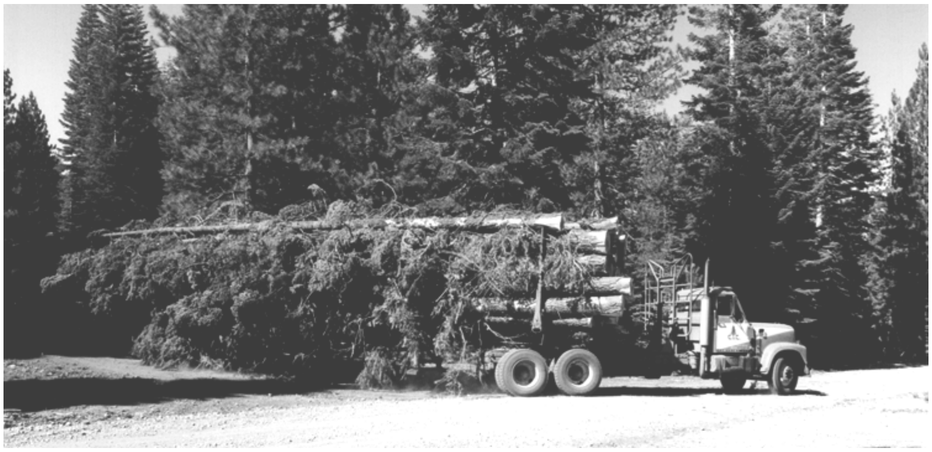

Figure 2—Trees being transported from the hot landing to the processing site at the Goosenest Adaptive Management Area.

To minimize damage to fine roots and active cambium, fall burning was used in application of prescribed fire. The initial burns were conducted in October 2001 to all five of the Pine Emphasis with Fire units. Burning of each unit took up to three days. Although Forest Service crews typically use a strip burning technique, tree centered firing was widely used when possible (Weatherspoon and others 1989).

Vegetation Sampling Methods

Each treatment unit has approximately 36 grid points, at 100 m (328 ft) spacing, depending on orientation of unit boundaries. The number is slightly less than 40 due to restrictions on grid point establishment near the plot boundary. Every other grid point was selected for vegetation sampling, such that the sampled grid points are spaced at 141 m (463 ft) intervals.

The initial post-treatment vegetation sample was conducted in 2002, the season following completion of all harvesting and the initial application of prescribed fire. Typically such sampling would be done after the end of the growing season. However, due to the scale of this study, that was not possible. We may have to adjust growth to account for the period lengths between subsequent remeasurements (Flewelling and Peters 1997). The initial vegetation sampling and permanent plot establishment took two three-person crews the entire summer to complete.

At each sampled grid point, three nested fixed-area plots were established. Trees > 29.2 cm (11.5 in) breast-height diameter were sampled on a 0.81 ha (0.2 acre) plot centered on the grid point. Trees from 9.1 to 29.2 cm (3.6 to 11.5 in) diameter at breast height were sampled on a 0.020 ha (0.05 acre) plot, also centered on the grid point. Trees < 9.1 cm (3.6 in) diameter were tallied by 2.5 cm (1-inch) diameter classes on 0.004 ha (0.01 ac) plots.

Species, height, diameter, and height to crown base were recorded for each tree. We also recorded damage, with particular emphasis on damage related to treatment implementation. Damage was primarily bole scars from either skidding or fire scorch.

In order to evaluate growth on dominant (site-quality) trees, we sampled 36 site quality trees (Barrett 1978) on three treatment units. Increment cores were obtained from each tree and radial growth rates (periodic annual increment) were observed for the 2-yr period immediately before and after treatment. We conducted a paired t-test on pre and post treatment growth for each of the treatment units.

Coarse woody debris was sampled on a 100 m (328 ft) transect using the methods described by Brown (1978). Transects were centered on every grid point where trees were sampled. Material was identified as being either decay class 1 (recent material), class 2 (some decay but still structurally intact), or class 3 (advanced decay with no structure).

Results

The study was designed to evaluate responses over a 50-year time line. However, initial results can indicate something of the potential for effectiveness over the long-term as well as provide an indication of short-term changes in stand structure and habitat.

Stand Structure

The initial entry increased the proportion of pine for both Pine Emphasis and Large-Tree treatments, although the increase tended to be greater for the Pine Emphasis units. Large-Tree units showed a small increase in pine because a thin from below, with a dominance of fir in the understory, necessarily produces a reduction in fir (*fig. 3*). As the pine plantations develop, this change in proportion of pine should become more pronounced in the Pine Emphasis units.

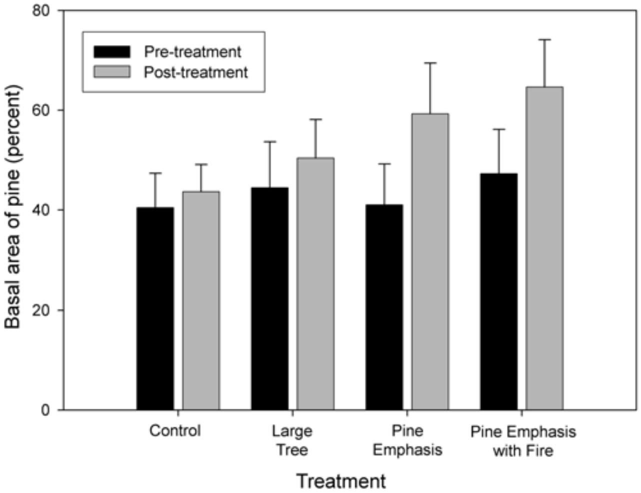

Figure 3—Change in species composition for four treatments in the Goosenest Adaptive Management Area (error bars = standard error).

A comparison of pre- and post-treatment diameter distributions, averaged across treatment units (*fig. 4*), shows that the skewed pre-treatment size distribution has been modified to a more symmetric distribution. There were very few trees above 60 cm (24 in) in diameter throughout the study (generally less than 15 trees per ha or 6 trees per acre). The Pine Emphasis units appear to have a higher proportion of pine, particularly in the middle diameter classes.

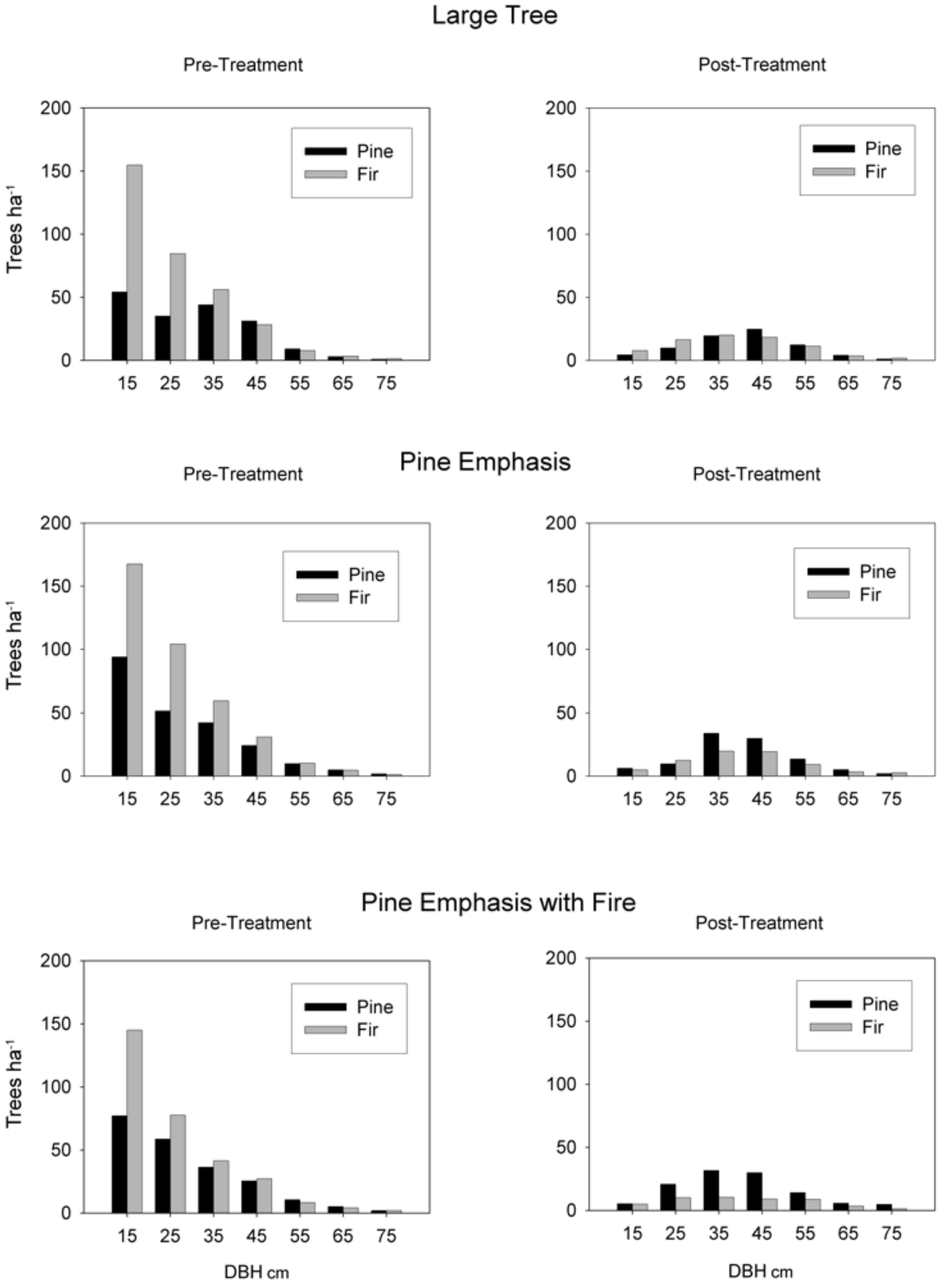

Figure 4 — Diameter distributions before (1996) and after (2002) treatment, averaged across experimental units, for the treatments (excluding Controls) in the Goosenest Adaptive Management Area.

Because treatments targeted the lower diameter classes, there is a pronounced “chainsaw effect” on *QMD*. *QMD* increased in treated units an average of 12.5 cm (4.9 in), ranging from 2.3 to 19.3 cm (0.9 to 7.6 in); *QMD* is substantially higher than in controls, as seen in the following tabulation.

Control	**Large Tree**	**Pine Emphasis**	**Pine Emphasis with Fire**
30.8 cm ± 1.5	43.5 cm ± 1.4	45.1 cm ± 1.6	43.6 cm ±1.6

The shape of the current diameter distribution in these stands is more consistent with hypothesized late-successional forest structure, where frequent low-intensity fires maintained a more open understory.

Number of large trees, defined as those > 60 cm (24 in) in diameter, varies across all treatments (*fig. 5*) from 5 to 35 trees ha^{-1} (2 to 14 trees ac^{-1}). Basal area was reduced an average of 29 percent for Pine Emphasis plots and 35 percent for Large Tree units. Stand density index (Reineke 1933) now ranges between 356 (143 ac^{-1}) and 453 (182 ac^{-1}) for the five Large Tree units and between 326 (131 ac^{-1}) and 572 (230 ac^{-1}) for the five Pine Emphasis Units.

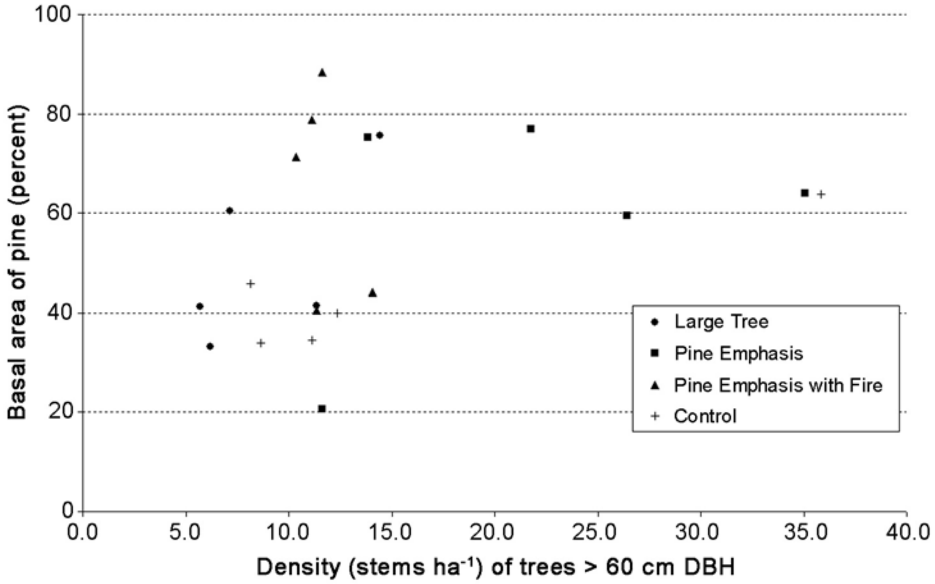

Figure 5 — Post-treatment (2002) proportion of pine, and number of large trees, defined as those > 60 cm (24 in) *DBH*, for treatment units in the Goosenest Adaptive Management Area.

Mortality and Damage

Initial mortality rates are low, indicating little damage to residual stands immediately after harvest and application of prescribed fire. Logging damage was also generally low (estimated below 6 percent) regardless of treatment and tree size. Damage was significantly less (P=0.001) among trees > 50 cm (20 in) in the Large Tree treatment when compared with the Pine Emphasis treatments (*fig.6*). This difference may be attributable to a more clumpy distribution of trees in the Pine Emphasis units.

Burn damage to residual trees in the units receiving prescribed fire was also low. Initial mortality among trees > 29 cm (11.5 in) diameter was less than 1 percent. Fire-related mortality among trees between 9 and 29 cm (3.5 and 11.5 in) *DBH* averaged 3.5 percent. This mortality rate is probably an underestimate because it only includes those trees killed within the first year after burning and does not reflect any secondary mortality that may occur over the next several years (Ryan and others 1988).

Initial Growth Response

Thinning in ponderosa pine generally increases growth on residual trees (Cochran and Barrett 1995, Cochran and Barrett 1999). A preliminary analysis of two-year thinning response, using a sample of trees from three different treatment units, revealed no significant change in tree growth in response to thinning. However, this may be due to high variability within treatments or because it may take several years for trees to build sufficient crown and roots to take advantage of the increase in available resources. The only evidence of accelerated growth was among small trees. Because most of the smaller trees were removed, there are very few residual trees less than 30 cm (12 in) in the treated units, so sample size was limited.

The treated plot with the greatest gain in diameter increment was the Pine Emphasis plot, with an increase of 14±12 percent (±90 percent confidence interval). In the Large Tree and Pine Emphasis with Fire Treatments we observed approximately 11±23 and 11±22 percent radial growth increase respectively. Note that the confidence interval includes zero for both the Large Tree and Pine Emphasis with Fire Treatments.

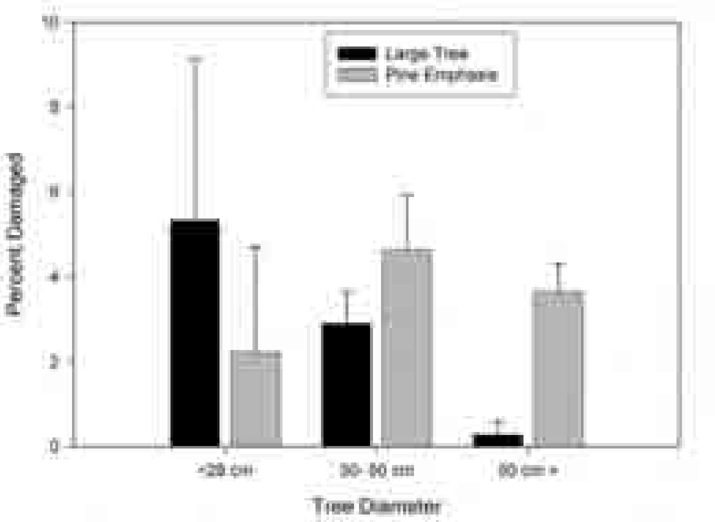

Figure 6 — Percentage of damaged trees by diameter class for Large Tree and Pine Emphasis Treatments (error bars = 1 standard error).

Woody Debris

The primary treatment impact on woody debris was from fire. There was no detectable difference between mechanical treatments. However a comparison of Pine Emphasis units and Pine Emphasis with Fire units showed a significant reduction in volume. Debris in a more advanced state of decay is substantially (73 percent by volume) reduced by fire (P=0.013). Material in a less advanced state of decay was reduced about 43 percent on average (*P*=0.120). Differences between burned (Pine Emphasis with Fire) and unburned (Pine Emphasis) was limited to smaller diameter class material (*table 2*).

Table 2 — *Post-treatment means (with standard error) for number of woody debris pieces ha^{-1} for Pine Emphasis (no burn) and Pine Emphasis with Fire (burn), by size class.*

	Length					
Diameter	**< 2m**		**2 – 4 m**		**4 – 6 m**	
	burn	no burn	burn	no burn	burn	no burn
< 25 cm	171 (16)	697 (186)	49 (8)	227 (65)	16 (6)	81 (20)
25-50 cm	75 (18)	175 (37)	48 (15)	81 (29)	14 (9)	34 (11)
50-75 cm	19 (4)	18 (4)	6 (3)	6 (3)	6 (3)	18 (8)
> 75 cm	3 (3)	6 (4)	3 (3)	3 (3)	0 (0)	1 (1)

Discussion

Observations of changes in growth rate are small at this point, primarily because it takes a few years for trees to develop leaves and roots to exploit the increased availability of resources. The growth rate increase observed for dominant trees appears to be less than that for stands with similar levels of growing stock observed by Cochran and Barrett (1999) who found that diameter growth increased by approximately 40 percent. However Cochran and Barrett (1999) had 30 years of observations. The results suggest that growth rates in the prescribed fire units are slightly less than those in Pine Emphasis units. This is consistent with research showing that diameter growth was suppressed, in the short term, following application of prescribed fire (Busse and others 2000).

Differences in diameter distribution are fairly subtle, except for the obvious difference between harvested and control units. Mortality processes initiated by fire and insects may take an extended period to significantly impact the dynamics of these stands. The expected distribution of large trees in late successional pine stands is difficult to ascertain. One of the few un-disturbed sites left in California averaged 17 trees ha^{-1} in trees > 60 cm (7 trees ac^{-1} >24 in) *DBH*, with a *QMD* of 53 cm (21 in) (Beaver Creek Pinery, data on file PSW Research Station, Redding CA). Treated unit *QMD* ranges from 36 to 48 cm (14.0 to 18.7 in). Youngblood and others (2004) found *QMD* of 60 cm in (24 in) old-growth ponderosa pine stands and density of overstory trees ranging from 15 to 53 ha^{-1} (6 to 21 ac^{-1}). With current growth rates, it could take a decade or more to increase stand diameter another 10 cm (4 in).

As one would expect, there were no discernable trends with regard to the number of trees > 60 cm (24 in) in the 2002 post treatment data. If any treatment effect is observed, it may take decades to detect.

The snag retention in units with prescribed fire is not consistent with published guidelines that range from 2-10 snags ha^{-1} (5 to 25 ac^{-1})(Bunnell and others 2002; USDA and USDI 1994b; USDA 2004). Maintenance of snags with frequent low-intensity fires is difficult because fire-induced mortality is low while many existing snags are consumed.

The rate of consumption of down material by fire at this site suggests that, with frequent application of prescribed fire, there will be very little coarse woody debris present, particularly with regard to material in an advanced state of decay.

Mortality and damage resulting from thinning was kept at low levels by laying out all skid trails in advance, marking all leave trees, and closely supervising operators.

Acknowledgements

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Positive Seedling-Shrub Relationships in Natural Regeneration of Ponderosa Pine[1]

Christopher R. Keyes[2] and Douglas A. Maguire[3]

Abstract

An understanding of natural regeneration processes, and the stand structural features that influence those processes, is vital to attaining goals associated with natural regeneration. This paper discusses natural regeneration concepts and the interactions that occur between shrubs and natural regeneration of ponderosa pine. The interactions observed recently in a series of related seedling recruitment studies in central Oregon are summarized. Evidence suggests a positive relationship between shrubs and the occurrence of ponderosa pine seedlings during the first two years of the seedling establishment phase. The vast majority of germinant mortality occurs during summer months, and especially the weeks immediately after emergence. Shrubs do not enhance emergence rates, but do enhance overall recruitment rates by reducing rates of first-summer mortality. The mechanism driving these patterns appears to be microenvironmental amelioration by shrubs, abating germinant moisture stress and desiccation. The manifestation of this relationship is a positive spatial relationship between shrubs and small seedlings during this establishment phase.

Introduction

For most forest management objectives, the recruitment of a cohort of seedlings is the first and most critical process following regeneration harvest or stand-replacing disturbance. The spatial and temporal patterns of seedling recruitment set the stage for all subsequent stand development patterns; hence, the seedling recruitment stage is the first step in determining future stand structures, habitat conditions, susceptibility to disturbances, and silvicultural options. An understanding of the natural regeneration processes in forests of ponderosa pine (*Pinus ponderosa* P. & C. Lawson), and the stand structural features that have bearing on those processes, is vital to manipulating controllable factors in order to successfully secure natural regeneration. This subject is particularly relevant under the current Forest Service paradigm of ecosystem management, wherein reliance upon natural processes in natural systems management is stressed, and funds for reforestation are scarce.

An understanding of regeneration processes is equally important for those management objectives wherein regeneration is undesirable. For example, managers may wish to maintain crown fire resistance by promoting ponderosa pine stands with

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open-understory structures that are free of ladder fuels. Currently, management and research in this region is directed toward dealing with the existing condition of dense ponderosa pine stands. However, it is worthwhile to analyze the processes that have contributed to the development of these dense stands in the era of fire suppression and exclusion. It is reasonable to theorize that the effects of fire exclusion on ponderosa pine regeneration have been both direct and indirect. If the relationship of shrubs to seedling recruitment is positive, for example, it then follows that one effect of fire exclusion may have been the indirect proliferation of pine seedlings by enabling dense shrub understories to establish and persist. An understanding of these and other potential indirect effects of fire exclusion on pine proliferation is essential to prescribing practices that will create and sustain stands with crown fire resistance.

Despite an abundance of previous research on the subject, the effects of shrubs on processes of ponderosa pine natural regeneration are still poorly understood. Research efforts have been geographically limited, short in duration, and restricted to a single stage of regeneration, a situation that is typical of regeneration research in other species and regions as well (Clark and others 1999). The result has been sometimes-conflicting and often-confusing advice for forest managers dealing with regeneration objectives. This paper addresses the relationship of shrubs to the establishment of ponderosa pine regeneration with a discussion of shrub effects within the context of recent conceptual models of facilitation and competition that have been applied in similarly xeric ecosystems. We also offer evidence of positive seedling-shrub relationships in a summary of the results of three integrated establishment-phase regeneration studies that were recently conducted in central Oregon.

Constraints on Regeneration

Several ubiquitous biological and physical factors are capable of limiting seedling recruitment in ponderosa pine forests, and often do so (*table 1*). Many of these factors may occur simultaneously during any given year, thus severely curtailing opportunities for regenerating ponderosa pine without human intervention. Among those factors, moisture stress represents the most significant of practicably controllable factors. The potential roles of phytotoxins as chemical regeneration inhibitors have also been investigated (Kelsey and Harrington 1979), but were revealed to be absent.

In previous reviews of ponderosa pine regeneration processes, Heidmann (1992) and Barrett (1979) both identified aspects of climate having greatest influence on the establishment of seedlings from seedfall. Heidmann (1992) opened his review article, *Regeneration Strategies for Ponderosa Pine*, with the bluntly succinct statement, “Regenerating ponderosa pine (*Pinus ponderosa*) is difficult.” He elaborated that,

> The primary obstacle to regeneration of this species throughout its natural range is drought… Annual precipitation in the western and southwestern United States is generally adequate for tree growth but erratic distribution during the year makes seedling establishment difficult.

Table 1—*Factors that potentially limit ponderosa pine regeneration, the affected regeneration stages, and relevant literature specific to ponderosa pine (references not specific to ponderosa pine are denoted by parentheses).*

Factor	**Regeneration stage**	**Sample reference**
Seed deficiency	flower, seed	Pearson 1923 Fowells and Schubert 1956 Daubenmire 1960 Curtis and Foiles 1961 Shearer and Schmidt 1970 Foiles and Curtis 1973 Dahms and Barrett 1975 Barrett 1979 McDonald 1992
Molds	seed	Fowells and Schubert 1951 Wagg 1958 Roth 1970
Predation	seed, germinant, seedling	Pearson 1913 Eastman 1960 Hooven 1966 Cochran 1970 Blake and others 1986 Blake and others 1989 Shearer and Schmidt 1971
Damping-off fungi	germinant	Wagg and Hermann 1962 Roth 1970
Unfavorable forest floor substrate	germinant	Larsen 1924 Foiles and Curtis 1965
Litterfall burial	germinant	(Koroleff 1954) (Tappeiner and Helms 1971)
Excessive heat	germinant	Baker 1929 Larson 1967
Moisture stress, drought	germinant, seedling	Pearson 1923 Hermann 1968 Larson and Schubert 1969 Cleary 1970 Djavanshir and Reid 1974
Frost heaving	germinant, seedling	Heidmann 1976
Summer frosts	germinant, seedling	Cochran 1972 Cochran 1984
Excessive shade	germinant, seedling	Pearson 1936 Atzet and Waring 1970

In his report, *Silviculture of Ponderosa Pine the Pacific Northwest*, Barrett (1979) summarized the primary constraints on ponderosa pine regeneration with the observations:

> Prolonged moisture for germination, growth, and life during the critical heat of July and August is critical… Germination usually occurs in early spring, but rapid drying of the germinating medium often causes death of the seedling… Because summers are usually dry, conservation of moisture is critical.

In the Pacific Northwest, ponderosa pine forests experience a harsh climate with cold winters and hot, dry summers. Annual precipitation is limited, and a regular period of drought occurs throughout much of the growing season (*fig. 1*). On the east slope of the Cascade Range, these climatic conditions are exacerbated by the presence of relatively young, poorly developed, mineral deficient, and highly porous volcanic soil types that dominate the region. These climatic and edaphic conditions present, through the moisture stress they induce, significant environmental challenges to tree regeneration, survival, and growth.

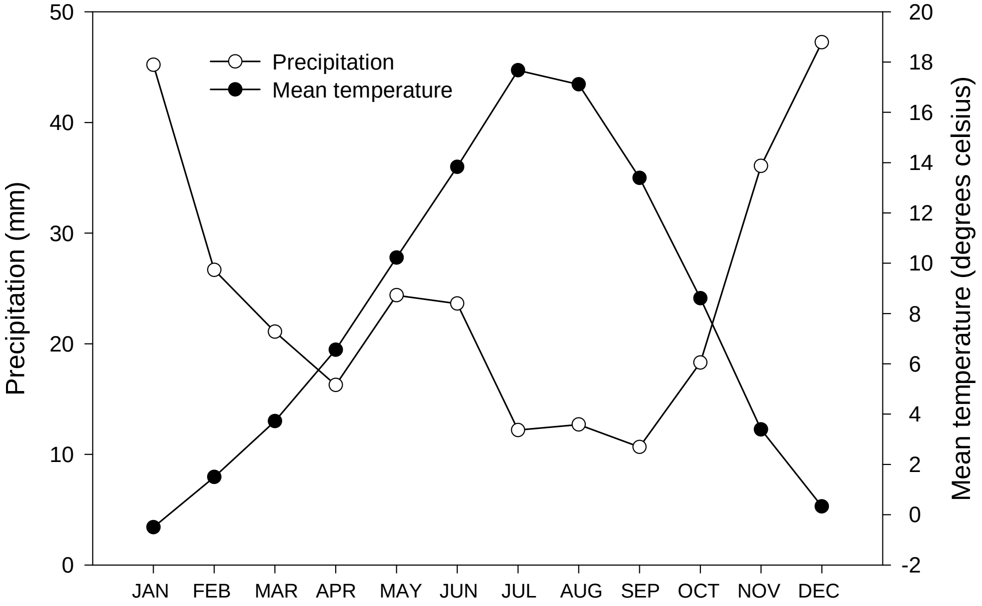

Figure 1—Climatic conditions in the central Oregon area illustrating the annual coincidence of high temperatures with low precipitation (1928-2004 mean values from National Climate Data Center cooperative station no. 350694 at Bend, Oregon).

Given these constraints, the microenvironments most favorable for seedling establishment will be those where moisture stress is abated. Moisture stress may be abated by a reduction in solar radiation, wind, soil temperature or ambient temperature; by an increase in relative humidity; and by retention of soil moisture near the forest floor surface further into the summer months.

Shrub Influences on Regeneration

This challenging regeneration environment suggests the potential for positive effects of shrubs on ponderosa pine seedling recruitment by their microenvironmental influences and attendant effects on germinant and seedling moisture stress. Past observations of regeneration have provided indications of numerous influences of shrubs on ponderosa pine recruitment processes, both positive and negative (*table 2*).

Table 2—*Potential influences of shrubs on ponderosa pine seedling regeneration.*

Positive influences	Negative influences
Enhanced soil fertility via N fixation	Accelerated soil moisture depletion
Extended soil moisture retention via shading	Excessive shade
Lower ambient temperatures via shading	Reduced seedling growth
Physical browse interference	Enhanced seed-predating mammal habitat
Enhanced soil moisture via hydraulic lift	Burial by litterfall
Improved seedling form	Consumption by fire (via fuelbed continuity)
Reduced thermal flux and frost heaving	

But in some cases these influences are conflicting. On balance, do shrubs promote or disfavor tree regeneration? Do they facilitate seedling survival by ameliorating the harsh climate, or are they important competitors for the highly-restricted below-ground resources? Moreover, do their effects change over the course of seedling establishment and growth? The situation is likely characteristic of the so-called ‘seed-seedling’ conflict described by Schupp (1995). Schupp determined that inter-specific conflicts and intra-specific conflicts had been well addressed in ecological literature, but that intra-individual conflicts had received considerably less attention. For trees, intra-individual conflicts occur when the conditions favorable for growth are diametrical to those favorable for establishment (*fig. 2*).

Such conflicts have been recognized for centuries by silviculturists and have been addressed operationally in the utilization of shelterwood regeneration systems. Shelterwood systems, which have been employed as formal systems since at least the 18th century (Kostler 1956), were developed to balance the requirements for seedling establishment with those for seedling growth. Under these systems, partial live tree overstories are retained at harvest in order to facilitate regeneration by providing a seed supply and a shaded forest floor (ameliorated microenvironment); once an understory cohort has become established, the overstory is removed to maximize growth of the new cohort (Matthews 1989, Smith and others 1997). Although this concept is well known and understood by foresters, other potential intra-individual conflicts have not been widely recognized by foresters or adequately investigated by researchers. For ponderosa pine, investigations into seed-seedling conflicts during the periods of germinant emergence and establishment are absent.

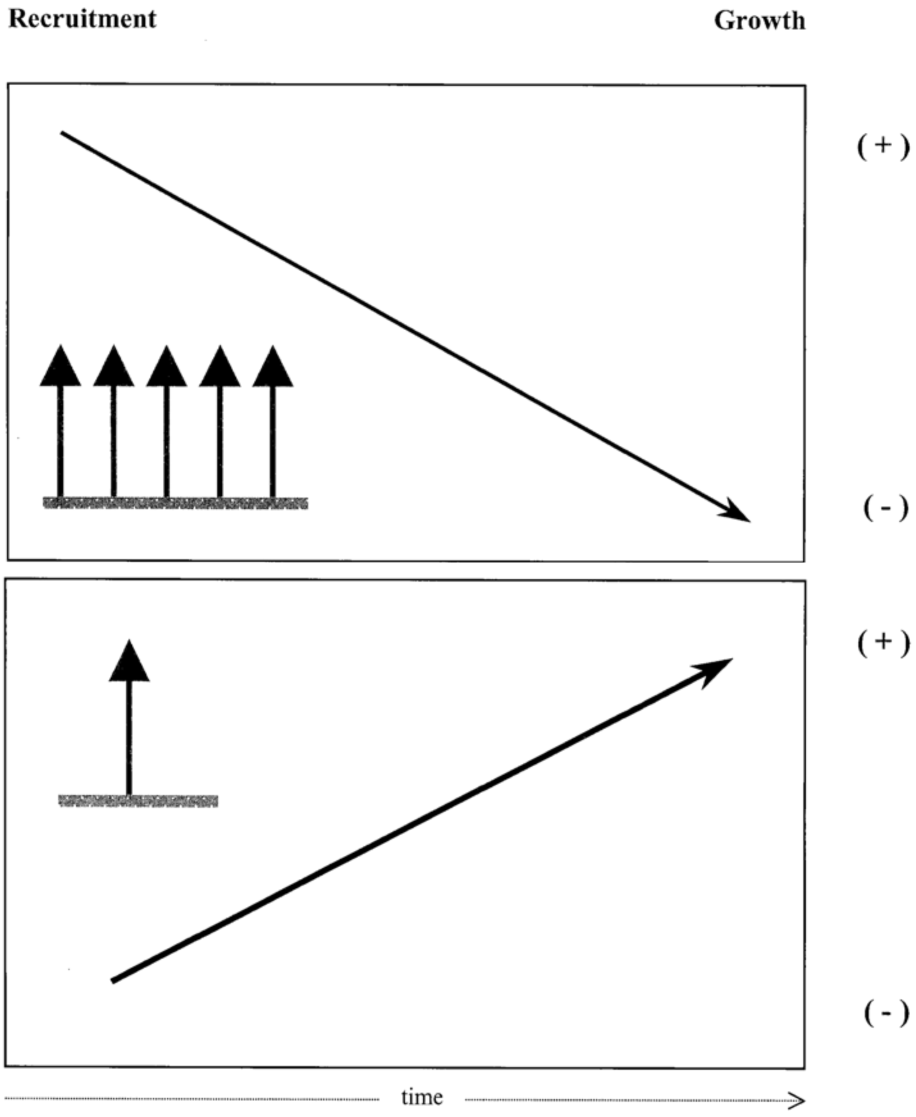

Figure 2—Illustration of intra-individual regeneration conflicts. Above, germinants recruited beneath partial live overstories have greater initial survival potential, but survivors of the establishment phase have reduced growth potential. Below, seedlings establishing in the open or beneath a sparse overstory have reduced survival potential, but survivors have greater growth potential.

In ecosystems where plants are similarly subjected to severe environmental stresses, evidence is mounting that positive interactions (facilitations) are common. Positive interactions and facilitated recruitment in vascular plants are most often observed in those natural communities where either physical stress is great or consumer pressure is intense (Bertness and Callaway 1994). For example, in many ecosystems where severe desiccating conditions are typical – including grasslands, tundra, sand dunes, deserts, salt marshes – positive interactions frequently occur (Holmgren and others 1997). All observed plant interactions can be generalized as the net result of co-occurring facilitation and competition (Callaway and Walker 1997). In xeric plant systems, facilitation is the dominant force when the costs associated with reduced light levels are more than offset by improved water relations (Holmgren and others 1997). This balance of competition and facilitation is dependent upon plant life stages, so the relationship between individuals may differ over time. Adapting these concepts to ponderosa pine systems, it is reasonable to theorize that ponderosa pine germinants may benefit from shrub presence during the establishment phase in a facilitative relationship that becomes competitive once seedlings become established and increase their demands for growth resources.

Regeneration Field Studies

The relation of shrubs to ponderosa pine seedling regeneration was recently analyzed in a series of related field studies located in central Oregon (Keyes 2002, Keyes and Maguire 2000). The goal of the project comprising the studies was to quantify the relative effect of stand structural features on the earliest phases of seedling recruitment. An objective of each study was to determine whether the relationship of shrubs to seedlings was one of attraction (positive spatial association) or repulsion (negative spatial association). A summary of the relevant findings of the three field studies is presented here.

The first of these is a case study of regeneration in an old-growth ponderosa pine stand. The second is an observational study of ponderosa pine germinant emergence and 1-2 season survival, and the relationship of those processes to the presence of shrubs. The third is a seed-sowing experiment relating the presence of shrub cover to ponderosa pine germinant emergence and 1-season survival rates.

Each of the studies presented here was conducted in pure ponderosa pine stands on the lower east slope of the Cascade Range in central Oregon. All stands were at sites that were flat or nearly so. The case study was conducted in one undisturbed stand at the Metolius Research Natural Area, located 22 km northwest of the town of Sisters. The observational study was conducted in four stands at the eastern edge of the Deschutes National Forest southeast of the city of Bend. The experiment was conducted in four stands of the Deschutes National Forest; two were located on the forest’s eastern edge, and two were located on the forest’s western half, near the Pringle Falls Experimental Forest. The stands in the observational study and experiment had been subjected to operational regeneration harvests performed within the past 20 years, but retained partial overstories. The stands in the observational study had not been cut within the previous decade; the stands in the experiment had been cut more recently.

Case Study: Regeneration in an Old-Growth Stand

In the first study, an analysis was conducted of the spatial relationships between ponderosa pine small seedlings (≤10 cm tall), large seedlings (10.1-140 cm tall), saplings (0.1-10 cm dbh), overstory trees (≥10 cm dbh), and shrubs (Keyes and others 2001). The study was conducted in a multi-cohort old-growth stand containing a range of tree sizes (including 4 or more trees in each 10-cm dbh class from 0-100 cm) and was virtually pure in ponderosa pine. A grid of 28 2-meter radius regeneration plots was installed within a 4.5-hectare study area. A tally was made of the number of seedlings and saplings in each plot. Percent shrub cover (primarily antelope bitterbrush; *Purshia tridentata* Pursh DC.) was visually estimated to the nearest five percent from illustrations of shrub cover drawn in the field. Local overstory basal area was calculated within a 5-m radius of each plot center. Simple and multiple Poisson regressions were used to identify significant relationships between seedling or sapling density (stems per hectare) and percent shrub cover. It was theorized that positive or negative spatial associations would be suggestive of facilitative attraction or competitive repulsion.

The analysis revealed that small seedlings were negatively related to local overstory basal area (P=0.0007) and sapling density (P=0.0029). Large seedlings were also negatively related to local overstory basal area (P=0.0043) and sapling density (P=0.0269). Shrub cover, however, was negatively related to saplings (P=0.0003), had no statistically-significant relationship to large seedlings (P=0.0888), and was positively related to small seedlings (P=0.0141). Consistent with these results, when local overstory basal area was included in two-factor models, positive relationships of shrubs to small seedlings (P=0.0005) and large seedlings (P=0.0070) were observed. These spatial associations suggest a facilitative relationship between shrubs and seedlings that yields to a competitive relationship between shrubs and saplings. They also suggest that the presence of shrubs benefits seedlings in a way that saplings and overstory trees do not.

Observational Study: Patterns of Recruitment

In the observational study, the temporal patterns of seedling emergence and survival over two seasons (one winter) were addressed. All fresh germinants from natural seedfall were identified in 64 10-m^2 circular regeneration plots systematically located in grids within 1-hectare study areas in four stands. Germinants were labeled as shaded by live local vegetation, shaded by dead local vegetation, or unshaded. Shrubs were antelope bitterbrush and greenleaf manzanita (*Arctostaphylos patula* Greene). Two cohorts of germinants were tracked over the course of 1-2 years for patterns of mortality.

Only about 3 percent of viable seed resulted in emergents. A small fraction of emergents survived their first summer. Overall, most mortality occurred immediately after emergence, and nearly all mortality occurred during the summer months. Local shade did not improve emergence rates. Most germinants (72.4 % in 1999, 57.8% in 2000) emerged without understory shading, and in fact the proportion of shaded emergents during 1999 (27.6%) was significantly lower than the expected proportion assuming random germination and survival, and based on estimates of percent shrub cover (34.5%) (Fisher's Exact Test; P=0.0138). But although most new germinants were unshaded, few of those survived their first summer (0.8% in 1999, 4.4% in 2000). Most germinants surviving the first summer received local shade, and the only survivors of two summers were shaded germinants. During both years, the proportion of survivors within shade was significantly greater than the expected proportion (Fisher's Exact Test; P<0.0001 during 1999, P=0.0005 during 2000). This positive effect of shrubs on germinant survival were quantified in cumulative mortality distributions that differed significantly (at α=0.05) among the three conditions of shading, with mortality extended furthest into the summer by live shade, and secondarily by dead shade.

Experimental Study: Shrub Effects on Recruitment

The experimental study was a seed-sowing experiment consisting of ponderosa pine seed sown under controlled conditions to test for differences in germination and survival rates among shrub-shaded and unshaded locations. Shrubs were antelope bitterbrush, greenleaf manzanita, and snowbrush ceanothus (*Ceanothus velutinus* Dougl. ex Hook.). Survival was monitored over one summer under conditions controlled by the split-plot design of the experiment. Whole plots tested caching and substrate treatments. Because mechanisms driving the germination and survival patterns were of interest in this experiment, measurements were taken to assess whether differences in relative humidity, air temperature, soil temperature, and light intensity existed between shrub-shaded and unshaded treatments.

Less than 30 percent of all seed sown in fall resulted in live germinants the following spring, and just 5 percent resulted in live germinants by the following fall. Shrubs had no effect on germinant emergence rates ($P=0.349$). However, mortality to germinants under shrubs occurred at a slower rate than germinants in the open, and hence establishment rates were higher ($P=0.032$). At unshaded sites, the mortality rate was greatest immediately after germination, whereas the rate remained relatively constant throughout the summer at shrub-shaded sites. As a result, by midsummer the survival rate among shaded germinants was 2.2 times the survival rate of unshaded germinants; by November, nearly 4 out of every 5 germinants were shaded by shrubs. Shrub-sheltered germinants received about one-half to one-third the solar radiation received by adjacent unshaded germinants (data collected 31 July to 1 August 2000). This shading was not reflected in relative humidity or air temperature differences, but did result in cooler soil temperatures, especially during the hottest parts of the day ($P<0.05$).

Summary of Study Findings

The studies described above focused on establishment-phase regeneration processes during the first two growing seasons after emergence. They provide additional evidence supporting moisture stress as a major factor in ponderosa pine seedling recruitment. For this early phase of regeneration, the relationship of shrub presence to germinant recruitment is a positive one. Results from these studies can be summarized as follows:

- Seedling spatial patterns are positively associated with shrub cover and negatively associated with sapling and overstory tree density, suggesting that shrubs benefit seedlings in ways that saplings and overstory trees in the same stand do not.
- The positive spatial association of shrubs to regeneration does not persist once seedlings achieve sapling proportions (taller than 140cm). Adopting the conceptual model of Callaway and Walker (1997), this suggests that the relationship transitions from one of facilitation to one of competition between the seedling and sapling stages.
- Local shade increases recruitment rates over unshaded sites by reducing rates of germinant mortality. Desiccation is the primary source of mortality, and nearly all mortality occurs during summer (most during the first summer).

- Probability of germinant survival is high if desiccation can be avoided during the first weeks after germination. The germinant survival rate in the second season is higher than survival rate in the first season.
- Shrub cover is not associated with greater emergence rates but greatly improves germinant survival. Establishment from seeds sowed beneath shrubs was several times that from seeds sowed in the open.
- Shrub cover appears to indirectly reduce moisture stress in germinants. The specific mechanism by which shrub cover affects germinants may be by reducing soil temperature or by reducing solar radiation.

Taken together, these findings indicate that the objective of securing natural regeneration in this region will be best served by the retention of understory shrubs. They also indicate that by enabling dense shrub understories to establish and persist, the practice of fire suppression has had an additional (indirect) effect on the development of dense ponderosa pine stands in this region during the past century. Dense shrub understories are likely to continue contributing to the development of ladder fuels by their positive effect on seedling recruitment. Therefore, understory shrub structure should be taken into consideration by managers responsible for prescribing long-term silvicultural treatments to minimize fire behavior.

The time frame encompassed by these studies represents the period when natural regeneration is most strongly influenced by stand structural features, but the studies should be regarded as a foundation for further research into ponderosa pine regeneration processes. Monitoring of seedlings from germination onward is necessary to identify the long-term influences of shrubs on regeneration. This would serve to clarify the role of shrubs on survival of new seedlings beyond the first two seasons, as well as the effects of shrubs on seedling growth. It will probably be useful to analyze shrub cover as a quantitative variable (such as distance from dripline or distance from shrub center) to describe the range of conditions for seedling regeneration within the area of shrub crowns. Differences in regeneration effects among different species of shrubs common in ponderosa pine forests – including antelope bitterbrush, greenleaf manzanita, and snowbrush ceanothus – should be determined, if these differences can be assessed without being confounded by site quality.

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Managing Ponderosa Pine Forests in Central Oregon: Who Will Speak for the Soil?[1]

Matt D. Busse[2] and Gregg M. Riegel[3]

Abstract

The soils of the central Oregon pumice plateau are relatively young and infertile, yet support an array of plant diversity and growth in the region's pine forests. Whether these coarse-textured, pumice and ash soils are resilient to forest disturbance is not well understood. We present results from a long-term experiment that examined changes in soil quality in response to combinations of thinning and repeated prescribed fire. Soil quality was generally unaffected in fifteen years following pre-commercial thinning. The soils were also resilient to fire with the exception of the loss of nearly 25 percent of the ecosystem's total nitrogen due to burning in 1991 and 2002. Natural replacement of N by N-fixing shrubs such as snowbrush ceanothus (*Ceanothus velutinus* Dougl. ex Hook.) and antelope bitterbrush (*Purshia tridentata* (Pursh.) DC.) is an important means to offset N losses, along with careful planning of burn prescriptions to limit forest floor N consumption.

Introduction

How can the
non-farmer jet-set bureaucrats

Speak for the green of the leaf? Speak for the soil?

Gary Synder, 1974

Soil is cursed. This complex medium that covers much of the surface of our planet and helps ensure the survival of the human race suffers from an unavoidable fact: it is terminally boring. It just sits there with little aesthetic value, unable to attract attention by changing colors, growing, or communicating. Adding insult to injury, soil is known synonymously as dirt, the common irritant found under fingernails and on pant legs. This is not to suggest that the value of soil is lost upon society, however. Far from it. Both historic and contemporary examples of degraded soils and the human hardships that often follow (Lal and others 2004) provide a constant reminder that soil is (1) an indispensable natural resource, (2) generally nonrenewable in our lifetime, and (3) susceptible to disturbance and mismanagement. Thus, preventing soil degradation is a common-sense requirement for managing forest lands, regardless of the desired social or economic product, a requirement of both legal and ethical proportions for federal land managers as directed by the United States Congress.

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Have the soils of central Oregon avoided productivity losses during the past century of unprecedented changes in forestry practices (fire suppression, railroad logging, and aggressive fuel reduction)? Or, are there clear examples of irreversible soil damage? Unfortunately, no archived soil collection exists from the pre-fire suppression era, and, thus, a true evaluation of soil changes is not possible. As a substitute, we present results from a forest manipulation experiment (the Black Bark Study) that tracks recent trends in soil quality associated with forest management practices. Changes in soil quality and plant growth in 15 years following a combination of prescribed fire, thinning, and nutrient additions are explored. In particular, we discuss the potential loss of soil quality associated with repeated prescribed burning.

Soil Quality

Soil quality is a fairly new term to the discipline of soil science. Thus, a brief discussion of its origin, definition, and application to forestry is appropriate before examining possible implications for central Oregon soils.

Coined by agricultural soil scientists in the early 1990s (Doran and Parkin 1994), soil quality is a holistic, feel-good concept derived from its cousin and predecessor, soil productivity. Consider the definition of these two terms from the Soil Science Society of America (http://www.soils.org/sssagloss):

> *Soil productivity: the capacity of a soil to produce a certain yield of crops or other plants with a specified system of management.*
>
> *Soil quality: the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health.*

The language used in defining soil quality is key; terms like function, ecosystem, sustain, environmental quality, and health are pleasing to our 21st-century ears. As a result, soil quality is politically correct, in vogue. Soil productivity, although often used synonymously with soil quality, is a plant-centric term perhaps more fitting for agricultural soils, where farmers can track annual crop yields and examine short-term (5 to 10 year) productivity trends. Translating the concept of soil productivity to forestry is troublesome, however, given the dynamics of stand development, changing management systems within a forest rotation, and the Herculean effort required to measure total vegetative growth. Equally telling, the term productivity is stigmatized in many public circles with respect to managing public lands.

Soil quality is admittedly in its infancy as a practicable and reliable indicator of the soil condition. Lacking is a scientifically-accepted set of indices that reflect disturbance-related changes in soil properties. Proposed indices encompass a combination of soil physical (e.g. water infiltration, erosion, soil strength, aeration), chemical (e.g. pH, organic matter, total C, total N, salinity, cation exchange capacity), and biological (e.g. microbial biomass, respiration, N mineralization) properties (Doran and Parkin 1994). Difficulty in selecting indices that are both insightful and affordable is a stumbling block and a common theme discussed in several review articles (Burger and Kelting 1999; Page-Dumroese and others 2000; Powers and others 1998). As Page-Dumroese and others (2000) state, "efforts to construct definitive soil quality/sustainability standards and guidelines are still in their infancy..... because of the diversity of soil properties to be measured, appraisal techniques, and soil uses.”

Regardless of any technical ambiguities, soil quality has gained strong support among land resource managers and the scientific community as a conceptual and communicative tool. If told there is a substantial decline in soil quality due to forestry practices, resource managers will likely listen (even if they don’t know exactly what it means). From there, the door is open for professional soil scientists to identify the soil properties that fail to meet forest guidelines, and to propose ameliorative treatments when needed.

Soils of Central Oregon

Central Oregon offers a beautiful landscape, starting with the high mountain peaks of the Cascades to the west, and traversing across mixed-conifer forests, to ponderosa pine (*Pinus ponderosa* P. & C. Lawson) and lodgepole pine (*Pinus contorta* Dougl. ex Loud.) forests at lower elevations, and finally to the high desert plateau to the east. The soils are diverse and reflect a geomorphically-complex landscape, which includes 5 of the 8 geomorphic provinces found in Oregon. Soils supporting ponderosa pine growth, in contrast, are fairly uniform, and are primarily derived from wind deposits of pumice and ash from the eruption of Mt. Mazama (Crater Lake). Smaller volcanic activity scattered across the plateau has added subordinate parent material diversity.

In general, the soils are poorly developed Andisols with loamy sand and sandy loam textures. The benchmark soil (Lapine series), for example, has a 6 to 10 cm (2.4 to 4 inch) surface horizon (A horizon) above a 12 to 24 cm (5 to 10 inch) transition horizon (AC) and an undeveloped C horizon (see http://www.nrcs.usda.gov for details). The depth to the buried soil (itself a poorly developed, infertile soil) is 60 to 100 cm (24 to 39 inch), and generally declines with increasing distance from Crater Lake. Clay contents are 8 percent or less throughout the profile. Other dominant soil characteristics include (1) low fertility, and (2) unusually high porosity and water-holding capacity for a sandy soil. Poor fertility is a function both of the young parent material and the semi-arid conditions on the pumice plateau (30 to 75 cm annual precipitation), which limits plant growth and, ultimately, soil organic matter input. The soils have impressively low N concentrations beneath the A horizon (Dyrness and Youngberg 1966; Youngberg and Dyrness 1963).

The Black Bark Study

The Black Bark study began in the 1980s as a collaborative project between the Deschutes National Forest and the Pacific Northwest Research Station (Bend Silviculture Laboratory) to better understand the ecology of central Oregon ponderosa pine forests. With a wide array of silvicultural treatments replicated across a productivity gradient of pine forests, the study provides an interesting insight to the resilience of the area’s soils.

The study concept was prompted by a large-scale mountain pine beetle infestation in the lodgepole pine forests south of Bend, OR. Whether the insect population would migrate into ponderosa pine stands was unclear and, consequently, of immense concern to the Deschutes National Forest. Prior evidence suggested that thinning of overstocked pine stands was the most successful means to prevent further insect damage (Mitchell and Preisler 1991). As a result, the Deschutes National Forest adopted an aggressive thinning program in its second-growth ponderosa pine stands. The targeted area included approximately 100,000 hectares (247,000 acres) of "black bark" ponderosa pine, considered at the time the most productive second-growth pine forests in central Oregon.

The management objective of the study (as conceived by Pat Cochran, retired PNW Soil Scientist, Bill Hopkins, retired Area Ecologist, and Don Peterson, retired Timber Staff Officer) was to evaluate ecological changes following partial overstory harvesting, and thus contribute to the monitoring needs of the Forest's thinning program. From a scientific standpoint, the objective was expanded greatly to evaluate soil and plant responses to manipulation of site organic matter. Sixteen treatments were selected which created a gradient of surface organic matter retention following combinations of thinning and prescribed fire (*table 1*).

Three study sites were selected along a west-to-east transect on the Bend-Fort Rock District. Swede Ridge, the most productive site, is proximal to the Cascade crest, and thus receives considerably more annual precipitation than the other sites; Sugar Cast, the middle site, is near the heart of the pumice plateau on the Lava Cast forest near Sunriver, OR; and East Fort Rock, the poorest site, is on the eastern edge of the ponderosa pine forests, within 5 km of the desert fringe. Additional site characteristics are presented in *table 2*. All treatments were replicated at each site (block) on 0.4 ha plots in a randomized complete block experimental design.

Thinning to a target basal area of 13.7 m^2 ha^{-1} (60 ft^2 ac^{-1}) was completed in 1989, and the prescribed burns were conducted in spring 1991 and repeated in spring 2002. The burns were generally low to moderate intensity (0.5 to 1.2 m (1.6 to 4 ft) average flame lengths) with 45 to 50 percent duff reduction. Details of the 1991 burns, including fuel loading, consumption, and soil heating, are presented by Shea (1993). Fertilizer (224 kg ha^{-1} N; 112 kg ha^{-1} P; 37 kg ha^{-1} S, or the equivalent of 200 lb ac^{-1} N; 100 lb ac^{-1} P; 33 lb ac^{-1} S) was applied in the fall of 1991 and again in 1996.

Table 1—*Treatment design of the Black Bark study. Treatments are ranked generally from the lowest to highest level of organic matter retention.*

Treatment rank	Thinning method	Additional treatment
1	Whole-tree harvest	Fire
2	Whole-tree harvest	Fire + fertilize
3	Whole-tree harvest	Fertilize
4	Whole-tree harvest	None
5	Bole-only harvest	Fire
6	Bole-only harvest	Fire + fertilize
7	Bole-only harvest	Fertilize
8	Bole-only harvest	None
9	Thin, leave trees on site	Fire
10	Thin, leave trees on site	Fire + fertilize
11	Thin, leave trees on site	Fertilize
12	Thin, leave trees on site	None
13	No thin	Fire
14	No thin	Fire + fertilize
15	No thin	Fertilize
16	No thin	None

Table 2—*Characteristics of the three Black Bark study sites prior to treatment in 1988.*

Site characteristic	Swede Ridge	Sugar Cast	East Fort Rock
Relative site quality	High	Medium	Low
Site index (m; Barrett 1978)	35	31	25
Age (yr)	40	49	56
Density (trees ha^{-1})	783	707	495
Basal area (m^2 ha^{-1})	39	33	28
Dominant understory vegetation	Snowbrush ceanothus, antelope bitterbrush	Antelope bitterbrush, greenleaf manzanita	Antelope bitterbrush, greenleaf manzanita
Elevation (m)	1,520	1,398	1,554
Precipitation (cm)	65	50	38
Mean July temperature (^oC)	15	18	18

Soil nutrients, wood decay, tree height and volume, diameter at breast height, and fuel loading were measured on a 5-year cycle. Herbaceous production was measured annually by clipping all vegetation within ten 0.25 m^2 (2.7 ft^2) frames in each plot. Shrub cover was determined every third year by measuring the canopy size of all plants within three 100 m^2 (1076 ft^2) belt transects per plot. Percent shrub cover was converted to a dry weight basis using site-specific biomass equations. Additional soil quality measurements (microbial biomass, respiration, surface CO_2 efflux, C utilization, litter decay, phospholipid fatty acid content) were made periodically between 1989 and 2005.

Results

No trends in chemical or biological indices of soil quality were detected in the initial 15 years following thinning. In contrast, substantial increases in tree vigor and production of wildlife browse were evident. The periodic annual increment for diameter growth increased 60 percent, while shrub cover increased more than 200 percent due to thinning (*fig. 1*). Antelope bitterbrush (*Purshia tridentata* (Pursh.) DC.; hereafter bitterbrush), in particular, responded to thinning: average cover was 7 percent on unthinned plots compared to 25 percent on thinned plots by 2002. Maximum bitterbrush cover on thinned plots reached 50 percent. Herbaceous production also increased due to thinning, although this result is tempered by the fact that biomass production was extremely small regardless of treatment (14 kg ha^{-1} for thinned plots versus 4 kg ha^{-1} for unthinned plots). Plant diversity (number of species) was low regardless of treatment, and declined slightly due to thinning. Whether thinning has improved forest health and resistance to insect attack can only be assumed, since no insect-related mortality has been identified on thinned or unthinned plots.

Differences in soil quality or vegetation growth due to the method of thinning (whole-tree harvest; bole-only harvest; thin, leave trees on site) were slight. Thus, the gradient of organic matter retention associated with pre-commercial thinning appears to have little short-term effect in these pine ecosystems. Whether this will change with time as the larger organics decompose and their products become incorporated in the soil is unclear.

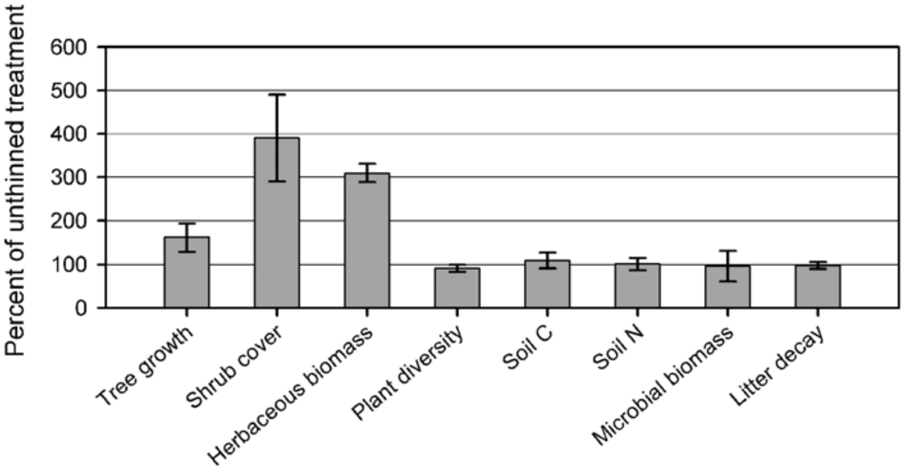

Figure 1—Thinning effects in central Oregon pine stands. Bars represent mean percentage plus standard error (n=3) of Whole-tree harvest (treatment 4) plots relative to No thin (treatment 16) controls (e.g. thinned and unthinned treatments are equivalent at 100 percent). Tree growth is the 10-year diameter increment. Soil C and N values include forest floor plus mineral soil to a depth of 60 cm.

The strongest ecosystem-level response to date has been to nutrient additions. Soil C and N, microbial biomass and activity, tree growth, and herbaceous production and diversity are among the numerous measures that increased substantially following fertilizer applications in 1991 and 1996 (*fig. 2*). Only shrub growth failed to respond positively, indicative of the poor competitive ability of shrubs relative to other vegetative lifeforms in acquiring and utilizing available nutrients. Our results not only support previous findings from central Oregon of soil nutrient limitations to tree growth (Cochran 1978; Cochran 1979; Youngberg and Dyrness 1965), but extend the results to include other important lifeforms and processes. Does this mean that the Deschutes National Forest should consider applying fertilizers in its second-growth pine forests? There is an important message from our finding, but it has little to do with the economic benefits of applying fertilizer. The results show soil-plant processes in central Oregon are not just controlled by moisture limitations; they are also nutrient limited. Forest practices that conserve (or even increase) total nutrient pools while meeting the primary ecological, social, or economic objectives should be emphasized.

The effects of prescribed fire have generally been benign to date. Strong declines in soil N content and shrub cover are the only exceptions. Soil N content was more than 20 percent lower on burn plots compared to controls as a direct result of consumption of forest floor organics (*fig. 3*). No changes in mineral soil N (total or available) were detected following either burn, nor were changes evident in mineral soil C, pH, extractable phosphorus, cation exchange capacity, C utilization or phospholipid fatty acid composition. Only an inconsistent decline in microbial biomass was found between 1991 and 2003 in the surface mineral soil of burned plots.

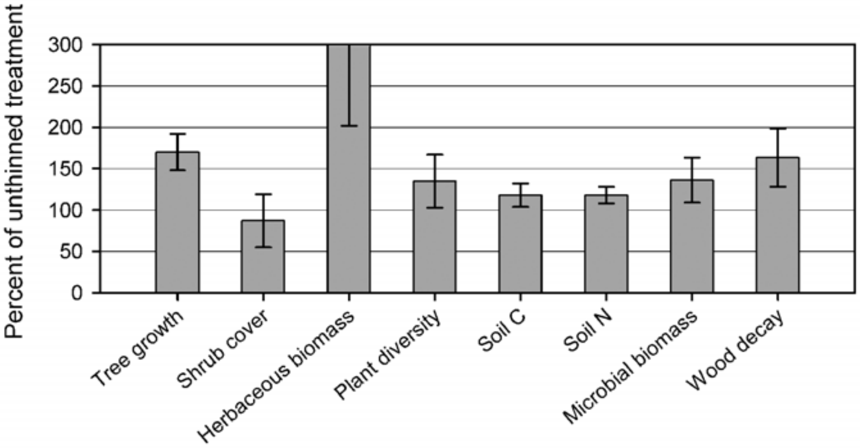

Figure 2—Nutrient limitations in central Oregon pine stands. Bars represent mean plus standard error (n=3) of fertilized plots (treatment 3) relative to unfertilized controls (treatment 4). Mean value for herbaceous biomass is 845 percent.

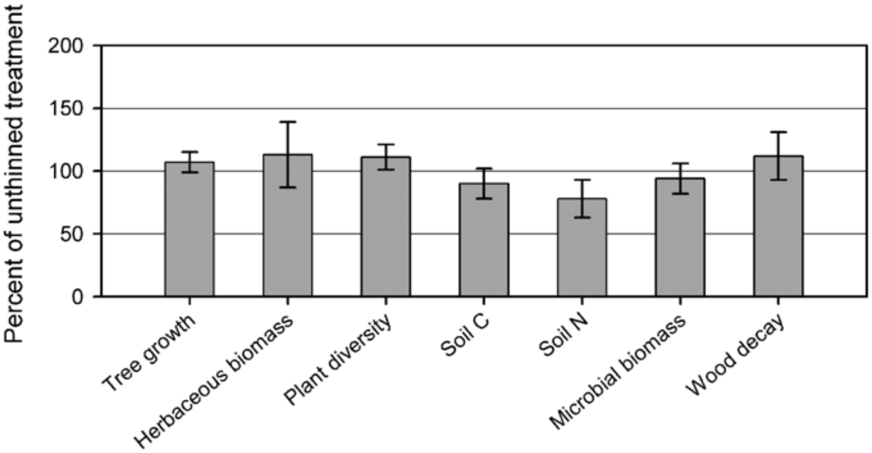

Figure 3—Fire effects in central Oregon pine stands. Bars represent mean values plus standard error (n=3) of burned plots (treatment 1) relative to unburned controls (treatment 4).

Bitterbrush cover was severely reduced after the initial burn, although post-fire seed germination led to partial recovery in the following eleven years. By 2001, bitterbrush cover on burned plots averaged 13 percent compared to 25 percent on unburned plots. After the second round of burns, however, bitterbrush was virtually eliminated from the stands, with little sign of basal stem sprouting or seed germination noted by the end of the second growing season.

Tree growth between 1991 and 2001 was slightly higher on burned plots compared to controls, which contradicts previous studies in central Oregon showing tree growth declines associated with burning (Busse and others 2000; Cochran and

Hopkins 1991; Landsberg 1993). We suspect the increase in tree growth was a result of a small pulse of available N following burning in combination with reduced understory competition for water and nutrients. It remains to be seen whether these 5 to 10-year changes in tree growth have long-term implications to stand growth.

Fire and Soil Nitrogen

Our planet and atmosphere are loaded with N. It's in the air, in the soil and ground water, and rising in exhaust emissions. Nitrogen is everywhere. So, there should be more than enough for every living organism to thrive, right? Then why is N generally considered the most limiting of all nutrients for plant growth on a global scale? And what role does fire play in altering plant-available N?

At the heart of the N conundrum is the fact that most organisms primarily assimilate inorganic N (ammonium and nitrate), which, in turn, comprises a miniscule fraction of the total N pool (Paul and Clark 1996). The great majority of soil N, for example, is found in organic complexes, and is released as inorganic forms in a slow bottleneck process by soil microorganisms. Nitrogen-fixing legumes and actinorhizal plants (e.g., snowbrush ceanothus (*Ceanothus velutinus* Dougl. ex Hook.; hereafter snowbrush) and bitterbrush) that convert atmospheric N directly into plant-available N are the main exceptions to the rule. Accentuating this problem is the fact that N is not distributed evenly in terrestrial ecosystems either at micro- or macro-scales. Dry environments, such as those found in central Oregon, are characteristically on the low end of the N distribution scale. Comparing the N content of forest ecosystems in central Oregon versus those in moist western Oregon provides a good example of this inequity. As much as three to ten times more total ecosystem N is found west of the Cascade crest than in central Oregon (*fig. 4*). Nitrogen conservation, therefore, is vital to the integrity of central Oregon forests. Any N disruptions, additions, or subtractions will have a much greater ecological effect in these poorly-buffered forests than in N "rich" forests west of the Cascade crest. As an example, a loss of 350 kg N ha^{-1} would result in a 15 percent reduction of total N in central Oregon, but only a 3 percent reduction of site N in western Oregon.

Fire plays a major role in determining total and available N in forests of the interior west by consuming organic matter, altering rates of soil N processes, and modifying post-fire plant community composition and growth. Total ecosystem N is reduced during burning primarily as a result of volatilization losses from forest floor material. Temperatures in the mineral soil, in contrast, rarely exceed the critical range for N volatilization of 300 to 500^{o}C (Hungerford and others 1991), and thus N loss from the mineral fraction is generally of little concern. The absolute quantity of N lost during burning is difficult to predict, and depends on factors such as pre-burn fuel loading, fuel moisture, fuel continuity, and fire severity. Estimates of N loss range from 20 to 400 kg ha^{-1} in central Oregon studies (*table 3*). On the positive side, fire typically results in an abundant release of plant-available N, as volatilization losses to the atmosphere are usually incomplete. This often-reported flush of inorganic N (Covington and Sackett 1986, 1992) can result in short-term increases in plant growth. Therefore, fire wields a double-edged sword: providing a short-term flush of plant-available N, while reducing the total ecosystem N pool.

An impressive quantity of forest floor N was lost during repeated burning in thinned and unthinned plots on the Black Bark study: the combined loss for the 1991 and 2002 burns was more than 500 kg ha^{-1} (*table 4*). Unfortunately, the post-fire flush of inorganic N was not quantified, although circumstantial evidence suggests that the amount released was inconsequential. Specifically, measurements taken 18 months after burning showed no differences in available soil N between burned and unburned plots. Also, herbaceous production did not increase following fire as might be expected with a flush of N, as was seen on fertilized plots.

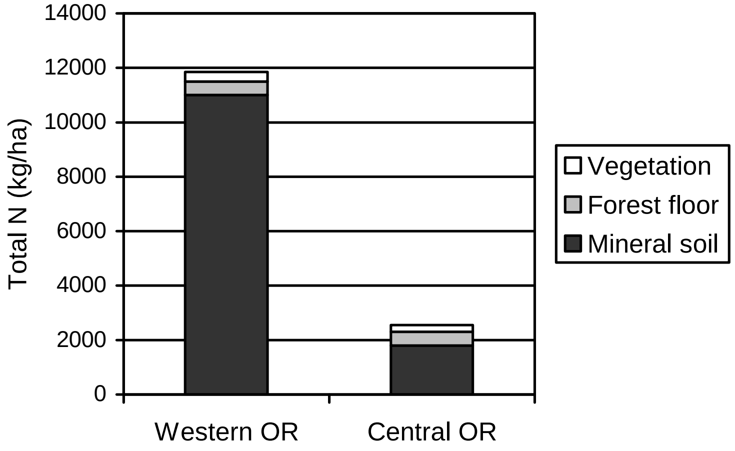

Figure 4—Comparison of total ecosystem N in a mesic-climate forest of western Oregon and a semi-arid ponderosa pine forest in central Oregon. Data for western Oregon is from R. Zasoski[1] and for central Oregon is from the Black Bark study[2].

Table 3—*Estimated N losses from prescribed fire studies conducted in central Oregon ponderosa pine stands.*

Source	**Fire intensity**	**N loss (kg ha^{-1})**
Landsberg (1993)	Severe	280
Shea (1993)	Moderate (1991)	366 to 397
Black Bark study	Moderate (2002)	126 to 183
Simon[1]	Low to moderate	100
Nissley and others (1980)	Low	20 to 260

[1] Data on file, Deschutes National Forest, Bend, OR

Table 4—*Forest floor nitrogen (kg ha^{-1}) before and after repeated prescribed fires in second-growth ponderosa pine stands in central Oregon. N=6 in 1991, n=3 in 2002.*

	Initial burns in 1991[1]			Repeated burns in 2002			
Treatment	Preburn	Postburn	N loss	Preburn	Postburn	N loss	Total N loss
Thinned	756	359	397	287	161	126	523
Unthinned	764	398	366	301	118	183	549

[1] Data for 1991 burns are from Shea (1994).

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[1] Personal communication, data on file, Deschutes National Forest, Bend, OR

[2] Data on file, Deschutes National Forest, Bend, OR

Table 5—*Estimated N losses due to silvicultural practices in central Oregon pine stands from the Black Bark study. Total ecosystem N includes all vegetation, downed woody material, forest floor, and mineral soil through the C1 horizon, and averaged 2467 kg ha^{-1} for the three study sites (Little and Shainsky 1995).*

Stand treatment	Estimated N loss (kg ha^{-1})	Percentage of ecosystem N
Prescribed fire (1991+2002)		
Thinned	523	21
Unthinned	549	22
Pre-commercial thinning		
Bole-only	36	1
Whole tree	94	4

Just how important is the loss of 500 kg N ha^{-1} to the sustainability of these pine forests? Certainly no detrimental effects to vegetation growth have been expressed so far in the study, indicative of a nutritionally resilient system. Forest productivity, therefore, has not declined. On the other hand, 500 kg N ha^{-1} represents a loss of nearly one-fourth of the total ecosystem N pool (*table 5*), indicating that soil quality was compromised. As a comparison of alternative methods to reduce fire risk, pre-commercial thinning had a relatively innocuous effect on ecosystem N, while mowing of shrubs, although not measured, would have minimal affect on total N. Interestingly, the practice of slash retention (bole-only harvest) as a means to conserve site organic matter resulted in little improvement in site N compared to whole-tree harvest.

Taking the concept of repeated fire in eastside pine forests one step further, Johnson and others (1998) estimated total N losses during a 100-year period for a range of fire-return intervals (1, 3, 5, 10, or 20 burns in 100 years). Assuming that each fire consumed 50 percent of the forest floor N, and given the natural rate of litter input and decay for their site, they found that burning every five years resulted in a loss of nearly 1500 kg ha^{-1} of N (*fig. 5*). Burning every 10 years resulted in a 50 percent reduction in total N by the end of 100 years. Admittedly their model is simplistic and does not account for changes in stand structure, litterfall, or burn prescriptions during the 100-year rotation. Nevertheless, the potential loss of soil quality is daunting.

This exercise in predicting N losses during 100 years brings up an interesting question. If repeated burning results in near-catastrophic losses of soil N, and if the pine forests in central Oregon burned on a 4 to 24 year interval prior to the era of fire suppression (Bork 1984), then why didn't the forests run completely out of N a few centuries ago? Several possible explanations come to mind: (1) the forests actually ran out of N; (2) N input from atmospheric deposition and N-fixing plants was sufficient to offset N losses; (3) the model of Johnson and others (1998) and the results from the Black Bark study over-predict N loss in forests prior to fire suppression. Of course, explanation (1) is nonsensical, and explanation (2) is highly unlikely since fewer N-fixing shrubs were present in the understory of ponderosa pine forests prior to the 20th century (Cochran and Hopkins 1990). The third explanation is intuitively acceptable (although unproven) if we acknowledge that the earlier forests had open, park-like structure with less litterfall and forest floor accumulation, which led to lighter burns and, consequently, less N volatilized per burn.

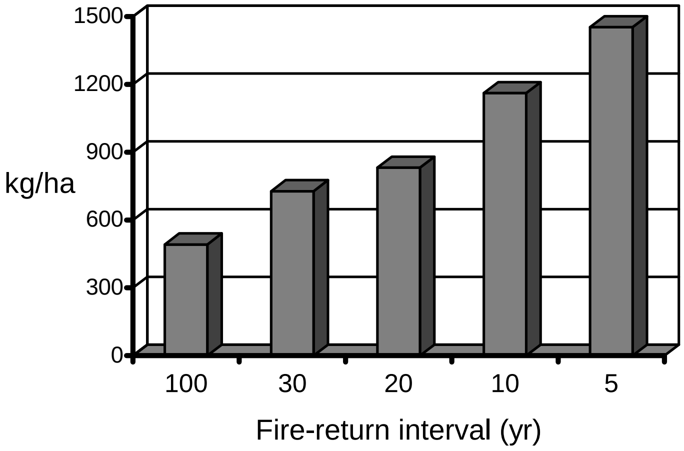

Figure 5—Predicted N losses during a 100-year rotation due to repeated prescribed burning at differing fire intervals. Data are from Johnson and others (1998) for stands of Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.).

Can N Be Replaced By Natural Processes?

Maybe so. Nitrogen fixation–the conversion of atmospheric N into biologically-available forms via atmospheric deposition, lightning, symbiotic N fixation, and non-symbiotic N fixation–can partially or completely counter N losses due to fire. Symbiotic N fixation, in particular, is an energy efficient and productive means of replacing N losses.

Bitterbrush and ceanothus are the most common N-fixing plants in central Oregon. Bitterbrush is found on mid- to low-elevation pine sites, while snowbrush is found at comparatively higher elevations. N-fixation rates range from 0.5 to 2 kg ha^{-1} yr^{-1} for bitterbrush and from 4 to 16 kg ha^{-1} yr^{-1} for snowbrush in central Oregon (Busse 2000). These are not large quantities of fixed N, like is common in pure stands of agricultural legumes. Thus, replacing N losses after fire is a long-term venture (*table 6*), and, in the case of bitterbrush, will never be complete. Snowbrush, by comparison, can replenish the total N pool in 25 years following a single prescribed fire.

Table 6—*Estimated time require to replace N losses (kg $ha^{-1)}$ from prescribed burning.*

		Years required to replenish lost N[3]	
Fire	N loss	Snowbrush	Bitterbrush
Single	280[1]	25	140
Repeated	523[2]	48	262

[1] From Landsberg (1993)

[2] Black Bark study, data on file, Deschutes National Forest, Bend, OR

[3] Nitrogen-fixation rates, based on the results of Busse (2000), are 10 kg ha^{-1} yr^{-1} for snowbrush; 1 kg ha^{-1} yr^{-1} for bitterbrush; and 1 kg ha^{-1} yr^{-1} for atmospheric deposition

Management Considerations

Let's face it, soils are boring to the masses. If soil was only impervious to disturbance, we wouldn't have to give it much thought. Then we would accept Kimmins (1994) statement "it is neither possible nor is it necessary to consider all of the processes and components in order to make useful predictions about the long-term consequences of forest management" as appropriate for forest soils. Alas, the misfortune of past civilizations that failed to respect their soils or those exposed to catastrophic events leaves a lasting imprint of the need to revere soil.

Obviously, the results from the Black Bark study in central Oregon pine forests paint a far less stunning picture of soil devastation compared to images from the Dust Bowl or from hydraulic mining in the Sierra Nevada mountains. Most soil characteristics and processes, in fact, were unaffected by combinations of thinning and fire. Repeated burning, however, resulted in a loss of soil quality due to reduced total N. Continued burning on the plots may only accentuate this problem.

We offer the following suggestions to avoid similar fire-induced losses in soil quality in central Oregon ponderosa pine forests:

- Prescribe low severity burns, particularly in stands lacking previous prescribed fire entry. Prescriptions that leave 75 percent or more of the duff layer intact will help conserve N.
- Reduce fire frequency. Soil N loss increases with increasing fire frequency.
- Incorporate mechanical treatments (thinning and mowing) to reduce fire risk. Little N is removed by these methods compared to burning.
- Value N-fixing understory plants. Nitrogen can be restored on burned sites, particularly at higher elevations where snowbrush is present.

Forest management does not revolve around protecting soil N pools. Instead, this is the era of wildfire risk reduction, threatened and endangered species protection, wildlife habitat restoration, and recreation use. Long-term changes in soil quality, however, need not occur in central Oregon if simple precautions are taken such as conserving soil N. Most importantly, they will be avoided if all that work (and play) in the forests speak for the soil.

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Management of Ponderosa Pine Nutrition Through Fertilization[1]

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Abstract

The results of a series of replicated fertilization trials established throughout the Inland Northwest were reviewed for information specific to ponderosa pine (*Pinus ponderosa* P. and C. Lawson) nutrition. Ponderosa pine nitrogen (N) status was often better than the N-status of other Inland Northwest species, and therefore growth response to N fertilization was often lower than that of other species. Fertilization of ponderosa pine with N alone sometimes appeared to cause increased tree susceptibility to mortality by insect, disease and perhaps abiotic stresses. Growth and mortality response to N fertilization appeared to be related to foliage potassium (K)/N ratio in some cases. The application of K and micronutrients in combination with N may have protected the trees from N-related mortality while stimulating a growth response. Sulfur fertilization was not found to evoke a growth response in ponderosa pine, and may have increased mortality rates slightly. On certain rock types and vegetation series, ponderosa pine showed high growth response to macronutrient plus micronutrient fertilization as well as herbicide treatment. Ponderosa pine generally did not show a strong growth response to N fertilization, except on 'good' rock types on moist sites. Multinutrient (macro- plus micronutrient) fertilization combined with an herbicide treatment often provided a better response than N alone on moderate to dry sites and/or 'bad' rock types. Other species in mixed-conifer stands, particularly grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.), often showed a better growth response to fertilization than ponderosa pine. The nutritional ecology of ponderosa pine is unique among Inland forest tree species and should be considered when evaluating nutrient management options.

Introduction

Since its inception in 1980, the Intermountain Forest Tree Nutrition Cooperative (IFTNC) has established numerous studies of forest tree response to various fertilization treatments throughout the inland portion of the northwestern United States (*table 1*). The initial studies, established in Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco) and ponderosa pine (*Pinus ponderosa* P. and C. Lawson) stands, tested various rates of nitrogen (N) fertilizer, while subsequent studies incorporated potassium (K). During the establishment of the Forest Health and Nutrition study in the mid-1990's, rock type was implicated in plantation success and stand health, leading to the establishment of a region-wide Seedling Establishment study (Moore and Mika 1997, Garrison and others

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1997, Garrison and Shaw 1998). A series of individual tree screening trials in ponderosa pine and other tree species tested growth response to various combinations of fertilizer and herbicide[4]. More recently, unfertilized foliage nutrient concentrations of four tree species, including ponderosa pine, were compiled to identify suggested sample sizes for nutrient testing, as well as to identify natural ranges of nutrient variability (Moore and others 2004). The objective of this paper is to summarize the implications of these studies for ponderosa pine nutrition. Changes in nutrient concentration and growth rate after fertilization or herbicide application will be addressed in detail.

Tree nutrition may be assessed by comparing foliage nutrient concentrations to published critical levels (*table 2*). Average concentrations of both unfertilized and fertilized ponderosa pine foliage from several IFTNC studies were measured one year after treatment (*table 3*). If the measured foliage concentration was below the critical level for ponderosa pine, the trees were considered deficient in that element, and the element was considered growth limiting. A yield response to fertilization with the deficient element was therefore expected. If a nutrient concentration was at or above the critical level, the trees were considered to have a sufficient quantity of that element for growth, and the element was considered non-limiting; thus, no yield response was expected.

Critical levels are based on traditional yield curves, and therefore provide a direct, species-specific and biologically significant technique for assessing the effects of nutrient amendments on tree growth and yield. A related method of nutrient assessment involves the use of optimum nutrient proportions (nutrient ratios). This technique is based on the hypothesis that for a given amount of one element, usually N, a certain proportion of other elements is required to maintain an optimum nutrient balance within the plant (Powers 1983, Blake and others 1990, Ingestad 1971). The optimum concentration of other elements therefore varies based on the N concentration. Several of the critical levels proposed in *table 2* were derived using optimum ratios in conjunction with the established critical N level for ponderosa pine. Because nutrient ratios are not tied to a yield expectation, they provide a useful assessment of internal plant nutritional balances, and are suited to assessment of forest stands for general nutritional status even in the absence of fertilization. Because one of the primary research directives of the IFTNC is to increase forest yield through improved tree nutrition, the critical level method is more commonly referenced throughout this paper.

Several terms used throughout this paper require some additional explanation. The terms ‘rock type’, ‘underlying geology’ and similar references to the dominant geology underlying forest stands is used intentionally, rather than the more common terms ‘parent material’ or ‘soil type.’ Forested sites are often underlain by several parent materials, including bedrock and one or more surficial deposits. The underlying geologic formation often dominates soil properties, and has been a useful variable in explaining forest nutrition and fertilization response during IFTNC research. Therefore, references to the underlying geologic formation are used as such throughout this paper. The term ‘vegetation series’ refers to a grouping of habitat types or plant associations named after a tree species that is expected to become

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[4] Unpublished data on file, Intermountain Forest Tree Nutrition Cooperative, University of Idaho, Moscow, Idaho

Table 1—*List of selected fertilization and herbicide studies established by the Intermountain Forest Tree Nutrition Cooperative between 1980 and 2000. Regions refer to northeastern Oregon (NE OR), central Washington (C WA), northeastern Washington (NE WA), central Idaho (C ID), northern Idaho (N ID) and western Montana (W MT).*

Trial name	**No. of sites**	**Year(s) estab-lished**	**Stand composition**	**Region**	**Treatments and rates[1]**
Douglas-fir trials	90	1980-1982	Douglas-fir	C ID, C WA, NE WA, NE OR, N ID, W MT	control; 224 kg N; 448 kg N
Ponderosa pine	10	1985	Ponderosa pine	NE OR, C WA	control; 224 kg N; 448 kg N
Montana ponderosa pine	6	1987	Ponderosa pine	W MT	control; 224 kg N; 224 kg N + 190 kg K
Umatilla mixed conifer	8	1991	Mixed conifer	NE OR	control; 224 kg N; 224 kg N + 112 kg S
Okanogan mixed conifer	8	1993	Mixed conifer	C WA	control; 224 kg N; 224 kg N + 190 kg K
Forest health and nutrition	31	1994-1996	Mixed conifer	C ID, C WA, NE WA, NE OR, N ID	1994 (12 sites): control; 336 kg N; 190 kg K; 336 kg N + 190 kg K 1995 (12 sites): control; 336 kg N; 190 kg K; 336 kg N + 190 kg K; 336 kg N + 190 kg K + 112 kg S 1996 (7 sites): control; 336 kg N; 190 kg K; 336 kg N + 190 kg K; 336 kg N + 190 kg K + 112 kg S + 5.6 kg B + 11.2 kg Cu +11.2 kg Zn + 0.1 kg Mo All years: Additional N+K combinations ranging from 0 kg N and 0 kg K to 672 kg N and 574 kg K, per experimental design
Seedling establishment	12	1998	Douglas-fir and ponderosa pine	C ID, C WA, NE WA, NE OR, N ID	control; 16 g N; 16 g N + 4.8 g S; 16 g N + 12 g K; 16 g N + 12 g K+ 4.8 g S; 16 g N + 12 g K + 4.8 g S + 4.1 g P + 0.61 g Mg + 0.01 g B + 0.03 g Cu + 0.26 g Fe + 0.04 g Mn + 0.01 g Mo
Screening trials	29	1999-2000	Ponderosa pine	C ID, C WA, NE WA, NE OR	control; 224 kg N; 224 kg N + 190 kg K + 101 kg S + 1.1 kg Mg + 1.1 kg Cu + 3.4 kg B + 1.1 kg Zn + 3.4 kg Fe (multinutrient); 3.4 kg hexazinone; multinutrient + 3.4 g hexazinone

[1] *Rates in kg ha^{-1} as broadcast application, except for seedling establishment study for which the rate is g seedling^{-1} as a subsurface (dibble) application.*

dominant under late successional conditions (Cooper and others 1991, Lillybridge and others 1995).Vegetation series are used as a proxy for site moisture regimes by the IFTNC, with the more common series in the IFTNC study region ranging from dry to moist in the order of ponderosa pine, Douglas-fir, grand fir, western red cedar (*Thuja plicata* Donn ex. D. Don) and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.). The habitat types or plant associations comprising each vegetation series are based on understory vegetation and also show a gradual progression from dry to moist. The term 'plant association' is more commonly used in Oregon and Washington, while the term 'habitat type' is more commonly used in Idaho and Montana. Both terms are used on occasion in this paper to refer to relative moisture regimes within a vegetation series.

Results from eight replicated studies were reviewed for this paper. Analysis of variance was used to detect treatment effects, and results were considered significant at or below a level of p=0.10.

Table 2—*Critical nutrient concentrations for current, upper-crown ponderosa pine foliage in the northwestern United States.*

Nutrient	Concentration[1]
Nitrogen (N; pct)	1.10[3]
Phosphorus (P; pct)	0.08[3]
Potassium (K; pct)	0.48[2]
Sulfur (S; pct)	0.08
Calcium (Ca; pct)	0.05[2]
Magnesium (Mg; pct)	0.05[2]
Manganese (Mn; ppm)	60[3]
Iron (Fe; ppm)	50[3]
Zinc (Zn; ppm)	30[3]
Copper (Cu; ppm)	3[3]
Boron (B; ppm)	20[3]

[1] Value for N from Powers and others (1985), values for P, K, Ca and Mg from Powers (1983) and Powers and others (1985). Critical S value derived for this paper using an N/S ratio of 14.7 with the critical N value (Turner and Lambert 1979, Blake and others 1990). Micronutrients from Boyer (1984, personal communication).

[2] Derived by cited author(s) using optimal proportions

[3] Derived by cited author(s) experimentally

Nutrition of mature, second-growth stands

Nitrogen and potassium

Nitrogen is probably the most common fertilizer element utilized in forest management applications in the western United States (Moore and others 1991, Mitchell and others 1996, Peterson and others 1984, Shumway and Chappell 1995, Tiedemann and others 1998, Chappell and others 1999, Carter and others 1998). In the Inland Northwest, N has traditionally been applied towards the end of the rotation, when the cost of application can be recouped through increased growth during the final few years. Fertilization with N typically results in increased foliage N concentrations the first year after fertilization (*table 3*), with concentrations in ponderosa pine foliage collected one year after fertilization typically well-correlated with N application rate. For example, increasing rate of N fertilization during the Forest Health and Nutrition Study resulted in an almost linear increase in foliage N concentrations of about 60 percent over the unfertilized control trees at 600 lb ac^{-1} N, with the K fertilizer having no notable effect on the foliage N response (*fig. 1*).

Table 3—Average nutrient concentrations of current-year, upper-crown ponderosa pine foliage measured one year after application of various fertilization treatments for several Intermountain Forest Tree Nutrition Cooperative studies.

Average foliage N concentrations (pct)

Treatment Study	Control	N[1]	N+K	N+S	N+K+S	Multi-nutrient[2]
Montana ponderosa pine	1.24	1.46	1.48			
Umatilla/Okanogan mixed conifer	1.27	1.63	1.56	1.44		
Forest health and nutrition	1.15	1.57	1.71		1.65	1.66
Seedling establishment	1.44	1.65		1.61	1.71	1.73

Average foliage K concentrations (pct)

Treatment Study	Control	N[1]	N+K	N+S	N+K+S	Multi-nutrient[2]
Montana ponderosa pine	0.75	0.72	0.79			
Umatilla/Okanogan mixed conifer	0.81	0.74	0.79	0.81		
Forest health and nutrition	0.89	0.90	0.89		0.99	0.92
Seedling establishment	0.65	0.56		0.64	0.62	0.64

Average foliage S concentrations (pct)

Treatment Study	Control	N[1]	N+K	N+S	N+K+S	Multi-nutrient[2]
Umatilla/Okanogan mixed conifer	0.06	0.06		0.06		
Forest health and nutrition	0.09	0.07	0.07		0.08	0.08
Seedling establishment	0.10	0.10		0.12	0.12	0.12

[1] *For treatment rates see table 1*

[2] *Multinutrient fertilization consisted of some combination of macronutrient and micronutrient fertilization and varied depending on the study (table 1).*

Nitrogen leads to increased tree growth by building tree foliage, which in turns leads to increased rates of photosynthesis and growth response (Miller 1981, Cole and Gessel 1992, Ballard and Carter 1985). An increase in ponderosa pine needle weight typically accompanies increased N-application rate. For example, needle weights in the same study also increased with increasing N fertilization rates, leveling off at around 400 lb N ac^{-1} with an increase of over 2 g 100 needles^{-1} over the unfertilized needles (*fig. 2*). Potassium fertilization resulted in somewhat decreased needle sizes at application rates greater than 200 lb K ac^{-1} when applied in the absence of N fertilizer.

While IFTNC studies have shown that foliage N concentrations increase after N fertilization in ponderosa pine, these studies have also shown that (1) ponderosa pine generally has higher foliage N concentrations than other common Inland Northwest forest species, and (2) other species are more often N-deficient and generally show greater increases in N concentration and needle weight following N fertilization. The

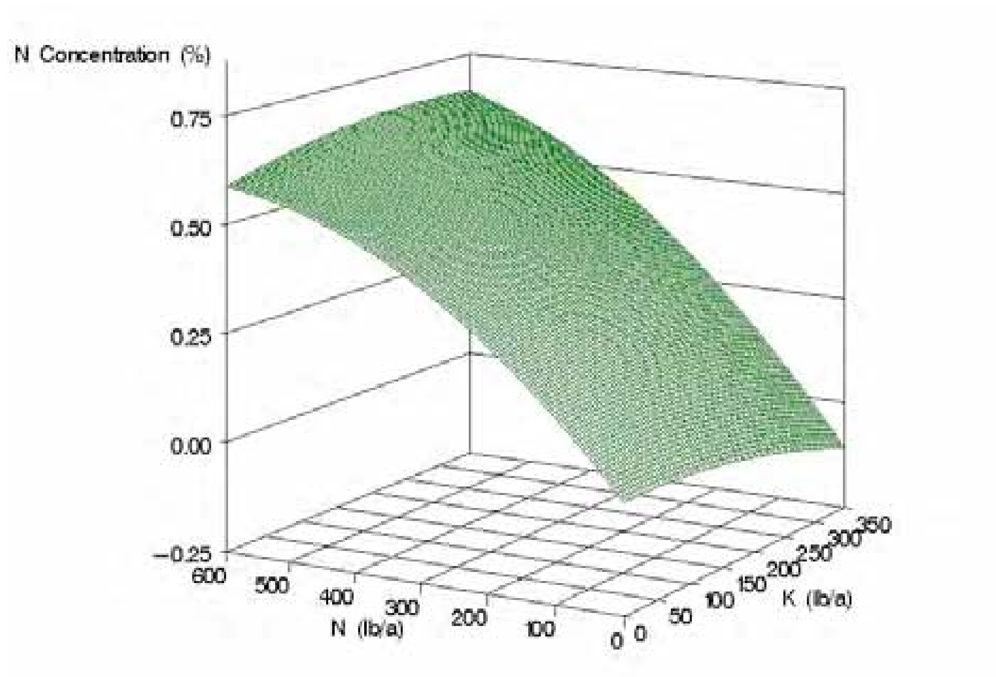

Figure 1—Percent difference between fertilized and unfertilized ponderosa pine foliage nitrogen (N) concentration one year after application of nitrogen (N) and potassium (K) fertilizer during the Forest Health and Nutrition study, for all rock types and vegetation series combined.

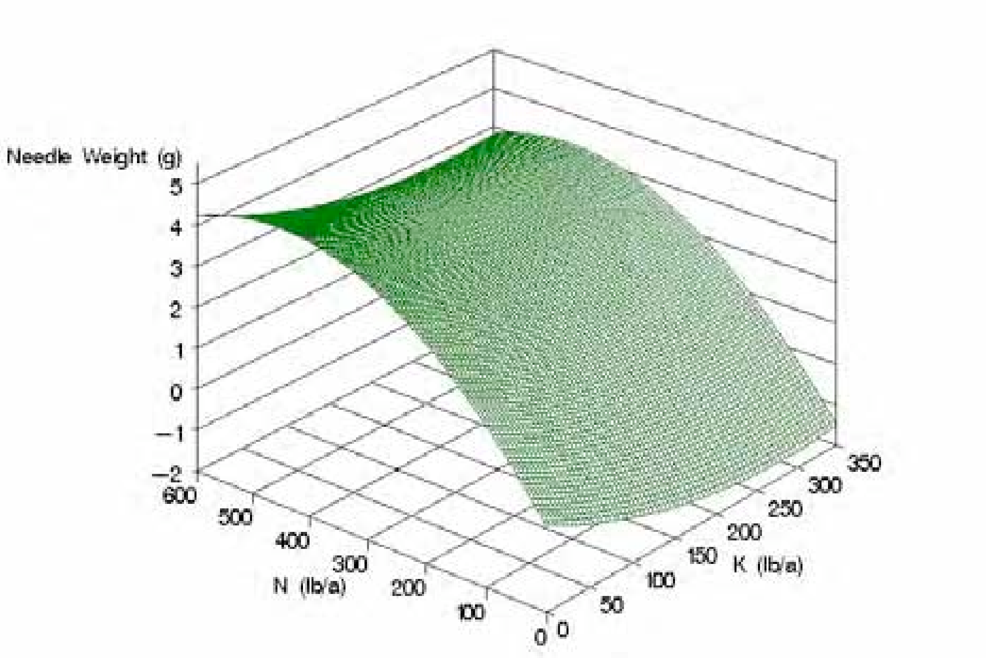

Figure 2—Difference between fertilized and unfertilized ponderosa pine needle weight (g 100 fascicles^{-1}) one year after application of nitrogen (N) and potassium (K) fertilizer during the Intermountain Forest Tree Nutrition Cooperative's Forest Health and Nutrition study, for all rock types and vegetation series combined.

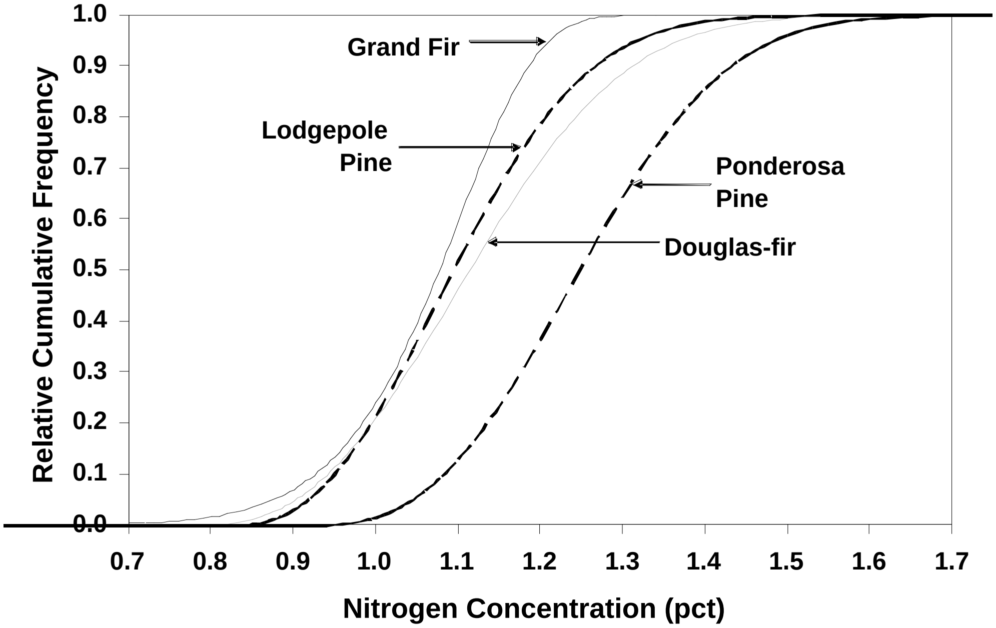

Figure 3—Cumulative frequency distribution of nitrogen (N) concentrations in unfertilized ponderosa pine, lodgepole pine, Douglas-fir and grand fir trees in the Inland Northwest. From Moore and others (2004).

first point was illustrated graphically in frequency distributions of foliage N in unfertilized ponderosa pine trees from 37 sites, Douglas-fir from 130 sites, grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.) from 14 sites and lodgepole pine (*Pinus contorta* Dougl. ex. Loud.) from nine sites across the Inland Northwest (*fig. 3*) (Moore and others 2004). The vertical axis of this graph is the proportion of all sites with foliage concentrations less than or equal to a particular value on the horizontal axis. Ponderosa pine foliage N concentrations were significantly higher than the other three species, which were all similar to each other. Critical nutrient concentrations were also evaluated using this distribution. For example, trees on about 97 percent of the Douglas-fir sites were below the critical foliage N concentration of 1.40 percent.

Most of the lodgepole and grand fir trees were also below their respective critical levels (1.20 and 1.15 percent, respectively). However, only about 15 percent of ponderosa pine N concentrations were below the critical level of 1.10 percent (*table 2*). Furthermore, foliage N response of ponderosa pine was usually of lower magnitude and showed more variation than Douglas-fir, grand fir or lodgepole pine on the same or similar sites. Correspondingly, volume growth response to N fertilization was typically lower for ponderosa pine than other species (Garrison and others 2000, Moore and others 1998). These results suggest that the critical level of 1.10 percent for N in ponderosa pine foliage may be somewhat low. A foliage N concentration of 1.20 percent (approximately the 35th percentile in the foliage N distribution) may be a more reasonable critical level for Inland Northwest ponderosa pine, based on positive four-year growth responses to 224 kg ha^{-1} N fertilization of ponderosa pine with N concentrations ranging from 1.18 to 1.24 percent (IFTNC 1992a, IFTNC 1992b). Positive growth responses of ponderosa pine to 448 kg ha^{-1} N fertilizer have been obtained with foliage N concentrations as high as 1.47 percent.

Conventional wisdom has generally held that growth response of ponderosa pine and other forest species will increase linearly with increased rates of N application. Studies by the IFTNC have shown that under ideal growth conditions, this is often the case (Mika and VanderPloeg 1991, Moore and others 1994). However, growth response of ponderosa pine to N fertilization appears to depend on several factors, including moisture regime, underlying rock type, and the availability of other nutrients on the site. These relationships were first illustrated during the initial ponderosa pine study established by the IFTNC in 1985, comprised of a series of N rate trials located in ten ponderosa pine stands in northeastern Oregon and central Washington. The stands were predominantly mature, second-growth, managed stands, and N was applied at the rate of 0, 224 and 448 kg ha^{-1} as urea. Growth response was measured as the difference between treated and control plots at each site, and results were adjusted to a common initial basal area. Growth response was expected to increase proportionately with increased N rate. However, the six-year results of this study showed that gross basal area response after adding 448 kg ha^{-1} was not different than after adding 224 kg ha^{-1}. Both responses were greater than the untreated controls (IFTNC 1992a, IFTNC 1992b). Net basal area response for the same time period was insignificant for both fertilization treatments. The difference between the gross and net results was due to mortality. Mortality was greater on N-fertilized plots than on the control plots during the first four years, particularly for the 448 kg treatment on the northeast Oregon sites, though mortality diminished during subsequent measurement periods (IFTNC 1992b). Growth and mortality were related to vegetation series, with lower growth and higher mortality occurring on the relatively drier Douglas-fir series compared to the grand fir series. Growth and mortality were also related to parent material, with lower growth and higher mortality occurring on basaltic sites, and higher growth response with less mortality occurring on sandstones. Also of interest during this study was the finding that, based on two-year periodic response analysis during the first six years, growth rates were the same for all three two-year periods, suggesting that response to fertilization was not declining and continued to be positive six years after fertilization.

Potassium was implicated in tree mortality following the 1985 ponderosa pine trials and an earlier series of Douglas-fir trials testing the same N treatments. Potassium status on all sites was assessed using a foliage K/N ratio (Ingestad 1971). Foliage K/N ratio of 0.50 is considered critical, and a ratio of 0.65 or higher is considered sufficient. The nutrient ratio method is a useful means of assessing K status, as K does not typically show a significant change in foliage concentration following fertilizer application (*table 3*). Foliage K/N ratios were examined in unfertilized trees on the IFTNC ponderosa pine and Douglas-fir trial sites. Those sites with unfertilized foliage K/N ratios greater than 0.65 appeared to have less mortality and greater growth response to the 448 kg ha^{-1} N treatment. In contrast, those sites with unfertilized foliage K/N ratios less than 0.50 experienced high mortality, particularly in response to the 448 kg N treatment. Thus, as foliage K status appeared likely related to mortality rates after N fertilization, K was incorporated into most subsequent IFTNC studies. An N+K combination treatment was first included in a study established in six ponderosa pine stands in western Montana in 1987. On these sites, the 448 kg treatment was dropped, and a treatment consisting of 224 kg ha^{-1} N plus 190 kg ha^{-1} K was substituted. This same series of treatments (unfertilized control, 224 kg N and 224 kg N+190 kg K ha^{-1}) was applied in eight mixed conifer stands on the Okanogan National Forest in 1993. A series of N+K treatments in a response surface design was also applied to 31 mixed conifer stands during the Forest Health and Nutrition study in the mid-1990's (*table 1*).

Foliage K concentrations in treated plots one year after fertilization did not show significant differences from control plots during any of these studies (*table 3*).

Potassium concentrations occasionally decreased slightly following N-only fertilization, due to an effect known as nutrient dilution. Nutrient dilution refers to a decrease in nutrient concentration due to an increase in foliage biomass that is not matched by accelerated uptake of a nutrient (Jarrel and Beverly 1981). In most cases, the combined N+K treatment restored foliage K concentrations to unfertilized levels, suggesting uptake of applied K.

Interestingly, foliage K levels occasionally appeared to increase as a result of S fertilization, as shown during a study established in 1991 in mixed conifer stands on the Umatilla National Forest (*tables 1 and 3*). This study did not include K fertilization, but did include an N+S combination, with the S and some N applied as ammonium sulfate, and the remainder of the N as urea. Foliage K concentrations were above critical levels for all treatments, with no significant differences among the controls and any treatment. Examination of total K content (K concentration multiplied by foliage biomass of 100 fascicles) indicated that the N+S treatment produced greater total K content than the controls (Garrison and others 2000). This K response to N+S fertilization may be explained in part by the chemical properties of ammonium sulfate, particularly when applied to soils high in clay such as those derived from basalts. The influx of NH_4^+ ions from ammonium-based fertilizers has been shown to compete with K^+ ions for sites on the soil exchange complex (Liu and others 1997, Chen and Mackenzie 1992). In our study, this appears to have resulted in an increase in exchangeable K available for plant uptake.

The 10-year results from the Montana ponderosa pine study supported the hypothesis that N+K application would decrease tree mortality compared to N-only fertilization (IFTNC 1998). While neither fertilization treatment increased net basal area growth over control plot growth, the N+K treatment did increase gross basal area growth. The difference between gross and net responses was due to mortality, with the N+K treatment showing positive growth response (10.4 percent gross basal area response) and low mortality (3.1 percent of gross basal area), and the N treatment showing low growth response (1.9 percent gross basal area response) and high mortality (7.2 percent of gross basal area) compared to the control plots (1.1 percent mortality; *fig. 4*). In other words, N+K fertilization appeared to protect the trees from mortality while allowing them to respond to N fertilization. Notably, most of the mortality during this study was caused by mountain pine beetle. The IFTNC hypothesized that beetles were responding to some physiological or chemical differences in the trees that were fertilized with N alone (Mika and others 1993). Alternatively, K provided in the N+K treatment may have enabled some protective mechanisms in those trees, allowing them to withstand beetle attack. Similar results occurred during the Douglas-fir trials established in the early 1980’s, though the mortality agents were different (Mika and Moore 1991).

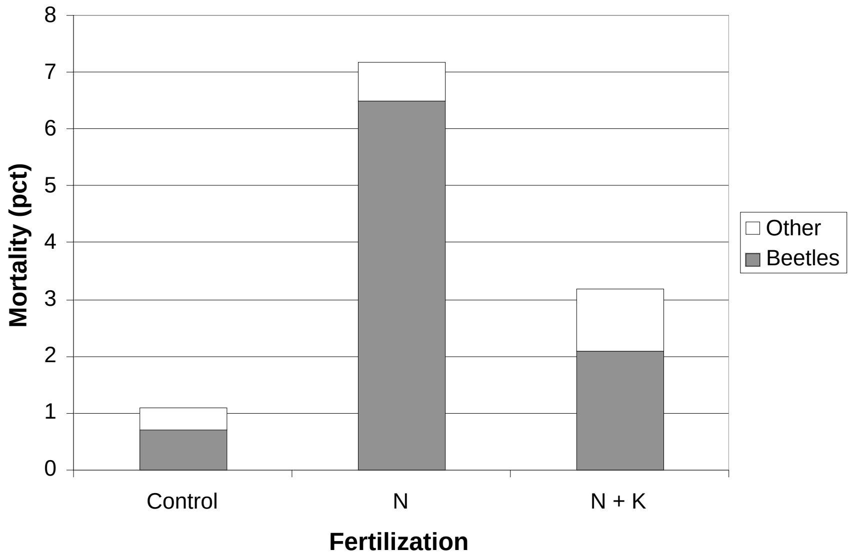

Figure 4— Mortality response and cause of mortality of ponderosa pine ten years after 224 kg ha^{-1} nitrogen (N) and 224 kg ha^{-1} nitrogen plus 190 kg ha^{-1} potassium (N+K) fertilization during the Intermountain Forest Tree Nutrition Cooperative's Montana ponderosa pine fertilization trials.

Relative volume growth response (four-year volume growth on fertilized plots relative to unfertilized plots) for ponderosa pine in the Okanogan and Umatilla mixed conifer studies was marginally greater for the N+K treatment compared to the controls (Garrison and others 2000). None of the other treatments applied during those studies (N or N+S) differed from each other or from the controls. These results were similar to those of the Montana ponderosa pine study. However, six-year basal area growth response during the Forest Health and Nutrition study confirmed the expected growth increase with increasing N rate[5]. Basal area increased by as much as 30 percent at the higher rates of N application and decreased somewhat with increasing K application to a low of almost 20 percent lower lasal area growth on the fertilized compared to control trees at the 0 lb N ac^{-1} rate (*fig. 5*). As suggested by the results of the earlier fertilization trials, variation in N response was likely related to moisture regime and perhaps rock type. The Montana and Okanogan studies were installed on relatively dry sites supporting Douglas-fir series and some drier habitat types and plant associations within the grand fir series. In contrast, the Forest Health and Nutrition study incorporated moister habitat types and plant associations within the grand fir series, as well as sites in the western red cedar and western hemlock series. While several of the Montana sites were on nutritionally poor metasedimentary rocks, the Okanogan and many of the Forest Health and Nutrition sites were on nutritionally better granitic and basaltic rock types. These factors likely interact, with higher response by ponderosa pine to N fertilization occurring on moister sites and better rock types. This pattern also holds true for Douglas-fir and grand fir, although these species performed better on the drier sites compared to ponderosa pine. On the relatively drier Okanogan and Umatilla study sites, Douglas-fir showed a strong growth response to N fertilization, while ponderosa pine

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[5] Unpublished data on file, Intermountain Forest Tree Nutrition Cooperative, University of Idaho, Moscow, Idaho

did not. During the Forest Health and Nutrition study, both species responded at about the same rates due to inclusion of moister sites.

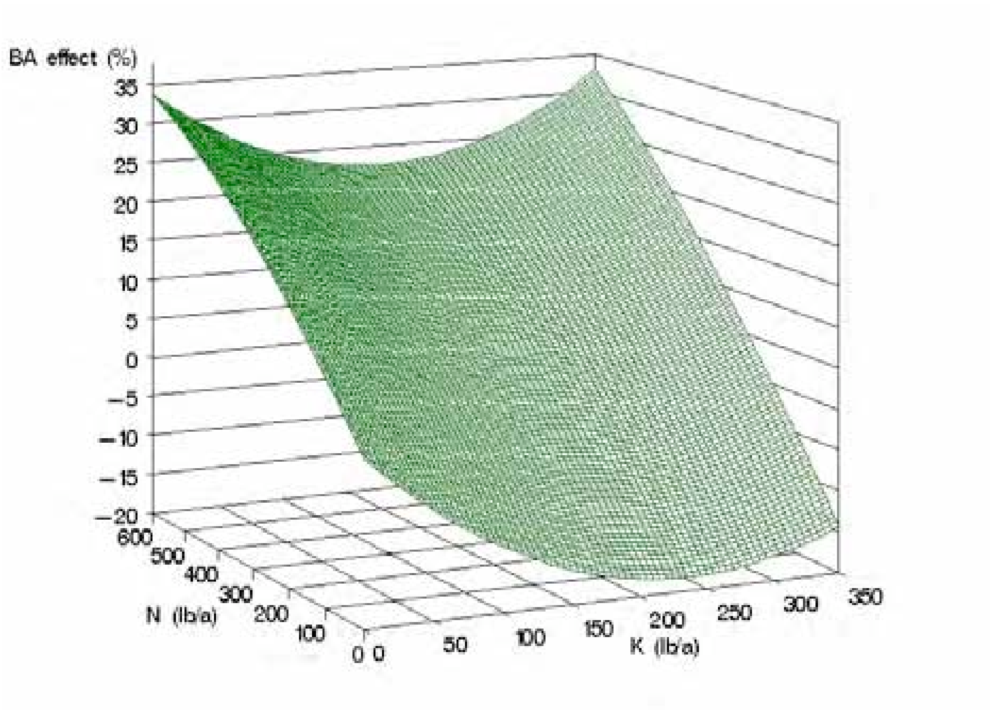

Figure 5—Relative basal area (BA) response of ponderosa pine on fertilized plots relative to unfertilized plots six years after application of nitrogen (N) and potassium (K) fertilizer during the Intermountain Forest Tree Nutrition Cooperative’s Forest Health and Nutrition study, for all rock types and vegetation series combined.

Table 4—*Cumulative frequency distributions of foliage weight and nutrient concentrations for unfertilized ponderosa pine from 37 sites across the Inland Northwestern United States. Modified from Moore and others (2004).*

Percentile	Weight[1] (g)	N (pct)	P (pct)	K (pct)	S (pct)	Ca (pct)	Mg (pct)	B (ppm)	Cu (ppm)
5	10.2	1.045	0.135	0.534	0.037	0.052	0.068	12.8	1.64
10	12.5	1.083	0.148	0.572	0.046	0.061	0.072	14.7	1.88
20	15.4	1.135	0.165	0.624	0.056	0.073	0.077	16.8	2.21
30	17.5	1.176	0.176	0.663	0.064	0.083	0.081	18.4	2.48
40	19.4	1.213	0.186	0.699	0.071	0.092	0.085	19.6	2.72
50	21.1	1.248	0.194	0.733	0.078	0.101	0.088	20.7	2.96
60	22.8	1.283	0.202	0.767	0.084	0.110	0.092	21.8	3.20
70	24.6	1.322	0.210	0.804	0.091	0.120	0.096	23.0	3.47
80	26.7	1.368	0.220	0.848	0.099	0.132	0.101	24.2	3.78
90	29.5	1.432	0.232	0.908	0.109	0.149	0.108	25.9	4.23
95	31.8	1.484	0.241	0.958	0.118	0.164	0.114	27.2	4.60

[1] Foliage weight for 100 needle fascicles and sheaths.

The cumulative frequency distribution of unfertilized plots by foliar K (*table 4*) suggests that the critical K level of 0.48 percent for ponderosa pine (*table 2*) may be reasonable, particularly given the generally low growth response (gross and net responses by both basal area and cubic foot volume) to K fertilization demonstrated in the various studies. Specifically, K availability in the Okanogan, Umatilla and Forest Health and Nutrition studies seemed adequate based on both initial K concentrations (all greater than 0.48 percent) and the low response of foliage K to fertilization. In the Forest Health and Nutrition study, while mortality of ponderosa pine appeared to increase with increasing N rate to almost 2 percent 6 years after fertilization at the highest N application rates, mortality decreased with increasing K rate to less than 1 percent at the highest K rates in the absence of N fertilization (*fig. 6*). Thus, K fertilization alone did not increase growth rates in this study (and may have slightly decreased growth rates), but appeared to decrease mortality rates. Results of this study were consistent with those of the Montana study, underscoring the importance of K in eliciting an N response by decreasing mortality. As with growth response, effects of N and K fertilization on mortality appeared related to rock type and moisture regime, with higher mortality rates occurring on drier sites and nutritionally poorer rock types following N-only fertilization, and lower mortality rates on moister sites and nutritionally better rock types. Potassium fertilization may mitigate mortality acceleration of N fertilization, as demonstrated in the Montana ponderosa pine study and the region-wide Forest Health and Nutrition study. However, it is important to note that K fertilization does not always have this effect, and that rock type (which affects soil characteristics), moisture regime and K fertilization all interact to determine growth and mortality response to N fertilization.

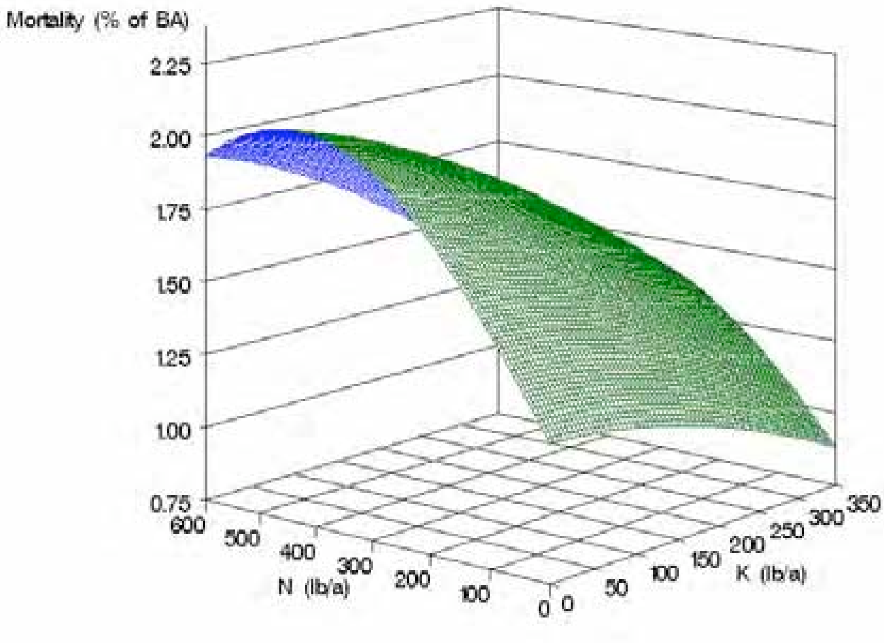

Figure 6—Ponderosa pine mortality as percent of gross basal area (BA) six years after application of nitrogen (N) and potassium (K) fertilizer during the Intermountain Tree Nutrition Cooperative’s Forest Health and Nutrition study, for all rock types and vegetation series combined.

Sulfur and micronutrients

Sulfur (S) was first tested by the IFTNC in 1991 during the Umatilla study (Garrison and others 2000), and again as a treatment applied to several sites during the Forest Health and Nutrition study in 1995 and 1996[6]. In both studies, foliage S concentrations for ponderosa pine were deficient on most of the control plots, and were above critical level following N+S fertilization. Grand fir and Douglas-fir in both studies were also S-deficient on unfertilized plots. While grand fir S concentrations increased to above-critical levels on most plots following S fertilization, most Douglas-fir S concentrations remained deficient. Thus, ponderosa pine behaved more like grand fir in terms of foliage response. Examination of ponderosa pine total foliage S content (S concentration times foliage biomass of 100 fascicles) in the Umatilla study showed that S contents (g 100 fascicles^{-1}) increased on the N+S treatments, providing evidence for S uptake by ponderosa pine. However, in neither study did S appreciably stimulate volume growth for ponderosa pine. Grand fir growing in the same stands with ponderosa pine in the Forest Health and Nutrition study showed positive responses to S fertilization, as did Douglas-fir to a lesser extent. During the Umatilla study, neither grand fir nor Douglas-fir responded particularly strongly to S fertilization, though Douglas-fir showed a marginally positive response, and both species responded better than ponderosa pine.

The cumulative frequency distribution of foliage S concentrations of ponderosa pine across the Inland Northwest (*table 4*) suggests that about half of the ponderosa pine stands in the Inland Northwest are S-deficient (critical S level = 0.08 percent, *table 2*). However the lack of growth response to S fertilization during various IFTNC studies suggests that this may not be the case. Reasons for the lack of growth response to S fertilization in ponderosa pine, particularly on apparently S-limited sites, are not entirely clear. A slight decrease in foliage weight for N+K+S treatment compared to N+K did occur during the Forest Health and Nutrition study, and when viewed together with the increase in foliage S concentration, might suggest S toxicity. A more reasonable hypothesis, however, was that some other element besides S might be limiting growth response on those sites.

The inclusion of several micronutrients in a multinutrient fertilization treatment during the 1996 Forest Health and Nutrition trials provided one of the first opportunities to test growth response to those elements in mature, second-growth mixed conifer stands in the Inland Northwest[7]. The multinutrient treatment included N, K, S, boron (B), copper (Cu), zinc (Zn) and molybdenum (Mo). Foliage nutrient characteristics measured one year after fertilization showed that ponderosa pine B concentrations increased from 16 to 39 ppm following micronutrient fertilization, raising foliage B levels from deficient (below critical level) to adequate (above critical level). While B concentrations also increased for Douglas-fir and grand fir, neither species was B-deficient in unfertilized trees. Ponderosa pine foliage Cu concentrations increased slightly from 2.6 to 2.9 ppm following fertilization, but remained below the critical level of 3.0 ppm. Grand fir and Douglas-fir Cu concentrations were both adequate following Cu fertilization. Ponderosa pine Zn concentrations increased from 28 to 60 ppm following micronutrient fertilization, raising Zn concentrations to adequate. Douglas-fir and grand fir Zn concentrations were adequate in unfertilized trees, and did not change following fertilization. Molybdenum concentrations increased from 0.3 to 0.5 ppm in ponderosa pine. Neither Douglas-fir nor grand fir showed foliage Mo responses to Mo

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[6,7] Unpublished data on file, Intermountain Forest Tree Nutrition Cooperative, University of Idaho, Moscow, Idaho

fertilization. Ponderosa pine needle weights increased from 27 g 100 needles^{-1} following N+K+S fertilization to 33 g 100 needles^{-1} following N+K+S+micronutrient fertilization, an increase attributable to the micronutrients. Grand fir and Douglas-fir needle weights did not differ between the same two treatments.

The increases in ponderosa pine foliage nutrient concentrations and needle weights following fertilization with N+K+S+micronutrients suggest that a growth response to that treatment might be expected. In fact, six-year gross basal area growth was 41 percent greater on those plots than the controls. Comparison of that response to the N+K+S response indicated that 8 percent of the growth response was due to Cu, Zn, Mo and B fertilization, demonstrating the potential importance of these less-studied elements. The rest of the growth response was due to N, but not to K or S. Because the micronutrient elements were applied in combination, it was not possible to determine which particular element(s) caused the growth response, but this finding did provide evidence that ponderosa pine may respond to micronutrient fertilization. In the same study, Douglas-fir and grand fir growing in mixed conifer stands did not show positive growth responses to micronutrient fertilization, though they did respond to S fertilization (though only marginally so for Douglas-fir). Sulfur fertilization increased six-year mortality in ponderosa pine and Douglas-fir, while S led to decreased mortality in grand fir. In contrast, micronutrients led to decreased mortality in ponderosa pine and Douglas-fir, but did not affect grand fir mortality. Disease, insect and weather-related death were the leading causes of mortality in all species.

Growth and mortality responses indicated that in mixed conifer stands, grand fir responded best to S fertilization but not to micronutrients. Douglas-fir responded marginally in growth to S and showed a slight decrease in mortality (but no change in growth) due to micronutrients. Ponderosa pine did not respond to S fertilization, but did respond in both increased growth and decreased mortality to micronutrients. Thus, stands dominated by grand fir might respond better to N, K and S fertilization, while those dominated by ponderosa pine might respond better to a combination of N, K and micronutrients. Douglas-fir should respond well to N and perhaps N+K fertilization (where site K limitations exist), but may respond only marginally to S fertilization, and not at all to micronutrient application.

Notably, these results at least partially disproved the previous hypothesis that another element might be limiting growth response to S fertilization in ponderosa pine. Even in the presence of positive foliage and growth responses to four important micronutrients, no S response was observed in ponderosa pine. Some additional experimentation with other elements may be warranted; however, possible S-toxicity suggested by foliage analysis also merits further consideration.

Nutrition of outplanted seedlings

While the various IFTNC studies provided new information on the growth and nutrition of ponderosa pine following various fertilization treatments, an additional factor affecting forest health became apparent during the site selection process for the Forest Health and Nutrition study. The study design included three sites each in a four by three sampling matrix based on four rock types (metasedimentary, granitic, basaltic and mixed) and three moisture regimes (Douglas-fir, grand fir and western red cedar/western hemlock vegetation series). Despite extensive review of numerous candidate sites, no suitable stands were found on metasedimentary rock types in the Douglas-fir series. Furthermore, only two sites were found on this rock type in the grand fir series, one of which was subsequently found to overlay granitics on about half the plots. The importance of rock type had been evident in previous studies; however, the difficulty of finding stands on metasedimentary rocks in the Forest Health and Nutrition study design further implicated underlying geology as an important component of stand health. Therefore, the Seedling Establishment study was designed with to compare establishment of seedlings on 'good' and 'bad' rocks.

The overall intent of this study was to determine whether fertilization could mitigate the perceived rock type effect detected during the Forest Health and Nutrition study. The study design called for selecting pairs of sites on differing rock types, with all other site characteristics being matched as closely as possible (IFTNC 1997). Six paired sites were selected throughout Idaho, Oregon and Washington. The rocks underlying the sites in each pair were rated as 'good' or 'bad' relative to each other (*table 5*). At each site, four blocks of six plots each were established. In the spring of 1998, two blocks each were planted with Douglas-fir and ponderosa pine, with species randomly assigned to the four blocks. Immediately after planting, fertilizer was applied as a subsurface treatment next to each seedling (*table 1*). The six fertilization treatments were randomly assigned to the six plots within each block. In 1999, a second fertilization was applied to the seedlings, using the same fertilizer treatments as previously applied on each plot, but doubling the rate and applying the treatment as a spot broadcast rather than subsurface. In 2000 a third fertilization was applied to a subset of sites. This consisted of a block-wide broadcast application of multiple nutrients plus hexazinone herbicide to one of the two blocks per species per site.

Table 5—*Relative rating of rock types as 'good' or 'bad' for paired study established at six locations in five regions during the Intermountain Forest Tree Nutrition Cooperative's Seedling Establishment study.*

Region	'Good' rock	'Bad' rock
Northeastern Washington	Quartz monzonite	Sericite schist
Central Washington	Teanaway pyroclastic	Roslyn formation sandstone
Central Washington	Basalt of Camas Prairie	Andesite of Laurel
North Idaho	Basalt of Onaway	Striped Peak quartzite
Central Idaho	Columbia River basalt	Quartz diorite gneiss
Northeastern Oregon	Ferro-basaltic andesite	Andesite

Seedling caliper and height were measured every year following establishment. Three-year volume growth was heavily affected by rock type. For ponderosa pine, those sites associated with 'good' rocks showed higher growth response and somewhat lower mortality than those associated with 'bad' rocks. Furthermore, only the multinutrient plus hexazinone treatment on good rocks produced positive growth increases over the controls. As in the Forest Health and Nutrition study, micronutrients seemed key for eliciting a ponderosa pine growth response, this time for seedlings. Furthermore, the importance of herbicide as a tool to reduce competition and increase availability of resources for seedlings was underscored. While Douglas-fir showed stronger growth responses than ponderosa pine, the pattern of better response on 'good' rocks was similar.

Treatments that included N increased Douglas-fir mortality on 'bad' rocks relative to the same treatments on 'good' rocks. For ponderosa pine, similar trends were evident, though not all treatments with N increased mortality. Moisture deficit patterns were examined for all sites during these three years, to determine whether moisture deficit, rather than rock type, might have affected the results. Patterns were similar for all three years, with about half the regions showing higher moisture deficit on the good rocks, and half showing higher moisture deficit on the bad rocks. Moisture deficit patterns therefore did not vary consistently with respect to ‘good’ or ‘bad’ rock, and did not explain the growth response results. In contrast, response did appear to be affected by rock type.

Foliage nutrient concentrations the first year after planting were affected by the nutrient-loading that occurred at the nursery. Some nursery effect was still present after the second fertilization treatment, though during both years effects of field fertilization were also evident in the foliage chemistry. The most significant finding of foliage analysis was a notable B deficiency. Boron concentrations on all plots following the first two treatments, and on the control plots following the third treatment, were at or below the recommended minimum of 20 ppm for both species. However, with application of the third multinutrient treatment, which included a higher B rate than the previous two fertilization treatments, B concentrations were well above critical levels, and notably higher than previous foliage B levels. The results suggested that B application rates applied in previous treatments were below levels required for Douglas-fir and ponderosa pine response.

The seedling establishment study has been followed on only four of the original 12 sites, largely because mortality due to vegetative competition and browsing resulted in too few live trees to continue the study with statistical reliability. However, initial results of growth response (caliper and volume) on all 12 sites as well as the continued growth response on the four remaining sites suggest that (1) a significant rock type effect exists and (2) boron was initially deficient in these seedlings and remained deficient after the first two fertilization treatments, likely hindering growth response to other applied elements. Mitigation of rock type effects through fertilization remains inconclusive, in part because of the apparent growth-limiting effect of B on seedling fertilization response during this study.

Nutrition of young plantations

A series of individual tree screening trials were established to test the effects of herbicide and various nutrients on tree growth. Screening trials provide a means of assessing a variety of fertilization and herbicide treatments within a relatively small area and short time frame. Treatments that provide the greatest short-term response are then selected for long-term plot-based trials. Between 1999 and 2000, 29 ponderosa pine screening trials were established in young (15 to 30 year-old) plantations (*table 1*), all of which contained ponderosa pine as a significant component. Each screening trial included up to nine treatments incorporating various combinations of fertilizer and herbicide. Five treatments common to all sites were an unfertilized control, N only, a multinutrient blend, herbicide only, and herbicide plus the multinutrient blend. The multinutrient blend included N, K, S, magnesium (Mg), Cu, B, Zn and iron (Fe) (*table 1*). The trials were located on sites with three moisture regimes, classified by Douglas-fir, grand fir and western red cedar series; and four rock types, classified as metasedimentary, mixed (principally glacial deposits), granitic and basaltic. While sites did not occur on all combinations of rock type and vegetation series, several comparisons of response across rock types and vegetation series were possible.

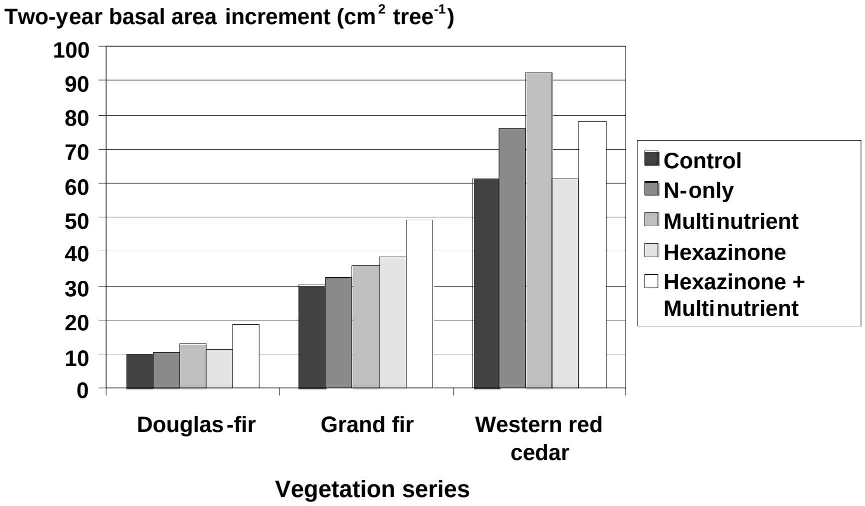

Figure 7—Two-year basal area increment of ponderosa pine trees from three vegetation series on granitic substrates following five fertilization and herbicide treatments during the Screening Trial study.

Trials on all three vegetation series on granitic rocks showed the positive effect of increased moisture on response to fertilization and/or herbicide (*fig. 7*). In the absence of fertilizer or herbicide, two-year basal area growth of ponderosa pine was significantly greater on western red cedar series than on Douglas-fir series, with grand fir series intermediate between the two. In the absence of herbicide, basal area growth was greater following both the N-only and multinutrient treatments on western red cedar series compared to Douglas-fir or grand fir. When herbicide was applied, alone or in combination with fertilizer, two year basal area growth on the grand fir series was greater than on the Douglas-fir series, but was not different from the western red cedar series. In other words, the addition of fertilizer alone increased the growth response on the moister sites compared to the moderate or drier sites, while the addition of herbicide, alone or in combination with fertilizer, increased the growth response on moderate sites over that of drier sites. Thus, management recommendations for enhanced basal area growth on granitic sites might include fertilizer alone on moist sites, herbicide plus fertilizer on moderate sites, and no fertilizer or herbicide on dry sites

Ponderosa pine response was compared between mixed, granitic and basaltic rock types within the grand fir series, and between metasedimentary, mixed and granitic rock types within the western red cedar series. Across the grand fir series, rock type did not have a great effect on basal area growth (*fig. 8*). Herbicide plus multinutrient fertilizer did produce a positive growth response on mixed rocks, but no other treatments showed growth responses. In contrast, rock type did seem to affect ponderosa pine growth on the western red cedar series (*fig. 9*). This finding in itself was interesting because it suggested that as moisture became less limiting, the rock type effect became more apparent. Of those sites on the red cedar series, unfertilized trees on metasedimentary rock types had lower absolute two-year basal area growth than unfertilized trees on granitic rocks, while unfertilized trees on mixed rocks were intermediate between the two. Application of fertilizer alone, whether N-only or multinutrient, resulted in greater absolute two-year basal area growth on granitic rocks compared to mixed rocks and metasedimentary rocks. However, herbicide applied with or without fertilizer resulted in greater absolute two-year basal area growth on mixed and metasedimentary rocks compared to granitic rocks. Thus, the lower growth rates on metasedimentary and mixed rocks appeared to be mitigated by the application of herbicide, whereas fertilizer alone produced the best response on the granitic sites. These findings suggested that while ponderosa pine growing on 'good' rock types and high moisture regime sites may benefit from multinutrient fertilization alone, ponderosa pine on most rock types and moisture regimes should benefit from the application of herbicide in combination with multinutrient fertilization.

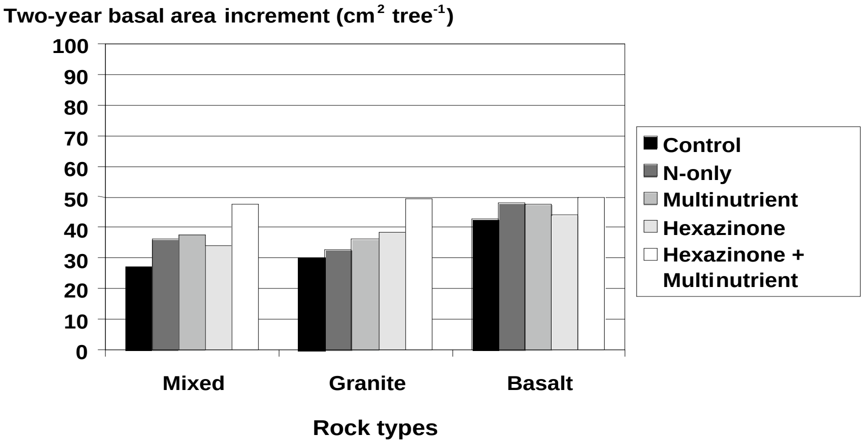

Figure 8—Two-year basal area (BA) increment of ponderosa pine trees on three rock types on grand fir series following five fertilization and herbicide treatments during the Screening Trial study.

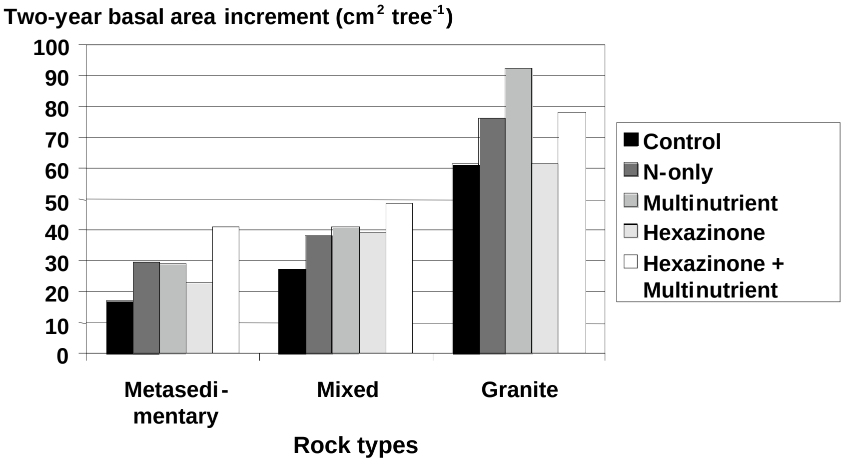

Figure 9—Two-year basal area (BA) increment of ponderosa pine trees on three rock types on western red cedar series following five fertilization and herbicide treatments during the Screening Trial study.

Conclusions

Nitrogen nutrition of unfertilized ponderosa pine in the Inland Northwest is generally better than that of its close associates, even when growing with those other species in mixed-conifer stands. Consequently, growth response to N fertilizer is somewhat lower for ponderosa pine compared to other species. While K fertilization does not usually elicit either a foliage K response or a growth response in any of these species, some evidence does exist that K fertilization can lead to decreased mortality in ponderosa pine. Sulfur fertilization has not been overly successful in increasing growth rates in ponderosa pine, and is probably not a cost-effective addition to operational fertilization regimes, unless other more responsive species such as grand fir are present. The most promising results for ponderosa pine growth response have occurred following fertilization with a combination of N, K and micronutrients, particularly B, Cu, Zn and Mo. Additional research into micronutrient and perhaps additional macronutrient fertilization of ponderosa pine is warranted. Also promising are results related to herbicide application, particularly on drier sites where competition for available moisture may be significant. Rock type appears to interact with moisture in determining tree growth response to herbicide application, such that tree growth on moist sites with 'bad' rock substrates may also benefit from herbicide plus multinutrient fertilization.

Ponderosa pine responds differently to fertilization than grand fir, and somewhat differently than Douglas-fir. Because ponderosa pine and Douglas-fir commonly occur together in stands, the two species could be considered for similar fertilization regimes. Fertilization blends containing N, micronutrients and perhaps K (on K-limited sites) should suffice for these species. For stands containing grand fir, the addition of S to the fertilization regime may elicit a growth response in that species and in Douglas-fir, but not ponderosa pine. Average fertilizer response is greater on moister sites such as the western red cedar series, western hemlock series and moister habitat types and plant associations within the grand fir series, compared to drier sites such as the Douglas-fir series and drier habitat types and plant associations within the grand fir series.

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The authors acknowledge and thank all current and past members of the Intermountain Forest Tree Nutrition Cooperative, whose collective direction and support during the last 24 years have contributed tremendously to the current state of knowledge regarding Inland Northwest forest tree nutrition. Thanks also to the two technical reviewers of this manuscript, whose comments and suggestions greatly improved the clarity and content of this paper.

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Ungulate Ecology of Ponderosa Pine Ecosystems in the Northwest[1]

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Abstract

Ponderosa pine ecosystems provide important foraging habitats for both wild and domestic ungulates. Livestock typically graze ponderosa pine ecosystems from May through October. Mule deer and elk may utilize these habitats on a yearlong basis in some areas. Stand density has a significant effect on understory production. Competition for soil moisture and nitrogen limit understory production. Since these systems typically exist at lower elevations, south aspects and on rather shallow soils, soil moisture is usually unavailable to understory species by mid summer. Optimal forage quality, therefore, occurs from late spring through mid-summer. Livestock use for optimal production should occur during this time frame. Herbivory by both native and wild ungulates can influence the structure and composition of understory vegetation. Ungulates, through the act of selective foraging influence the competitive ability of understory plants utilized. Ungulates can by considered agents of change in ecosystems by three processes: the regulation of process rates, modification of spatial mosaics, and action as switches controlling transitions between alternative ecosystem states. These understory composition and structure changes may have important implications to such diverse attributes as nutrient cycling, energy flow, biodiversity, stand density, fire type and interval, forest productivity, and ungulate productivity. Key Words: habitat, deer, elk, herbivory, fire, and nutrition.

Introduction

Ponderosa pine (*Pinus ponderosa* P.& C. Lawson) ecosystems are the most extensive and valuable grazed forestlands in North America (Skovlin and others 1976). In Oregon and Washington 4.5 million hectares (11 million acres) of ponderosa pine-bunchgrass type exists (Skovlin and others 1976), and in the four-state region of Colorado, New Mexico, Arizona and Utah there exists 10.5 million hectares (26 million acres) (Lowe and others 1978). These ecosystems additionally provide valuable habitat for wild ungulates (primarily mule deer and elk) (Skovlin and others 1976, Urness and others 1975). Specific habitat provided varies greatly with stand density, and may provide primarily foraging habitat at lower densities and security cover at higher densities. Habitat quality for both wild and domestic ungulates is often a function of past management practices such as logging and fire exclusion or natural disturbances such as fire and insect outbreaks. Herbivory, by both wild and domestic ungulates, can act as a chronic form of disturbance

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interacting with periodic disturbances (logging, fire) to develop alternative steady states of understory succession, which, in turn, may confound restoration efforts (Riggs and others 2000). Weigand and others (1993) related that ungulate herbivory could have substantial impacts on site productivity when used in conjunction with prescribed fire, stocking control, and species selection.

Habitat Characteristics

Ponderosa pine communities may be used by mule deer and elk on a year round basis (Skovlin and Vavra 1979), depending on elevation and snow conditions. Cattle use of these ranges is usually in summer and early fall (Skovlin and others 1976). Depending on stocking density, ponderosa pine communities have the potential to provide security cover and foraging habitat to ungulates. Certainly site specific variables such as aspect, soil depth, and annual precipitation have an influence on understory production of ponderosa pine communities. The principal variable, however, is probably stand density. Competition for nitrogen and water between the overstory and understory vegetation limits understory development (Riegel and others 1991).

Ponderosa pine stands in the western U. S. have long been recognized as overstocked and not at potential for producing forage for ungulates (McConnell and Smith 1970, Weaver 1951). However, with the passage of the Healthy Forest Restoration Act in 2003, increases in forage production can be expected (Germaine and others 2004). Germaine and others (2004) went on to caution that, if large treatment blocks are initiated, then use by mule deer could be precluded if all resource needs are not met within the treated area. The same can be hypothesized for elk. Manipulation of ponderosa pine communities has the potential to impact understory productivity and diversity, forage nutritional quality, and microhabitat characteristics. Post-manipulation management, e.g. road access, also has the potential to affect use of treated areas at least by wild ungulates.

Understory productivity

Of primary concern to domestic and wild ungulates is meeting nutritional needs. Herbivores commonly seek foraging areas that are high in understory production (DelCurto and others 2000). Understory productivity is controlled to a large extent by the density of the overstory (Riegel and others 1991). With fire exclusion and the resultant regeneration and ingrowth of trees (Germaine and others 2004), and the near cessation of public land timber harvest, ponderosa pine forests now provide much less forage than they did historically. Rehabilitation treatments will focus on reduction of basal area, overstory canopy cover, and stem density (Germaine and others 2004).

Previous research on thinning and understory production effects provides an indication of the increases in forage that can be expected. McConnell and Smith (1970) thinned 48-year-old overstocked pine sapling stands to 625, 330, and 165 trees per hectare (253, 134, and 67 trees per acre) while maintaining control stands at 6,916 trees per hectare (2,800 trees per acre). The authors reported a 10 kilogram per hectare (9 pounds per acre) increase in grass yield and a 2.2 kilogram per hectare (2 pounds per acre) increase in shrub yield for each 30.8 cm (one foot) increase in pine spacing. Total understory production varied from 112 kilograms per hectare (100 pounds per acre) for controls to 384 kilograms per hectare (342 pounds per acre) for the165 trees per hectare (67 trees per acre) thinning. Gibbs and others (2004) working in South Dakota found that heavily thinned stands of ponderosa pine exhibited a standing biomass 2.9 times greater in shrubs, 6.1 times greater in graminoids, and 13.0 times greater in forbs than standing biomass of unthinned stands. Moderately thinned stands had a biomass 3.4 times greater in shrubs, 3.4 times greater in graminoids, and 4.9 times greater in forbs than did unthinned stands.

Walburger and others (2005) reported on a long term data set collected after a commercial thinning in ponderosa pine stands. Tree density was reduced from 346 to 148 trees per hectare (140 to 60 trees per acre) and canopy cover reduced from 61 to 29 percent. The study area was logged in 1985. Total understory production was twice as high in the thinned stands compared to controls in 1989 but the difference declined since this initial response (*fig. 1*). The site potential for understory production of ponderosa pine communities is excellently portrayed by these studies. Busse and others (2000), however, remarked that central Oregon ponderosa pine stands had six-fold lower understory production than did comparable stands in northeastern Oregon. Bennett and others (1987) found herbage yields similar among a gradient of soil types with dense canopy covers. However, with open canopies, soil production potentials were fully expressed with the more productive soils producing 3 times the herbage of the least productive at 10 percent canopy cover.

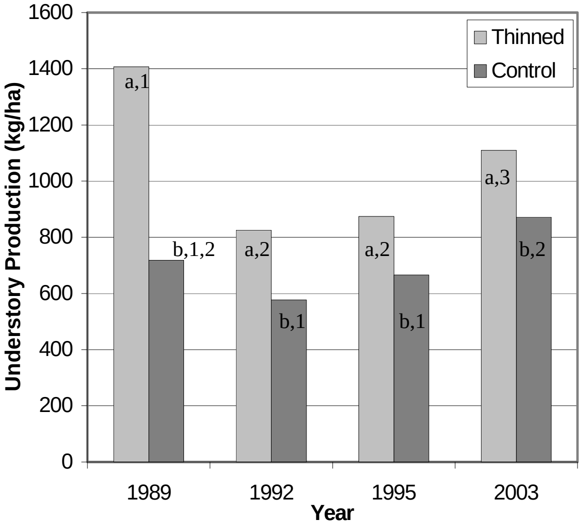

Figure 1— The effects of timber harvest through commercial thinning (Thinned) vs. no harvest (Control), on understory production (kg/ha) in a Ponderosa pine habitat type in northeastern Oregon. Bars with differing letters, within year, differ at α = 0.05, and bars with differing numbers differ at α = 0.05.

Fire may also have dramatic influences on forage production. Fire that reduces canopy cover may have the same stimulatory effect on understory production as overstory thinning. Busse and others (2000) provided citations of research that reported inconsistent understory response with application of low-severity understory burns. Their reasoning for lack of response was that low-severity fires did not reduce tree canopy and therefore competition for soil nitrogen and water remained nearly the same. Their research did not reveal any significant differences in grass or forb productivity as a result of fire, but shrub cover (principally *Purshia tridentata*) was reduced 50 percent compared to controls.

Wildfires that are typically stand replacing in nature usually occur on areas where extensive fuels have built up over time. In this situation the suggested benefits of fire-reduced competition for soil resources, reduced litter, and increased nutrient availability (Busse and others 2000) may increase understory productivity. Lowe and others (1978) generalized expected responses of grasses and forbs to wildfire (*fig. 2*). The frequency of the principal shrub in their studies, *Ceanothus fendleri*, dropped to zero as a result of the fire then increased by three to four times the pre-fire condition. The initial response of the understory may be typical across the western U.S. Persistence of improved understory biomass is, however, dependent on the rapidity and increasing density of the regenerating overstory.

Understory productivity can be enhanced by a disturbance or management practice that reduces canopy cover. Fire treatments may lead to an increase or decrease in the shrub component over time depending on species composition. Low intensity fires that do not cause mortality in the overstory will probably provide an inconsistent response in understory vegetation.

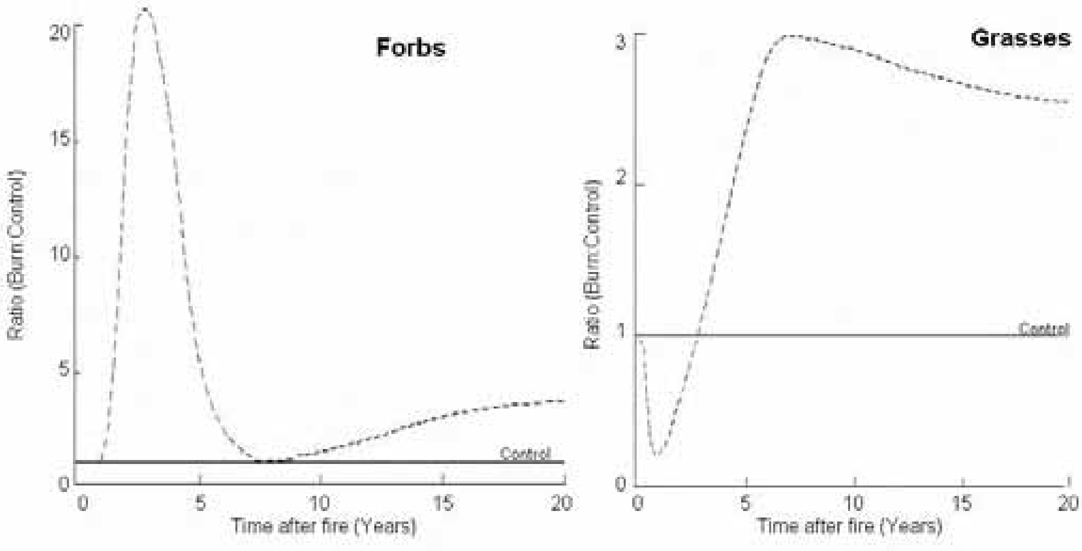

Figure 2— Time-trend response curves for basal area of forbs and grasses (Lowe and others 1978).

Forage nutritional quality

Meeting nutritional needs is not just a case of adequate forage productivity; the animal must obtain forage of sufficient quality to satisfy physiological requirements (lactation, growth). Management manipulations and natural disturbances may improve the quantity of forage present but can in some cases result in a decline in forage quality compared to untreated stands (Svejcar and Vavra 1985).

The results of several studies are available that provide information on forage quality of individual plant species important to ungulates (Hickman 1975, Skovlin 1967) and quality of actual cattle diets, (Holechek and others 1981, Vavra and

Phillips 1979). Generally, in ponderosa pine ecosystems with summer drought, a decline in herbaceous forage quality occurs as plants mature and senesce, and nutrients are translocated into the roots. The rate of decline is affected by annual precipitation (Holechek and Vavra 1983).

Management manipulations that decrease canopy cover may accelerate the decline in nutritional quality of forage. Svejcar and Vavra (1985) speculated that decreasing canopy cover allows more sunlight to reach the soil surface which warms the soil more rapidly in the spring and dries it more rapidly in the summer, accelerating plant phenology. Low biomass productivity sites can also be expected to lose forage nutritional quality more rapidly as the summer progresses due to limited soil moisture and competition for that supply (Busse and others 2000). In northeastern Oregon, nutritional quality of ponderosa pine ranges deteriorates below required levels by mid- to late-August. Declining herbaceous forage quality can be mitigated in animal diets by the presence and consumption of palatable understory shrubs. It is generally accepted that mule deer and elk employ this dietary strategy (Ganskopp and others 1999). Typically on forested ranges, cattle will also increase their consumption of shrubs as herbaceous forage quality declines (Holechek and others 1982). Therefore management strategies that promote development of a diversity of understory species should provide adequate nutritional quality to ungulates later in the season. Problems of grazing distribution might, however, develop late in the gazing season in regard to concentration of animals around riparian areas. Typically, upland water dries up or has stagnated in late season, and upland forage is mature and dry. Both of these factors contribute to increased utilization of riparian vegetation.

Cover

Much of today's ponderosa pine ecosystems are densely stocked, with closed canopies and provide no concern for lack of cover. With the implementation of the Healthy Forest Restoration Act, however, concern may develop if large blocks are uniformly treated. Additionally, silvicultural activities during restoration may disrupt mule deer and elk behavior (Edge and Marcum 1985, Ward 1976). Elk appear to be more affected by the act of timber harvest and concomitant activities like road building and resultant traffic (Leege 1976, Ward 1976, Wisdom and others 2004). From a review of the literature, Lyon and Christensen (2002) found elk could be displaced from habitats by as much as five miles. Most often, however, the distance moved appeared to be the minimum required to avoid contact with equipment and people. Continual timber harvest within an individual watershed for five consecutive years can impose learned behavior that delays return to previously used habitats (Lyon 1979). Edge and others (1985) did report that home ranges of individual animals were not altered when areas of extensive cover remained available within their home range. The authors speculated that where cover becomes limited, harvest activity may increase home-range size and reduce home-range fidelity.

If timber harvest activities decrease or increase human access to an area, elk distributions can be expected to shift, with elk avoiding areas with increased access, and selecting areas with little or no access (Wisdom and others 2004). Specifically, road density and traffic rates in areas open to vehicular travel negatively influence elk distribution; elk avoid habitats near roads open to traffic (Rowland and others 2000, Wisdom and others 2004). The influence is not demonstrably linear, however, with roads having no apparent influence at zero or very low traffic rates. Cole and others

(1997) found that road management areas where access was restricted to administrative uses reduced Roosevelt elk home range size. Additionally, Lyon (1976) found that elk used habitats with greater canopy closure in areas of higher road density.

Mule deer security cover and bed sites may be compromised (Germaine and others 2004) as a result of restoration activities. These authors found that, in a ponderosa pine landscape treated under a typical restoration prescription, mule deer day-bed (security) microhabitat was reduced 50-100 percent, while available foraging microhabitat was increased 30-60 percent compared to untreated forest. The authors went on to suggest that stands of saplings and pole-sized trees having greater than 40 percent midstory canopy closure should be retained in patches of at least 0.04 hectares (0.1 ac) during restoration efforts. Thomas and others (1979) suggested that an optimal landscape for mule deer and elk would be comprised of 40 percent cover and 60 percent foraging habitat. Clary (1972) prescribed timber basal areas of 9.1 to 18.3 square meters per hectare (40 to 80 square feet per acre) to improve big game habitats in ponderosa pine forests. Additionally, one pine sapling thicket per 40 hectares (100 acres) was recommended for big game bedding cover. Hillis and others (1991) defined a security area for elk as a 101 hectare (250 acre) block that was non-linear in shape and at least one-half mile from roads. Such areas should compose at least 30 percent of a watershed. Gibbs and others (2004) felt that unthinned stands that provide essential thermal and hiding cover could become more important where availability of these habitats is more limited.

In contrast to these conclusions, Wisdom and others (2004) noted changes in elk spatial distribution during timber harvest but found no loss of animal performance attributable to that change. During harvest activities elk traveled over areas twice as large as areas covered prior to harvest indicating disturbance to the animals (Edge and others 1985). After harvest animal spatial distribution decreased from the levels occurring during harvest but was still higher than pre-cut levels.

No major changes in cattle spatial distribution during a timber harvest operation due to harvest activities are expected since cattle are domesticated animals, so are familiar with and tolerate human activity. After harvest, however, the physical changes in the grazing environment related to harvest may redistribute cattle. Harris (1954) reported that cattle seldom use dense overstory canopies except in conditions of extreme heat or intense insect harassment. Hedrick and others (1968) found it more difficult to obtain moderate or heavy utilization under dense overstory canopies than under low-density canopies. Following timber harvest and the concomitant decrease in overstory canopy a release of understory production usually occurs. This release may vary from two to eight times pre-harvest forage production (Svejcar and Vavra 1985) depending on intensity of the cut, site potential, and soil disturbance (Hedrick and others 1968). Miller and Krueger (1976) reported that 60 percent of the forage consumed in a given pasture by cattle was from areas logged and reseeded. Road construction to facilitate harvest also provides improved distribution of cattle by improving access (Hedrick and others 1968). Restoration efforts therefore, should provide new grazing areas for cattle.

Stand replacement wildfires that are unusually intense due to long-term fire exclusion and unnaturally high stand densities create landscapes usually devoid of cover. In these cases effects on ungulates are similar to severe canopy reduction from timber harvest, so the same mitigating management options should be considered. Forage is not a limiting factor but cover may be. Increased vulnerability to hunting, distribution alteration due to open roads, and increased vulnerability to predators are possible.

Ungulate use of large burns is usually a function of forage availability, but may be influenced by other factors like livestock grazing (Lowe and others 1978, *fig. 3*). For example, elk summer fall use declined after introduction of sheep grazing in summer on a 20-year burn (*fig. 3*).

Herbivory Effects

Following disturbances like large fires or fuels reduction treatments, secondary succession of the understory and regeneration of conifers are initiated. These areas often become focal points of ungulate herbivory for two reasons: 1) vegetation developing after disturbance is often more palatable to ungulates relative to that available on undisturbed sites; and 2) surrounding untreated or unburned forest communities with dense canopies contain limited forage in the understory. Large herbivores are attracted to areas that are characterized by relatively high biomass of palatable food resources, and thus can be expected to focus foraging activity in recently disturbed areas.

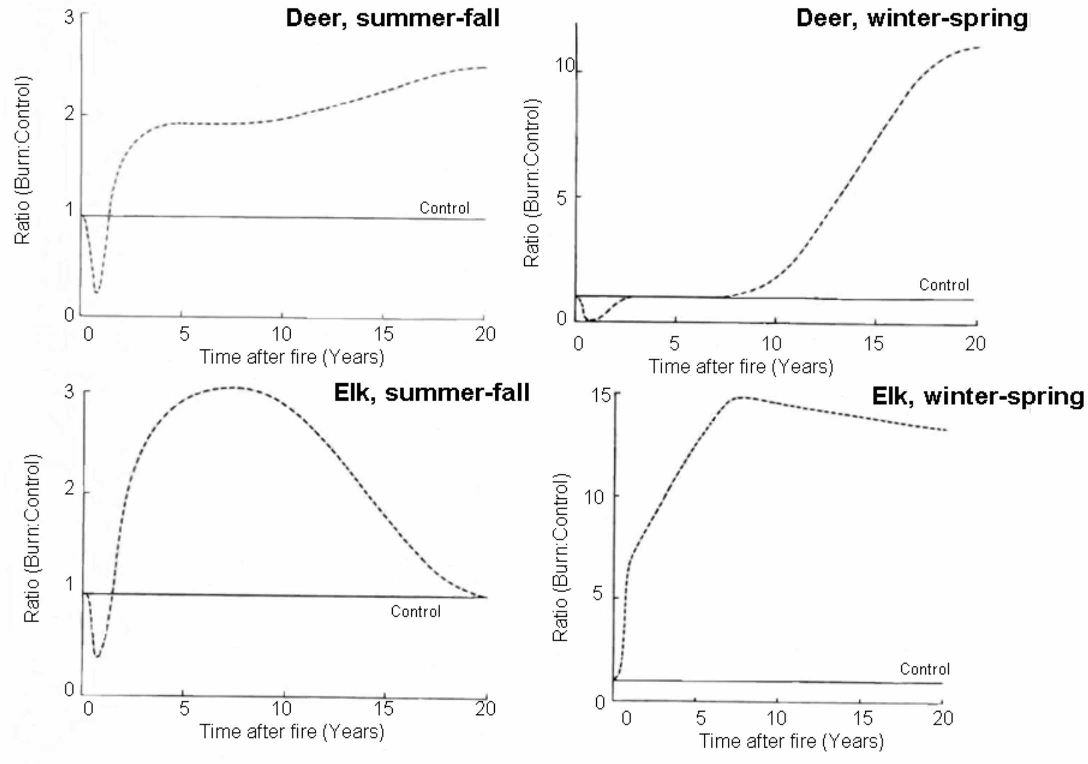

Figure 3 —Time-trend response curves for deer and elk use as a ratio of burn use to control use (Lowe and others 1978).

Only rudimentary data exist (Riggs and others 2000), but ungulate herbivory is strongly implicated as a significant agent in altering successional trajectories following disturbance (fire, logging, fuels reduction) in the Blue Mountains. Raedeke (1988) related that selective feeding of forest animals can result in complete changes in the structure, composition, and productivity of the forest. In general, plant communities within exclosures are more diverse than the surrounding forest community subjected to continual herbivory (Raedeke 1988). Recent literature reviews (Augustine and McNaughton 1998, Hobbs 1996) clearly indicate the important role of herbivory not only in modifying the composition of plant communities, but also of other biotic communities in the ecosystem. Ungulates are important agents of change in ecosystems by three processes: regulation of process rates, modification of spatial mosaics, and action as switches controlling transitions between alternative ecosystem states (Hobbs 1996). In the Blue Mountains herbivory has long been recognized to be a competitive factor in ungulate relationships (Cliff 1939, Pickford and Reid 1943) and in understory shrub suppression (Mitchell 1951). However, the role of herbivory is not well recognized in the predominant management paradigms, either in the Blue Mountains or any other forest ecosystem of the western U. S., and knowledge is more anecdotal than predictive (Riggs and others 2000).

Jones (2000) reviewed the effects of cattle grazing and reported that cover of grasses and shrubs, as well as total vegetation biomass often was reduced. Riggs and others (2000) reported that, in the Blue and Wallowa Mountains of eastern Oregon, understory biomass in ungulate exclosures was 2.1 times greater inside than outside enclosures, and forest-floor biomass was 1.5 times greater inside than outside. Shrub biomass was influenced more by ungulates than was grass or forb biomass. Mazancourt and Loreau (2000) stated that herbivory, in general, could lead to species replacement in plant communities. Augustine and McNaughton (1998), stated that species composition of plants can be dramatically altered by selective foraging of ungulates and that this phenomenon is a trademark of plant-ungulate relations. The authors noted that by altering the competitive relations among plants, differential defoliation tolerances to grazing of co-occurring plant species appear to be important determinants of how woody and herbaceous plant communities respond to herbivory.

Augustine and McNaughton (1998) went on to include effects on overstory species as well, and listed several species of coniferous and deciduous trees that were reported as herbivory intolerant. Healy (1997) found that white-tailed deer interrupted the sequence of stand development and simplified understories in eastern oak forests. Likewise, Alverson and Waller (1997) found that white-tailed deer abundance reduced the size and number of hemlock seedlings and explained the widespread regenerative failure of eastern hemlock. The literature review of Alverson and Waller (1997) also indicated that white-tailed deer substantially altered tree, shrub and herbaceous components of plant communities.

Ungulate herbivory shapes vegetation pattern in coniferous forests of the Northwestern U.S.(Schreiner and others 1996, Woodward and others 1994). Research by these authors indicated that ungulates maintained a reduced standing crop, increased species richness of forbs, and determined the distribution, morphology, and reproductive performance of several species of shrubs. Woodward and others (1994) stated that the extent to which herbivores can change forest ecosystem processes might depend on the scale and magnitude of other disturbances. All of the observations are consistent with available information in interior forests.

With ungulates, attraction to and therefore selectivity for areas of lower canopy cover and greater forage production may create a chronic disturbance effect after timber harvest. Walburger and others (2005) reported that timber harvest had a greater effect on understory vegetation than did herbivory, but many species were affected by the interaction of timber harvest and herbivory (*fig. 4)*. Plant production was measured as peak standing crop in the absence of cattle grazing the year measured. However, herbivory affected production and presence of grasses, forbs and shrubs. Total understory production was 124 kilograms per hectare (110 pounds per acre) lower in cattle grazed pastures than in exclosures. Production of pinegrass and elk sedge was also reduced by cattle grazing; whereas, Kentucky bluegrass and other perennial grass production was not affected by grazing. The total production of forbs was not affected by herbivory; however, there were varying species effects.

Walburger and others (2005) also reported that shrub production was moderately affected by herbivory and that species diversity was not affected by either timber harvest or herbivory. This lack of response of species diversity and shrub production to herbivory may be due to years of fire exclusion or to prior management.

Forest succession is a function of edaphic factors, the density and viability of seed and sprouting rootstocks, episodic disturbance regimes, and herbivores (Riggs and others 2000). Herbivory alters the utilization profile of a plant community (palatable versus unpalatable plants) and thus can alter its successional trajectory. Herbivores influence growth, recruitment, and mortality rates of plants and may do so in ways correlated with plant density, frequency, or other neighborhood traits, or with competitive abilities (Huntly 1991). Moreover, herbivory may increase, offset, or generate reciprocal negative interactions among plants (Huntly 1991). Herbivory is highly variable in space and in time, and these spatial and temporal patterns of herbivory can generate heterogeneity in plant population structure, the existence of which strongly influences community dynamics.

Management Implications

Ponderosa pine ecosystems have been modified by fire exclusion for most of the twentieth century. Consequently, ungulate habitat provided by these ecosystems has changed dramatically. In the coming years, resources will be directed to the rehabilitation of these ecosystems to restore some target structure and functionality. If ungulate habitat is a priority then certain considerations are in order for improving effectiveness of rehabilitation treatments and follow-up management.

In order to persist in an environment, wild ungulates, must utilize food that results in a given level of fitness, allowing persistence or growth of their population in that environment (Belovsky and others 1999). Likewise, utilization of that food must occur with minimum risk to the individuals (predation avoidance). Therefore, in the management of ungulates, forage resources and cover are considerations. The current condition of most ponderosa pine ecosystems is such that cover is not limiting. As fuel reduction treatments are implemented, however, cover may become a limiting factor. Uncut or unburned cover patches should be considered where treatment patches are large. Topography plays a mitigating role, so that in landscapes with irregular topography, cover patches can be fewer and strategically placed. Cover provides for predator avoidance, both human and animal.

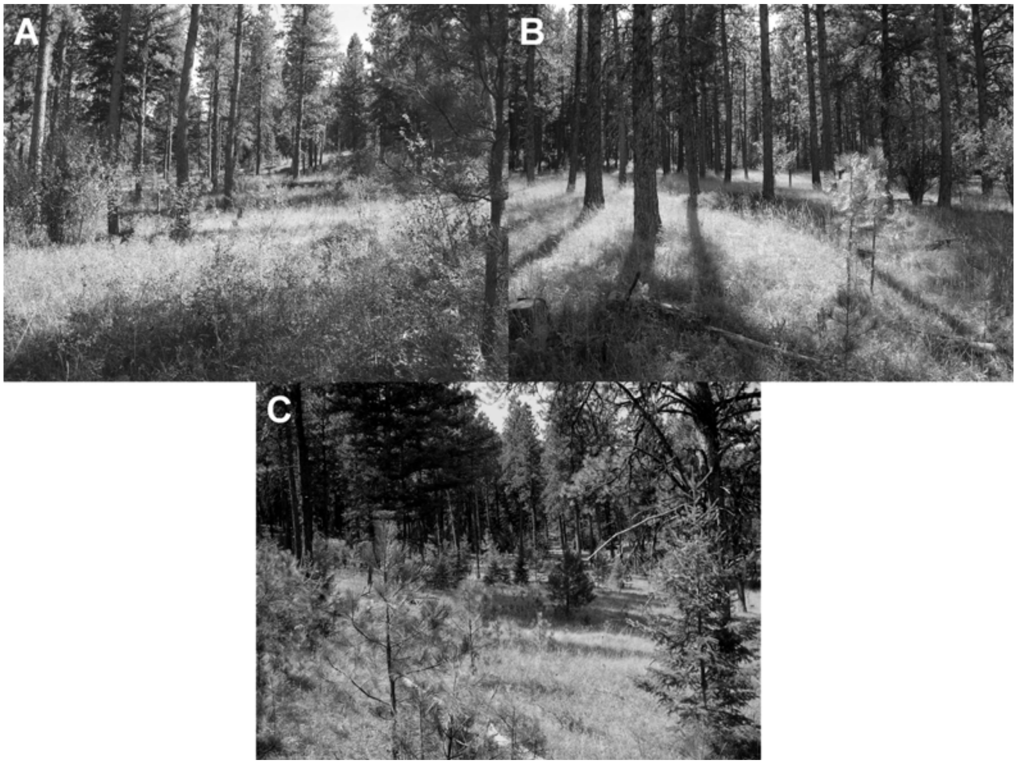

Figure 4 — Influence of herbivory, A) big game (elk and deer) and cattle exclosure, B) grazed by big game and cattle and C) grazed by big game only, on a commercially thinned Ponderosa pine site in northeastern Oregon.

Once areas have been treated, the roads developed to gain entry also allow easy access for hunters, increasing animal vulnerability to harvest. Open roads that provide both easy access and decreased tree density after thinning may result in animal harvests above desired levels. Road closures not only limit hunter access, but in the case of elk, prevent compression of available habitats because of their aversion to road traffic.

With a reduction in overstory canopy cover, increased forage production is possible. The degree to which it occurs will be dependent on site factors and the remaining tree density. Late summer forage nutritional quality, however, may be impacted by the fuels reduction/thinning process because maturation and senescence of herbaceous plants may occur earlier in the summer in thinned areas (Svejcar and Vavra 1985). Livestock should, where possible, utilize ponderosa pine communities early in the summer to optimize the capture of plant nutritional value (Vavra 1983). Problems of over-utilization of riparian vegetation likely will occur in late summer.

The development of prescribed fire intervals for ponderosa pine ecosystems poses questions for ungulate nutritional ecology. Frequent return interval prescribed burning may result in decreased productivity of forbs and shrubs (Tiedemann and others 2000). These two forage components are important in optimizing quality of ungulate diets. Shrubs play a particularly important role in late summer and fall when grasses have senesced and declined in nutritional quality (Vavra 1983). Tiedemann and others (2000) note that the issue was not whether to burn, but rather, defining the minimum interval to burn without unacceptable risks to productivity.

Herbivory by wild and domestic ungulates has the potential to impact vegetation dynamics following rehabilitation efforts on ponderosa pine ecosystems. In ponderosa pine stands Walburger and others (2004) did not note the large differences due to herbivory on grand fir associations observed by Riggs and others (2000). Changes to productivity and structure may be the most pronounced herbivory effects on ponderosa pine ecosystems. Within a given watershed, size and number of rehabilitation units, as well as anticipated ungulate (wild and domestic) utilization are important considerations if management goals include specific levels of understory plant community productivity, diversity and structure.

Ponderosa pine ecosystems are important to ungulates throughout the western U.S. These diverse and wide-spread ecosystems have been recognized to be undergoing alterations attributable to fire exclusion for more than half a century (Weaver 1951). Also, unregulated historic livestock grazing has played a role in the alteration process (Belsky and Blumenthal 1997). Rehabilitation efforts in the next few years will begin to drastically change the composition and structure of these ecosystems. Ungulate habitat will continue to be a critical product of these ecosystems, although specific use at a given location may change (security cover converted to foraging habitat).

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A Comparison of Bird Species Composition and Abundance Between Late- and Mid-seral Ponderosa Pine Forests[1]

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Abstract

We compared the relative abundance of bird species between two ponderosa pine (*Pinus ponderosa*) forests in northeastern California: one with a canopy of large old-growth trees present (Blacks Mountain Experimental Forest, BMEF) and the other with large trees essentially absent (Goosenest Adaptive Management Area, GAMA). We surveyed 24 units at BMEF and 20 at GAMA using point counts and compared the relative abundance of bird species detected at the two locations using Wilcoxon-rank tests. Overall bird species composition was similar at the two locations. Of the 51 species detected at both locations, 29 were detected at both GAMA and BMEF, 14 were only detected at BMEF and 8 were only detected at GAMA. Most of the species that were detected at only one site were rare at the site where they were observed. Plot diversity (the number of bird species detected on a plot) did not differ between the two locations. Bird species were lumped into four foraging guilds, woodpeckers, bark gleaners, foliage gleaners, and flycatchers, to examine if foraging ecology predicted differences in abundance between the two sites. Woodpeckers, bark gleaners, and flycatchers were more abundant at BMEF while foliage gleaners were more abundant at GAMA. Differences in the abundances of individual species were generally consistent with the overall guild differences. For instance, Williamson's Sapsuckers (*Sphyrapicus thyroideus*), Hairy and White-headed Woodpeckers (*Picoides villosus* and *P. albolarvatus*), and Northern Flickers (*Colaptes auratus*) were all significantly more abundant at BMEF than GAMA. The only species whose abundance was not consistent with the difference in guild abundance was the Red-breasted Nuthatch (*Sitta canadensis*), a bark gleaner, which was more abundant at GAMA. These results are consistent with other studies suggesting that woodpeckers and bark gleaners are strongly associated with large trees and snags. The dense canopy of small to medium sized trees at GAMA relative to BMEF may account for the higher density of foliage gleaners at GAMA.

Introduction

Ponderosa pine (*Pinus ponderosa* P. & C. Lawson) and related Jeffrey pine (*P. jeffreyi* Grev. & Balf.) forests which occur from Baja California to British Columbia, and from the Cascade and Sierra Nevada Mountains eastward to the Rocky Mountains, have perhaps changed more in the last century than any other forest type in western North America (Covington and Moore 1994). Two major changes have occurred in these forests: a decrease in the mean diameter of the trees and a change in

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the relative proportion of tree species. Many forest stands that had been composed almost exclusively of ponderosa pine are now numerically dominated by white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco), and grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.). The change in the mean diameter of the trees is a result of extensive logging of ponderosa pine forests over the last century while the change in species composition is due to the exclusion of fire and possibly other factors over the last several decades (Agee 1993, Mohr and others 2000, Odion and others 2004, Perry and others 2004, Skinner and Chang 1996).

Although there has been more focus on effects of logging on wildlife in wetter forests of the western United States (e.g., Hejl 1994, Morrison 1992, Sallabanks and others 2000, Zarnowitz and Manuel 1985), there nonetheless are several studies that have examined the effects of logging on wildlife in ponderosa pine forests (Blake 1982, Brawn and Balda 1988, Mannan and Meslow 1998, Szaro and Balda 1979a). From the literature, it appears that the presence of large-diameter trees is critical to some species of wildlife (Brawn and Balda 1988, Finch and others 1997, Weikel and Hayes 1999), but there is a general paucity of good controls or in multi-year replication in such evaluations (Sallabanks and others 2000).

This paper is the part of a series of evaluations of ponderosa pine wildlife and wildlife interactions in eastside pine forests of northern California (Farris and others 2002, Farris and others 2004, Liebezeit and George 2002, Zack and others 2002). We participated in the design and implementation treatments at two sites (George and Zack 2001, Zack and others 1999): Blacks Mountain Experimental Forest (BMEF) (Oliver and Powers 1998) in Lassen National Forest, and Goosenest Adaptive Management Area (GAMA) (Ritchie and Harcksen 1999) in the eastern part of Klamath National Forest. In brief, the treatments are large-scale (at least 40 hectares), replicated, with different thinning alternatives and with and without prescribed fire.

Here we provide a baseline and comparative basis for understanding the avifaunas at BMEF and GAMA prior to such treatments. BMEF was designated as an experimental forest in the 1930s, and our plots had an outstanding legacy of large and old ponderosa and Jeffrey pines (Oliver 2000). In contrast, GAMA was clearcut in the early 1900s (Ritchie and Harcksen 1999) and has had little logging activity since, making it typical of many western ponderosa pine forests. Thus, we have an opportunity to compare the avifauna between a relatively intact forest of large diameter trees (BMEF) with a mid-seral forest of medium-diameter trees (GAMA). Both sites prior to treatments had considerable encroachment of white fir and high densities of small trees and shrubs in the understory, all typical effects of fire exclusion.

Methods

Study Sites

Blacks Mountain Experimental Forest (lat. 40°40’N., long. 121°10’W.) is located 56 km NW of Susanville, Lassen County, California. Ponderosa pine is the principal tree species with small numbers of Jeffrey pine and incense cedar (*Calocedrus decurrens* (Torr.) Florin). White fir becomes increasingly prevalent at higher elevations. BMEF has two major diameter classes of trees—a scattered overstory of large 300-700 year-old pines and incense cedar and a dense understory of pines and white fir that is thought to have originated after wild fire exclusion (Oliver and Powers 1998). Although some logging was conducted at BMEF during the 1938-1947 as part of the Methods of Cutting study (Dolph and others 1995), much of the area was never logged making it an ideal location to examine the avifauna of a late-seral ponderosa pine forest. The elevation ranges from approximately 1700-1950 m. In 1995, 12 treatment units (each divided into two split-plots) were established at BMEF to examine the effects of different silvicultural treatments on the flora and fauna (Oliver and Powers 1998).

Goosenest Adaptive Management Area (lat. 41°34'N., long. 121°41'W.) is located in Siskiyou County, 2 km SE of Tennant, CA. The vegetation at GAMA is predominantly 80-year-old, second growth ponderosa pine forest. The dominant tree species are ponderosa pine and white fir. Incense cedar and sugar pine (*Pinus lambertiana* Dougl.) are also common on the northwestern portion of the site. The understory is dominated by greenleaf manzanita (*Arctostaphylos patula* Greene), snowbrush ceanothus (*Ceanothus velutinus* Dougl. ex Hook.), and white fir saplings. The topography is relatively flat with no perennial streams. Elevations range from 1500 to 1645 m. Mean annual precipitation ranges from 38 to 51 cm, with most precipitation falling as snow (Anonymous 1996). The average summer and winter temperatures are 15° C and -2° C respectively (Oregon Climate Service 2000). In 1996, 20 40-ha units were established within the GAMA to test different methods of accelerating the development of late-seral ponderosa pine forest structure (Ritchie and Harcksen 1999).

Point Counts

We estimated the relative abundance of birds on the experimental units at BMEF and GAMA using point counts. Surveys were conducted at BMEF from 20 June-24 July 1995, and at GAMA from 7 June-2 July 1996, prior to the implementation of silvicultural treatments at each site. All experimental units were surveyed at GAMA and both split plots of each of the 12 experimental units were surveyed at BMEF. Because the point count surveys on each split plot at BMEF were sufficiently far apart to avoid overlap, the surveys were analyzed separately. Thus there were 24 survey units at BMEF (split plots at BMEF are hereafter referred to as units). Eight points were surveyed on each unit at GAMA and BMEF. Each survey point was at least 100 m from the unit's edge and 200 m from other point count locations. At each site, each unit was surveyed two times by different observers.

Point count surveys started within 15 min of sunrise and continued until all points on a plot were surveyed (generally 2 hrs). Count duration was 8 min and started upon arrival at the point. The horizontal distance to each bird was estimated in 10 m bands to 50 m, 51-75 m, 76-100 m, and > 100 m. To reduce the chance of double counting, only birds detected within 100 m of the observer were used in analyses. Also excluded from analyses were birds detected flying over but not utilizing the unit. We computed an index of abundance for each bird species on each unit by dividing the number of detections on a unit by the number of surveys (2) on each unit. Species richness was computed for each unit by totaling all of the species that were detected in each unit during the two surveys. Bird species were grouped into four guilds based on their foraging behavior (*appendix 1*). We compared the relative abundance of bird species detected at the two locations using Wilcoxon-rank tests and a significance level of 0.05.

Results

The overall bird species composition was similar at the two locations. Of the 51 species detected at both locations, 29 were detected at both GAMA and BMEF, 14 were only detected at BMEF and 8 were only detected at GAMA (table 1). Most of the species that were detected at only one site were rare at the site where they were observed. When rare species (<1 detection/10 surveys) are excluded, 5 species were observed only at BMEF and 5 species were observed only at GAMA and 25 species were observed at both sites. The number of species detected on each unit did not differ between the two sites (*table 1*).

The relative abundance of woodpeckers, bark gleaners, and flycatchers was higher at BMEF while foliage gleaners were more abundant at GAMA (*table 2*). Differences in abundance of individual species were generally consistent with the differences in the abundance of guilds at the sites. Hairy Woodpeckers, Williamson's Sapsuckers, and White-headed Woodpeckers (see *appendix 1* for scientific names of bird species) were more abundant at BMEF than GAMA. In addition, Downy Woodpeckers were only detected at BMEF. Thus, both the density and diversity of woodpeckers was higher at BMEF than GAMA. Among the bark gleaners, Mountain Chickadees, White-breasted Nuthatches, Pygmy Nuthatches, and Brown Creepers were all more abundant at BMEF. Of the foliage gleaners, Cassin's Vireos, Warbling Vireos, Golden-crowned Kinglets, Hermit Warblers, and Western Tanagers were all more abundant at GAMA. The only species whose abundance pattern differed significantly from the overall trend in its guild was the Red-breasted Nuthatch that was more abundant at GAMA than BMEF.

Discussion

The songbird and woodpecker communities at BMEF and the GAMA were similar to other ponderosa pine avifaunas, especially when only considering the common species (Hall and others 1997, Laudenslayer and Balda 1976, Szaro and Balda 1979a, 1979b, Rosenstock 1996, Szaro and Balda 1986, Thomas and others 1979). These core species include Hairy Woodpecker, Steller's Jay, Mountain Chickadee, White-breasted Nuthatch, Red-breasted Nuthatch, Brown Creeper, American Robin, Townsend's Solitaire, Hermit Thrush, Western Bluebird, Cassin's Vireo, Yellow-rumped Warbler, Brown-headed Cowbird, Western Tanager, Dark-eyed Junco, Cassin's Finch, and Pine Siskin. All of these species occur at both BMEF and GAMA. Many of these species, in turn, occur widely throughout many western forest types (e.g., Chambers and others 1989, Raphael and others 1987).

Although the core group of common species was similar at BMEF and GAMA, there were five species that were relatively common at one site and absent at the other. The common species present at BMEF and absent at GAMA were the White-headed Woodpecker, Olive-sided Flycatcher, Western Wood-Pewee, Gray Flycatcher, and Fox Sparrow. The White-headed Woodpecker has been found to be associated with large diameter trees (Booth 1991, Dixon 1995), while the Olive-sided Flycatcher and Western Wood-Pewee are associated with a high density of large snags and forest openings (Hutto and others 1992). Thus the absence of these species from GAMA is consistent with the absence of large trees and snags at the GAMA site. Gray Flycatchers occur at the low elevation units at BMEF and are associated with sagebrush (*Artemisia* spp.) which was absent at GAMA (Sterling 1999). Fox

Table 1—*Number of birds detected (mean ± SE) on point counts (detections/survey) at Blacks Mountain Experimental Forest (BMEF) and Goosenest Adaptive Management Area (GAMA) in northeastern California. P values are given for those comparisons where P<0.05. Scientific names of birds are provided in appendix 1.*

Species	BMEF	GAMA	*P*
Sharp-shinned Hawk	0.04±0.03	0	
Cooper's Hawk	0.08±0.05	0	
Northern Goshawk	0.10±0.07	0	
Red-tailed Hawk	0.10±0.07	0.03±0.03	
Unknown Raptor	0	0.05±0.03	
Mountain Quail	0.02±0.02	0.03±0.03	
Mourning dove	0.06±0.05	0	
Great Horned Owl	0.02±0.02	0	
Unknown Owl	0	0.03±0.03	
Common Nighthawk	0.06±0.05	0	
Calliope Hummingbird	0.02±0.02	0	
Unknown Hummingbird	0.29±0.09	0	0.0026
Williamson's Sapsucker	1.27±0.25	0.15±0.06	0.0002
Downy Woodpecker	0.04±0.03	0	
Hairy Woodpecker	0.98±0.15	0.53±0.13	0.0300
White-headed Woodpecker	0.19±0.07	0	0.0094
Black-backed Woodpecker	0.10±0.06	0.03±0.03	
Northern Flicker	0.88±0.16	0.43±0.11	0.0386
Pileated Woodpecker	0.10±0.05	0.05±0.05	
Unknown Woodpecker	1.33±0.24	0.95±0.18	
Olive-sided Flycatcher	0.19±0.10	0	
Western Wood-Pewee	0.25±0.15	0	
Hammond's Flycatcher	0.52±0.22	0.73±0.26	
Gray Flycatcher	0.23±0.18	0	
Dusky Flycatcher	4.69±0.55	3.88±0.91	
Unknown Empidonax Flycatcher	6.94±0.74	2.23±0.57	<0.0001
Cassin's Vireo	1.96±0.40	3.35±0.67	
Warbling Vireo	0	0.43±0.19	0.0044
Gray Jay	0	0.45±0.22	0.0233
Steller's Jay	0.33±0.13	0.30±0.11	
Common Raven	0.60±0.20	0.23±0.13	
Mountain Chickadee	12.98±0.82	8.18±0.62	<0.0001
Red-breasted Nuthatch	5.79±0.56	7.98±0.62	0.0199
White-breasted Nuthatch	1.25±0.25	0.08±0.04	<0.0001
Pygmy Nuthatch	0.88±0.31	0.05±0.05	0.0102
Brown Creeper	3.75±0.36	0.63±0.19	<0.0001
Golden-crowned Kinglet	1.58±0.31	3.55±0.53	0.0045
Townsend's Solitaire	0.90±0.16	1.00±0.19	
Hermit Thrush	4.58±0.66	6.73±0.68	0.0486
American Robin	2.10±0.37	0.05±0.03	<0.0001
Nashville Warbler	0	6.10±0.76	<0.0001
Yellow-rumped Warbler	8.10±0.55	7.00±0.76	
Hermit Warbler	0	3.68±0.61	<0.0001
Western Tanager	1.94±0.37	5.13±0.67	0.0003
Green-tailed Towhee	0	0.08±0.05	
Chipping Sparrow	2.50±0.49	2.25±0.41	
Fox Sparrow	0.25±0.13	0	0.0177
Dark-eyed Junco	7.83±0.67	4.05±0.53	0.0002
Brewer's Blackbird	0	0.05±0.05	
Brown-headed Cowbird	0.08±0.05	0.43±0.14	0.0275
Purple Finch	0.71±0.31	0.03±0.03	0.0049

Species	BMEF	GAMA	*P*
Cassin's Finch	0.42±0.20	0.18±0.10	
Red Crossbill	0.06±0.06	0.38±0.17	0.0228
Unknown Carpodacus finch	0.88±0.30	0.65±0.25	
Pine Siskin	0.08±0.07	0	
Evening Grosbeak	0	0.28±0.13	0.0044
Black-headed Grosbeak	0	0.03±0.03	
Unknown	1.33±0.30	0.18±0.07	0.0013
Species Richness	18.88±0.56	18.05±0.47	

Table 2—*Number of birds (mean ± SE) in different foraging guilds detected on point counts (detections/survey) at Black's Mountain Experimental Forest (BMEF) and Goosenest Adaptive Management Area (GAMA) in northeastern California. P values are given for those comparisons where P<0.05.*

Guild	BMEF	GAMA	P
Flycatchers	12.81±0.70	6.83±1.33	0.0014
Foliage	13.58±1.06	23.15±1.25	0.0001
Bark Gleaners	11.67±0.77	8.73±0.70	0.0049
Woodpeckers	4.90±0.54	7.00±0.76	<0.0001

Sparrows are more common at elevations higher than those found at GAMA (Weckstein and others 2002).

Three of the common species present at GAMA but absent at BMEF were species that are generally associated with denser, more closed canopy forest. These include the Warbling Vireo, Gray Jay, and Hermit Warbler. The other two species either occur erratically across the landscape (Evening Grosbeak) or are associated with extensive shrub patches (Nashville Warblers) that were not present at BMEF.

Despite differences in the species composition of the bird communities at the two sites, bird species richness did not differ. Thus, the presence of species associated with large-diameter trees at BMEF and those associated with a more closed forest canopy at GAMA resulted in no difference in the number of species detected on units at the two locations. Thus, the presence of large trees and an associated more open canopy does not lead to an increase in the number of bird species. This underscores the point that bird species richness (or diversity) at a particular site should not be used as a measure of the success of restoration efforts in ponderosa pine ecosystems.

One of the most consistent differences in the bird communities between BMEF and GAMA was the greater abundance of woodpeckers and bark gleaners at BMEF. Woodpeckers as a group and several species individually were more common at BMEF including the Williamson's Sapsucker, Hairy, and White-headed Woodpeckers, and the Northern Flicker. White-breasted Nuthatch, Pygmy Nuthatch, and Brown Creeper, all bark gleaners, were more common at BMEF than they were at GAMA. Mountain Chickadees, which do feed as bark gleaners, but also in the foliage of trees, were also more common at BMEF. A strong exception to the above pattern is found with the Red-breasted Nuthatch, which is more common at GAMA. Red-breasted Nuthatches rarely feed on the tree bole, in contrast to the other bark gleaners in this guild (e.g., Adams and Morrison 1993). In addition, Red-breasted Nuthatches typically fed on medium-diameter, not large-diameter trees in the Sierra

Nevada (Airola and Barrett 1985) but the opposite was found in coastal Douglas-fir forests (Weikel and Hayes 1999).

Other species were more common at BMEF included hummingbirds (often unidentified to species), American Robins, Dark-eyed Juncos, and Purple Finches. Robins and Juncos, both ground foragers, seemingly take advantage of the generally more open forest structure at BMEF.

Foliage gleaners were consistently more abundant at GAMA than BMEF. Warbling Vireos, Golden-crowned Kinglets, Hermit Warblers, and Western Tanagers, all foliage gleaners, were more abundant at GAMA. Foliage gleaners may be favored at GAMA because of the high encroachment of white fir and associated high canopy cover and foliage volume at the site (Ritchie and Harcksen 1999). Other species that were more abundant at GAMA relative to BMEF include the Hermit Thrush, Brown-headed Cowbird and Red Crossbill. Hermit Thrushes are generally associated with higher elevation fir forests (Hejl and Verner 1988) and may also be responding to the high level of white fir encroachment at GAMA. Higher densities of Brown-headed Cowbirds at GAMA may be related to the higher densities of potential hosts for this brood parasite, particularly among warbler species. Red Crossbills are highly erratic in time and space across all forest types, and so may not indicate any real differences between sites.

The greater abundance of flycatchers at BMEF is difficult to explain because many of the flycatchers could not be identified to species at both sites (*table 1*). It is clear, however, that the species richness of flycatchers was much higher at BMEF as 3 of the 5 species were only detected there. Olive-sided Flycatchers responded strongly to differences in the abundance of flying arthropods between burned and unburned sites in north central California (Meehan and George 2003). Furthermore, the abundance of flying arthropods was positively associated with the cover of ground vegetation, down wood, and shrubs and saplings. It would be interesting to examine whether differences in ground vegetation, down wood and the associated arthropod fauna between BMEF and GAMA explains differences in flycatcher abundance between the two sites.

Despite the differences in abundance in some species between the two sites, it is important to recognize that several common species occur at similar densities. For example, Chipping Sparrow, Yellow-rumped Warbler and Cassin’s Finches are similarly common at both sites. Chipping Sparrows are ground-feeding birds, while Cassin’s Finches are often detected in the high canopy feeding on cones. Yellow-rumped Warblers are primarily foliage gleaners and therefore the lack of difference in abundance between the two sites is not consistent with overall greater abundance of foliage gleaners at GAMA.

The main differences between the ponderosa pine forest sites are the higher densities of bark gleaners at BMEF in comparison to the higher densities of foliage gleaners at GAMA. Large-diameter trees at BMEF mean more trunk surface to forage upon. Hairy and Black-backed Woodpeckers at BMEF often feed on larval bark beetles in snags, while White-headed Woodpeckers mostly feed on large living ponderosa pines (Hughes 2000). Large-diameter trees were preferred foraging substrates for Hairy Woodpeckers and Chestnut-backed Chickadees (*P. rufescens*) by Weikel and Hayes (1999) in their coastal study noted above. Further, they found that Brown Creepers selected trees with deeper furrows in the bark (Weikel and Hayes 1999). We have no measured bark furrow depth.

The forest at GAMA includes many small- and medium-diameter white fir, and this may provide a basis for the high densities of leaf gleaners. Brawn and Balda (1988) proposed that high densities of foliage gleaning warblers and tanagers in ponderosa pine forests was due to more productive foliage of intermediate-aged trees. Airola and Barrett (1985) followed insect gleaning birds in Sierra Nevada mixed-conifer forests (with oak) and found that Hermit Warblers preferred pine, whereas Golden-crowned Kinglets, Yellow Warrblers, and Western Tanagers preferred white fir. Cassin's Vireo showed no such preference.

Finally, we should also note that bark gleaners are cavity nesters, and foliage gleaners are cup nesters. Thus, it is possible that the driving difference between BMEF and GAMA bird densities is not tree diameter but rather is cavity availability. We have found that cavities, and occupied cavities, are much more common at BMEF compared to GAMA (Zack and others 2002), but we have no way to measure whether substrate for cup nesters is higher at GAMA. Brawn and Balda (1988) consider the paucity of snags with cavities to be a major reason for the hypothesized declines in chickadee, nuthatch and bluebird (*Sialia* spp.) populations.

This comparison of bird communities at our two sites provides a baseline for understanding the ongoing experimental treatments at both BMEF and GAMA. Sallabanks and others (2000), in a review of bird response to timber harvest, noted that only a minority of studies were the result of experimental treatments and most were not long term. The result is that most studies do not truly address cause and effect issues nor do they address the relationships of birds to treatments as the vegetation develops and changes following treatment. This study is indeed correlational, and only a comparison of patterns from one year at each site. But it is a foundation of ongoing evaluation of these avifaunas with experimental silvicultural treatments. While the details of the treatments differ (see Zack and others 1999, Ritchie and Harcksen 1999), both sites have plots that were thinned, and plots that were burned. Monitoring avifauna use after these treatments will allow us to more clearly understand broad differences in the responses of both bark gleaners and foliage gleaners.

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Effects of Site on the Demographics of Standing Dead Trees in Eastside Pine Forests[1]

William F. Laudenslayer, Jr.[2]

Abstract

In the last several decades there has been a growing awareness of the value of snags in forests. Recent work has resulted in management standards and guidelines that set objectives for species such as cavity-nesting birds, but the numbers of snags required may not be attainable or sustainable across the landscape. Work reported here on snag demography in eastside pine forests was initiated in 1989. While snags have been present on most of the sites throughout the study period, the number of snags and their "life-spans" depends on tree species, tree size, soil characteristics, and topography. Snags persisted for longer periods at Lassen Volcanic National Park sites where the soils are covered by up to 0.5 m of volcanic ash, but snag density was less than our other sites because of the low tree density. These findings suggest that standards and guidelines for snag management incorporate variation at local scales.

Introduction

Availability of standing dead trees, or snags, has long been a management issue in forests (Brawn and Balda 1988, Bull and others 1997, Cline and others 1980, Raphael and Morrison 1987, Raphael and White 1984, Rosenberg and others 1988, Swallow and others 1988, Thomas and others 1979, Waters and others 1990, Welsh and Capen 1992, Zack and others 2002), particularly in eastside pine forests – those forests dominated by ponderosa (*Pinus ponderosa* P. & C. Lawson) or Jeffrey pine (*P. jeffreyi* Grev. & Balf.) lying east of the crest of the Cascades and Sierra Nevada in California, Oregon, Washington, and Nevada (Studinski and Ross 1986). These concerns have resulted in management standards that are applied across large landscapes. For example, the Lassen National Forest requires 1.2 snags per acre (3 per ha) between 15 and 24 inches (38-61 cm) diameter at breast height (dbh) and 0.3 snags per acre (0.7 per ha) in excess of 24 inches (61 cm) dbh and 20 feet (6 m) in height (Lassen National Forest 1993).

While these standards may be relevant (but see Oliver 2002), there is such a wide variety in landscape variables in eastside pine forests, such as elevation, precipitation, soil composition, soil depth, and water holding capacity, as well as existing vegetation composition and structure, and historical vegetation trajectories, that the standards may not be applicable to certain areas or spatial and temporal

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scales. Gaining an understanding of snag demographics in eastside pine forests that differ in composition and site conditions is critical to evaluating the efficacy of the snag standards and guidelines for these forests. Variation observed should also help focus research and management attention to the processes underlying snag recruitment and loss rather than a forest's capacity to maintain a specific number of snags.

Eastside pine forests were burned frequently prior to about the turn of the beginning of the 20th century. Since then, fire has generally been excluded. Our knowledge of snag processes in these forests is from this long fire-free period and is unlikely to accurately reflect snag processes under frequent fire regimes. Changes in fire management practices coupled with the increased use of prescribed fire as a forest management tool will likely influence snag longevity.

The objectives of this paper are to (1) illustrate some of the variation in snag recruitment, loss, and occupancy in four relatively undisturbed eastside pine stands (stands that have not seen wildfire or lost large trees to timber harvest) and (2) discuss the ramifications of this variation relative to the snag management standards that are applicable to eastside pine forests.

Methods

The study sites discussed in this paper were established during a larger study of 24 sites (initiated in 1988) to evaluate the responses of birds to snag density and subsequent snag dynamics in eastside pine forests (see Landram and others 2002, Laudenslayer 2002). The four study sites were chosen because they had little historic management and reflected two different patterns of snag demography.

Snags on each study site were counted, mapped and their condition evaluated on 5-ha (12.4-ac), 100 x 500m (328 x 1640 ft) strip plots annually from 1989 to the present. All snags greater than 15 cm (6 in) dbh were included in the snag inventory. For this analysis, snags were divided into two groups. Small snags were those with diameters between 15 and 38 cm (6 and 15 in) dbh and with any height, or greater than 38 cm (15 in) dbh but less than 6 m (20 ft) tall. Large snags were those with diameters greater than 38 cm (15 in) dbh and height greater than 6 m (20 ft). The dividing line, 38 cm (15 in), separates those snags targeted by various snag retention standards (usually greater than 38 cm [15 in] dbh) from those smaller snags that are not generally considered for retention. Snag heights and diameter were also measured annually. Snags remained in the inventory until they were completely down (uprooted or broken at ground level) or had less than 1 m of bole remaining.

Study Sites

The four study sites chosen for this report were located in northeastern California on the Modoc and Lassen National Forests and within Lassen Volcanic National Park *(fig. 1)*. All sites were dominated by either ponderosa or Jeffrey pine and had little to no active management activity except fire exclusion.

Old Forest

The Old Forest study site was located within Blacks Mountain Experimental Forest (within the Lassen National Forest) (lat. 40°42'45" N, long. 121°9'45" W) between 1730 and 1740 m (5676 and 5709 ft) in elevation, and was primarily ponderosa pine with small numbers of white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex. Hildebr) and incense cedar (*Calocedrus decurrens* (Torr.) Florin). It is a rather open forest dominated by small numbers of widely-spaced large diameter trees (>61 cm [> 24in] dbh). The understory was occupied by grasses, forbs, shrubs, and small trees. The only manipulations included a thinning and prescribed burn in 1996 (Oliver and Powers 1998, Oliver 2000).

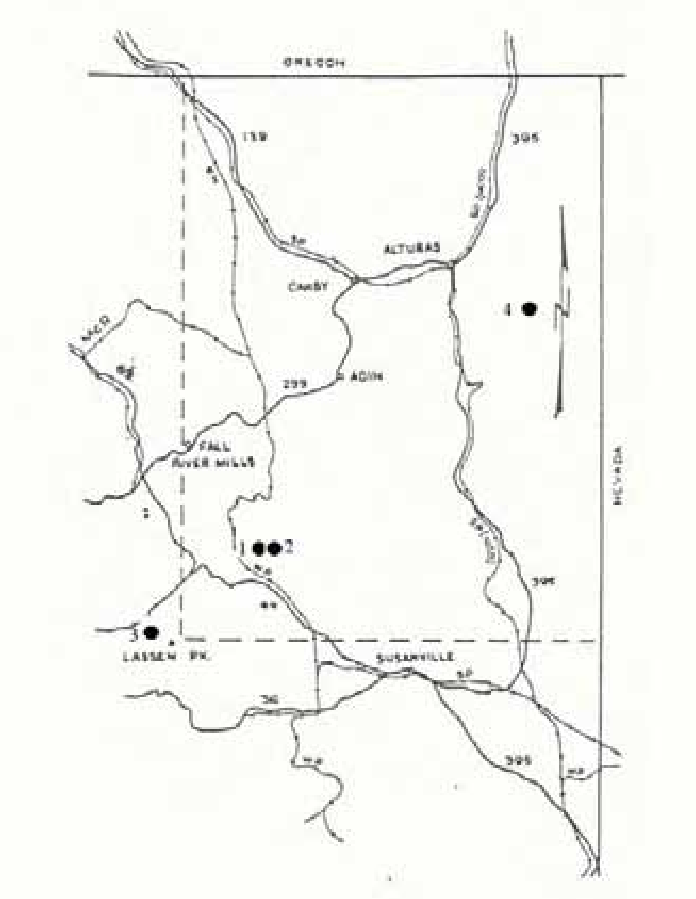

Figure 1—Four snag study sites in northeastern California: 1 = Old Forest; 2 = Research Natural Area; 3 = Hot Rock; and 4 = Soup Creek.

Research Natural Area

The Research Natural Area, also located within Blacks Mountain Experimental Forest (lat. 40°42’30” N, long. 121°8’15” W), was dominated by ponderosa pine and had a considerable amount of white fir and incense cedar in the understory. It lies between 1820 and 1830 m (5971 and 6004 ft) in elevation. White fir likely has been a natural component at this site for at least the last century. This site had the greatest canopy closure of the four study sites but rarely exceeds 50 percent, except in dense thickets of fir reproduction. The forest was dominated by relatively large diameter trees (>61 cm [>24 in] dbh), snag recruitment is high, and the numbers of large live trees is declining. Under the large trees are thickets of small trees, generally white fir with smaller amounts of incense cedar, and openings occupied by grasses, forbs, and shrubs.

Hot Rock

Hot Rock is located within Lassen Volcanic National Park (lat. 40°32'15" N, long. 121°29'30" W) and was dominated by Jeffrey pines with a small amount of white fir, generally in the understory. The elevation is about 1890 m (6200 ft). This site has the most open canopy of the four and the openness of the site increased through the study period. This site supported a noticeable number of snags that diameter appear to have been recruited in the 1970s. The forest was dominated by large trees (>61 cm [>24 in] dbh) but snag recruitment was high and the numbers of large live trees was declining. Generally, few young trees, shrubs, grasses, or forbs appeared in the understory, likely because of the thick layer of volcanic ash – up to 0.5 m in depth in some areas. However, where mass mortality occurred in the mid 1990s abundant fir regeneration has since established.

Soup Creek

Soup Creek is located in the Warner Mountains on the Modoc National Forest (lat. 41°18'0" N, long. 120°18'0" W) and is dominated by ponderosa pine with white fir both in the overstory and understory. The average elevation is about 1890 m (6200 ft). This site is on a steep slope in a canyon. The site includes several large aggregations of boulders, occupying as much as 0.5 ha (1.2 ac) of the site. The forest is dominated by large diameter trees (>61 cm [>24 in] dbh) but the density of large trees declines as one moves upslope. Past timber harvesting occurred on about 10 percent of the site.

Results

Most snags in the overstory were pines, but white fir comprised most of the snags in the understory of all sites, except Old Forest where firs were nearly absent. Numbers of snags increased substantially in two of these study sites, Research Natural Area and Hot Rock, and generally increased slightly in Old Forest and Soup Creek. Snag numbers increased starting in the early 1990s and generally ending about

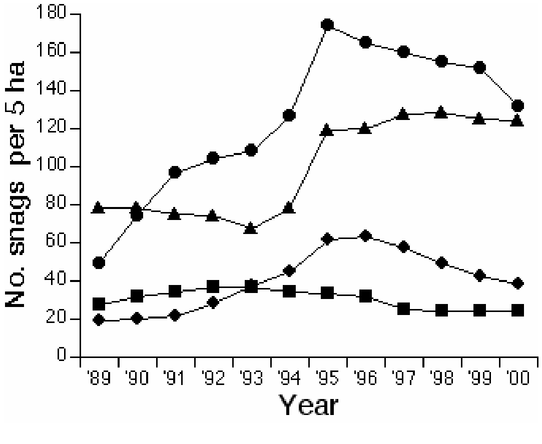

Figure 2—Numbers of snags greater than 15 cm dbh in each of four 5-ha eastside pine study sites over a 10-year period: Old Forest = squares; Research Natural Area = circles; Hot Rock = triangles; and Soup Creek = diamonds.

1996. Since then, snag numbers have declined at Old Forest, Research Natural Area, and Soup Creek (*fig. 2*). Numbers of large snags also increased substantially in all sites but Old Forest. At Hot Rock, large snag recruitment increased substantially in 1995. Large snag recruitment was greatest in the early 1990s and declined as the decade progressed. At the end of the decade, the number of large snags was beginning to decline in all but Old Forest (*fig. 3*).

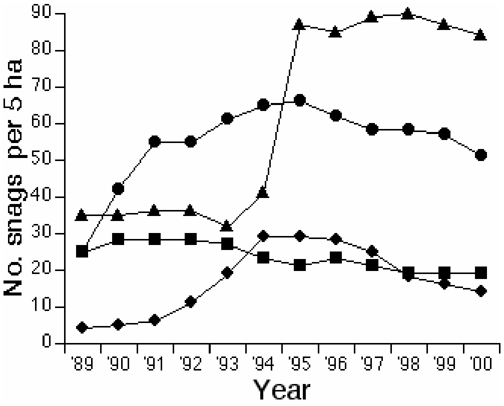

Figure 3—Numbers of snags greater than 38 cm dbh and 6 m in height in each of four 5-ha eastside pine study sites over a 10-year period: Old Forest = squares; Research Natural Area = circles; Hot Rock = triangles; and Soup Creek = diamonds.

At Old Forest, recruitment of large snags increased in 1990 and 1996 and recruitment of smaller snags was greatest in 1991 and 1997. Many large snags were lost in 1994 and 1997, whereas small snag losses were concentrated in 1996 and 1997. In high recruitment years only 3 - 5 snags were recruited and in high loss years 5 - 7 snags were lost (*fig. 4*). As a result, numbers of snags at this site changed little during this study period.

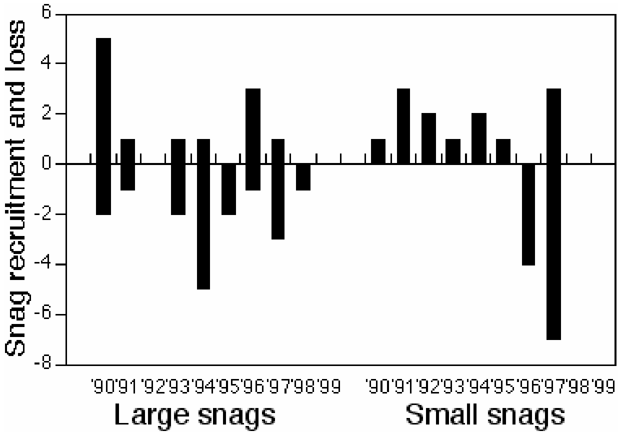

Figure 4—Recruitment and loss of large (greater than 38 cm dbh and greater than 6 m in height) and small snags (15-38 cm dbh and any height, and greater than 38 cm dbh and less than 6m in height) in the 5-ha study site at Old Forest.

At Research Natural Area, the best years for large snag recruitment were 1990 and 1991, but there was some recruitment from 1994-1996. For small snags, 1995 and to a lesser extent 1994 were high recruitment years. Relatively few large snags were lost during the observation period, with 1996 having the greatest loss. For small snags, however, 1996 and 1997 were high loss years. In contrast to Old Forest, Research Natural Area gained approximately 4 times as many large snags as it lost in a high recruitment year, and perhaps 15 times as many small snags (*fig. 5*). The trend at this site is characterized by high snag recruitment and declining numbers of large live trees.

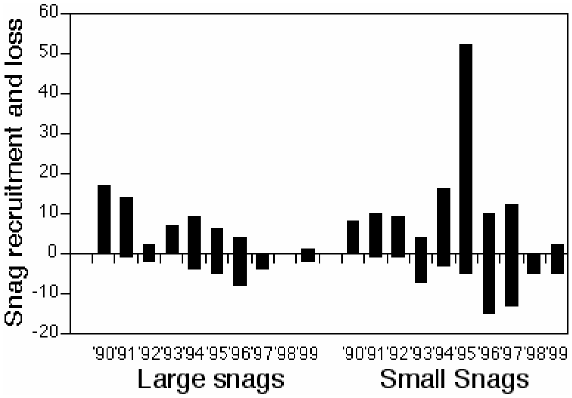

Figure 5—Recruitment and loss of large (greater than 38 cm dbh and greater than 6 m in height) and small snags (15-38 cm dbh and any height, and greater than 38 cm dbh and less than 6m in height) in the 5-ha study site at Research Natural Area.

At Hot Rock, the years of highest recruitment for large snags were 1994 and 1995. In 1995, all of the large live trees in approximately 0.05 ha (0.12 ac) died. Recruitment of small snags was low, with the largest recruitment occurring in 1997. Relatively few large snags were lost in this decade, and the loss was distributed over a number

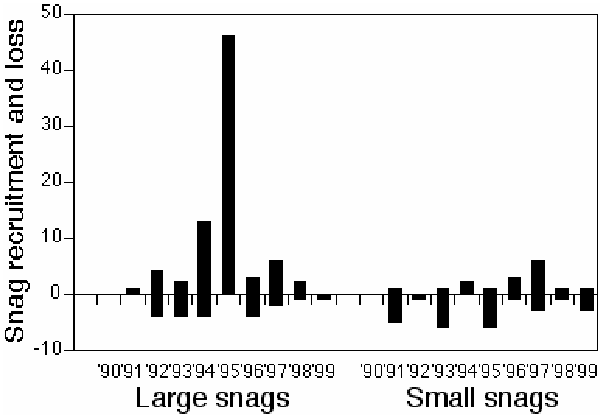

Figure 6—Recruitment and loss of large (greater than 38 cm dbh and greater than 6 m in height) and small snags (15-38 cm dbh and any height, and greater than 38 cm dbh and less than 6m in height) in the 5-ha study site at Hot Rock.

of years (1992-94 and 1996). Losses of small snags were distributed over several years (1991, 1993, and 1995). Hot Rock recruited the majority of snags, mostly large snags, in one year (1995) and without that large pulse of recruitment, the general magnitude of recruitment and loss would resemble Old Forest more than Research Natural Area (*fig. 6*).

At Soup Creek, 1992-1995 were high years for large snag recruitment. Recruitment of small snags was high in 1995 and 1996. Large snags were lost at a greater rate in 1995 and 1998. Small snags were lost at a relatively high rate from 1992 through the rest of the decade (*fig. 7*).

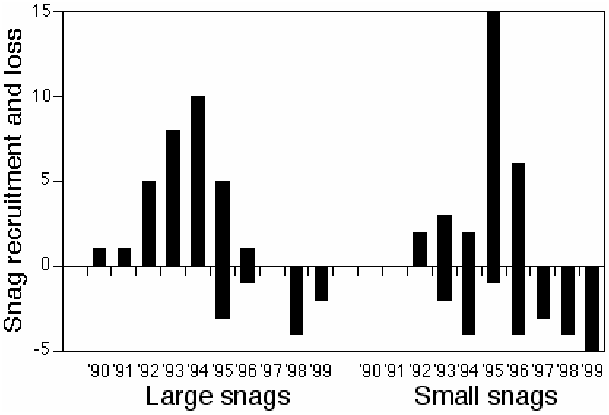

Figure 7—Recruitment and loss of large (greater than 38 cm dbh and greater than 6 m in height) and small snags (15-38 cm dbh and any height, and greater than 38 cm dbh and less than 6m in height) in the 5-ha study site at Soup Creek.

Discussion

Historically, management standards and guidelines for snags in eastside pine forests were based on numbers of snags with different characteristics. The standards and guidelines also often specified a certain number of live trees for future snag recruitment (Studinski and Ross 1986). The principal purpose for retaining and recruiting snags was to provide nesting sites for primary and secondary cavity nesters. Studies of snag densities in eastside pine forests (Dahms 1949, Keen 1929, 1955, Landram and others 2002, Zack 2002) suggest that many sites have low numbers of snags relative to other forest types, and that even better sites may not be able to meet the standards and guidelines for snag numbers in a sustainable manner.

However, snag density may not be as important to the system as ensuring recruitment and a continuous flow of snags through time. *Picoides* woodpeckers generally forage for only a few years on dying or freshly dead trees and prey especially on scolytid, buprestid, and cerambycid larvae (Farris and others 2002, Hughes 2000). Insect residence and thus woodpecker foraging occurs early in the life of a snag; the half-life of a ponderosa pine snag in northeastern California is between 7 and 10 years (Landram and others 2002). Primary tree-killing bark beetles [e.g., fir engraver (*Scolytus ventralis* LeConte), mountain pine beetle (*Dendroctonus ponderosae* Hopkins), and western pine beetle (*D. brevicomis* LeConte)] generally persist in a dying or dead tree for only 60 days to less than a year, depending on the time of attack, whereas the longer developing beetles, like California prionus

(*Prionus californicus* Motschulsky) may be present for 3 to 5 years (Furniss and Carolin 1977).

Tree mortality patterns reported here indicate that even at the relatively fine scale of 5 ha (12.4 ac), most plots had a flow of dying and dead trees to replace those lost. However, many of the recruited snags were of small diameter (less than 38 cm [15 in] dbh), which are unlikely to support the diversity or abundance of beetle larvae found in larger diameter snags. The temporal pattern of mortality suggests that much of the mortality in the early 1990s was related to the drought coupled with bark beetle attack. However, beetles may successfully attack trees that are overmature, are adjacent to windfalls, have root-rot, or are in overstocked or stagnant stands (Cobb and others 1974, Furniss and Carolin 1977, Oliver 1995), leading to some constant rate of background mortality even when moisture is not limiting.

Tree mortality generally consisted of single trees or small groups of trees scattered throughout each site with the exception of the large mortality event at Hot Rock. This event was largely confined to an area of about 0.75 ha (1.9 ac), within which all of the large pine trees died. Oliver (1995) reported that there may be mortality thresholds related to the basal area present and the growing capability of a site, so this mortality event at Hot Rock may have been driven by overstocked conditions. Since this site is near Mount Lassen, other less well known causes of mortality may be responsible, such as excessive carbon dioxide gas in the soil (Sorey and others 2000).

Generally, snag loss occurred in all years, but was punctuated with pulses that followed pulses of recruitment. Fall rates for snags in northeastern California vary from seven percent per year for ponderosa pine and Jeffrey pine to four percent per year for white fir (Landram and others 2002). Snag half-life is related to species and diameter. For small (13-38 cm [5-15 in] dbh) ponderosa or Jeffrey pines, half-life ranged from five to six years whereas for small white firs, the half-life was seven years. For larger (greater than 38 cm [15 in] dbh) ponderosa or Jeffrey pines, half-life was eight and seven years whereas for larger white firs, half-life was 10 years (Landram and others 2002).

Soil characteristics may also be important in controlling tree stocking, snag recruitment, and snag longevity. At Old Forest, Research Natural Area, and Soup Creek, there were areas greater than 100 m^2 (1076 ft^2) in size where trees were not present because of soil characteristics, soil depths, presence of rock, or some other factor. At Hot Rock, much of the site was covered by a layer of volcanic ash averaging about 0.5 m (1.6 ft) in depth (Clynne and others 2000). This thick layer of ash probably made it more difficult for young trees to get established and may have contributed to greater snag longevity by not retaining moisture close to the tree base, thereby reducing the rate of decay. Within the area experiencing very high mortality at Hot Rock, there were very few small trees present prior to the event. Following the event, a “carpet” of seedlings became established; however, most of the seedlings were white fir, suggesting that the tree species composition within this area may be shifting toward white fir in the foreseeable future. Further, it will take decades for these tree cohorts to grow to sufficient size to meet snag guidelines. In the interim, about 0.75 ha (1.8 ac) will shortly lack snags. However, snags may persist longer on these Lassen Volcanic Park sites than the other sites, likely because of the volcanic ash layer. At Hot Rock, trees that died in the bark beetle outbreak in the mid-1970s remain standing.

Conclusions

Snag recruitment and loss are likely related to local site characteristics and vegetation structure interacting with weather, climate, insects, and disease. Considerable variation in recruitment and loss occurs across the landscape even in sites that have not been managed or seen wildfire in decades. This variation suggests that standards and guidelines for snag retention should incorporate information from local scales rather than regional or larger scales. Work reported here suggests that processes contributing to snag recruitment and fall down operate at spatial and temporal scales that are highly variable. Thus management guidelines may need to reflect this variability.

While current numbers of snags are indeed important, managing for a suitable population of future snags is more critical to the continued functioning of forests over time. If forests are composed of appropriate sizes of living trees, then drought, insect and disease, and fire will create snags in an appropriate pattern.

This study focused on forests that have burned frequently under natural disturbance regimes. Reintroduction of prescribed fire, and more frequent occurrence of wildfire, will substantially alter forest structure, and therefore the snag component of forests as well.

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Woodpecker-snag interactions: an overview of current knowledge in ponderosa pine systems[1]

Kerry L. Farris[2] and Steve Zack[3]

Abstract

Standing dead trees (snags) with cavities are a critical ecological component of western coniferous forests. These structures provide foraging, roosting, and nesting habitat for numerous species of invertebrates, amphibians, reptiles, birds, and mammals. Snags may be created through a variety of interrelated processes including wildfire, drought, insects and disease. However, dead trees containing excavated cavities are primarily the result of nest excavation by woodpeckers. While the specific factors leading to cavity generation in certain snags is not well understood, the manner in which a tree dies likely plays a significant role. We provide an overview of woodpecker-snag interactions in relation to the major modes of tree mortality in ponderosa pine. Of particular interest is the effect of mortality agent on the temporal patterns of snag decomposition, woodpecker foraging use, and woodpecker cavity excavation. Generally, snags created by bark beetles, and/or fire decay fastest, and experience the greatest foraging and nesting use by woodpeckers. Consideration of these interrelationships may aid in snag management.

Introduction

Standing dead trees (snags) are important to the structure and function of western coniferous forests (Bull and others 1997, Harmon and others 1986, Thomas and others 1979). These structures provide critical habitat for numerous organisms, contribute to nutrient cycling, and influence forest productivity (Harmon and others 1986). Dead trees containing excavated cavities are of particular importance to many species of snag-dependant wildlife and are a primary focus of forest management guidelines in the western coniferous forests (Bull and others 1997, Thomas and others 1997). Snags may be created through a variety of interrelated processes including wildfire, drought, insects and disease; however snags containing cavities are primarily the result of nest excavation by woodpeckers. Once abandoned, woodpecker cavities provide important foraging, roosting, and nesting sites for numerous species of birds, mammals, reptiles, amphibians, and invertebrates (Thomas and others 1979, Bull and others 1997).

The factors that determine which snags are selected for cavity excavation by nesting woodpeckers are complex and poorly understood. Several authors have

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suggested that the quality of snags, as determined by the size and type of decay, may be more important than the quantity of snags on the landscape when determining the value of habitat for cavity-nesting wildlife (Bull and others 1997, Conner and others 2001, Jackson 1977, Jackson and Jackson 2004). In ponderosa pine forests, Zack and others (2002) found that fewer than 20 percent of pine and fir snags sampled in northeastern California contained nest cavities. Additionally, Ganey and Vojta (2004) reported that only 17 percent of snags contained cavities in ponderosa pine forests of Arizona. These figures suggest current management guidelines which focus solely on snag quantity (e.g. snags/hectare) may not adequately reflect habitat requirements of nesting woodpeckers.

Ponderosa pine (*Pinus ponderosa* P.& C. Lawson) is a particularly important snag species for cavity-nesting birds due to its relatively large volume of sapwood, which often decays quickly and provides suitable sites for nest excavation (Bull and others 1997). In mixed-conifer stands, ponderosa pine is often preferred over other tree species by nesting woodpeckers (Bull and others 1997, Bate 1995, Dixon 1995, Haggard and Gains 2001, Hutto 1995, Lehmkuhl and others 2003, Zack and others 2002). Moreover, recent research in ponderosa pine suggests that the foraging activity of woodpeckers during the first few years of snag's life may influence subsequent decomposition of the sapwood by promoting decay organisms associated with nest cavity excavation (Farris and others 2004). While the snag requirements of nesting woodpeckers have been studied widely, relatively little is known about the structures associated with foraging (Bull and others 1997, Conner and others 1994, Steeger and others 1995), which may prove to be instrumental to the creation of future nesting habitat. Foraging woodpeckers respond to snags originating from various modes of mortality, and the manner in which a tree dies likely has a large influence on its eventual use by woodpeckers and subsequent wildlife species.

The objective of this paper is to provide an overview of woodpecker foraging and nesting behavior as it relates to snags originating from each of the major mortality agents in ponderosa pine systems. We will begin with a brief outline of how foraging and nesting woodpeckers respond to snags created by wind, lightning, and diseases. This will be followed by a more detailed discussion of responses of woodpeckers to fire and bark-beetles, two mortality agents that have been extensively studied. Finally, we will discuss woodpecker use of artificially created snags, as these structures are becoming a common management tool in some forests. Of particular interest throughout our discussion is the effect of each mortality agent on the timing of woodpecker use and snag attrition.

Mortality Agents

There are numerous mortality agents that act to create snags in ponderosa pine forests. Major agents of tree mortality in forested ecosystems have been outlined by Harmon and others (1986) and Rose and others (2001) and include: wind, fire, insects, diseases, competitive suppression, senescence, flooding, landslides, lightening, and volcanic events. Distinguishing between proximate and ultimate causes of tree death can be problematic and it is likely that mortality agents rarely act independently of one another. For example, fires can weaken trees both structurally and physiologically leading to windthrow, insect attack, or disease inoculation. For the purposes of this review, we will treat each mortality agent independently.

Wind

Wind events can kill trees by uprooting and snapping the bole, or breaking large branches. This agent can act on large scales during events such as severe wind storms or hurricanes, or on a smaller scale by killing single or small clusters of trees. In western North America, wind events are typically severe and contribute to a larger proportion of snag creation on the west side of major mountain ranges such as the Cascades or Sierra Nevada, or in coastal forests where as many as 70 percent of the stems may be killed (Greene 1984, Harmon and others 1986). In contrast, only 15-20 percent of trees are killed by wind in interior ponderosa pine forests (Avery and others 1976). Depending on the distribution of tree sizes in the affected stand and subsequent insect colonization after wind events, the resultant snags and downed logs may be useful for woodpeckers as both foraging and nesting sites. Wichmann and Ravn (2001) documented large influxes of Scolytid beetles following a windthrow event in spruce forests. Wickman (1965) reported both foraging and nesting use of wind damaged ponderosa pine by Black-backed Woodpeckers (*Picoides arcticus* Swainson) foraging on Cerambycid beetles in northeastern California.

Lightning

Lightning can kill trees by shattering or exploding the tops or entire boles. It can also damage trees, resulting in a narrow furrow of disturbed bark and sapwood spiraling around the bole from the top to the butt of the tree (Scharpf 1993). This often results in dead and decaying wood which may be colonized by insects (Bull and others 1997), subsequently attracting foraging woodpeckers (Bull and others 1997). Additionally, these pockets of dead wood can serve as cavity sites for nesting woodpeckers (Bull and others 1997).

Disease

There are numerous diseases which affect ponderosa pine. Scharpf (1993) outline 7 major categories: needle diseases; cankers; diebacks and galls; rusts; mistletoes; root diseases; and rots. The responses of foraging and nesting woodpeckers to trees with these diseases present, or snags created by them, is largely unknown. From the perspective of woodpecker use perhaps the greatest influence these diseases may have is in weakening and/or killing trees which facilitates the invasion of bark and/or wood-boring beetles, the major prey of foraging woodpeckers. Trees weakened or killed by disease may also provide nesting habitat depending on internal decay patterns.

Insect Mortality

There are numerous species of bark beetles which act as mortality agents in ponderosa pine systems. Most common are members of the genera *Dendroctonus* and *Ips*. These insects typically attack weakened or suppressed trees, but at epidemic levels, they can overcome even healthy trees, resulting in patches of mortality that contribute to both vertical and horizontal diversity in forest structure. These beetles are an important source of prey for many woodpecker species: particularly those of the genus *Picoides*. Woodpeckers of this genus respond dramatically to outbreaks of these insects and have been suggested as control agents for endemic insect populations. For instance, Murphy and Lehnhausen (1998) noted a four fold increase in Hairy (*P. villosus* Linnaeus), Three-toed (*P. dorsalis* Baird), and Black-backed Woodpeckers (*P. arcticus* Swainson) following a bark and wood-boring beetle outbreak in Alaska. Several authors have suggested that woodpeckers may be important regulators of bark beetle populations: especially at endemic levels (Kroll and Fleet 1979, Kroll and others 1980, Koplin and Baldwin 1970, Koplin 1972).

Picoides woodpeckers forage most intensively on ponderosa pine infested by bark-beetles during the first 1 to 3 years after invasion (Farris and others 2002). Bark beetles of the genus *Dendroctonus* are the primary prey items of woodpeckers during the first year following tree death (Farris and others 2002, Shea and others 2002). During the second and third years following the tree's demise, phloem and xylem consuming insects of the families Buprestidae and Cermabycidae provide additional prey for woodpeckers (Farris and others 2002, Shea and others 2002).

Ponderosa pine killed by bark beetles, once adequately decayed, may also serve as nesting sites for woodpeckers. Little information exists examining the nesting preferences of woodpeckers in ponderosa snags specifically originating from bark beetle activity; however it is likely that many of the snags examined in several studies were influenced by bark beetles. In general, woodpeckers nesting in ponderosa pine select snags that are larger in diameter and of advanced decay (Bull and others 1997, Bate 1995, Dixon 1995).

Recent evidence suggests a potential relationship between nest excavation in beetle-killed ponderosa pine and previous foraging use by woodpeckers during the early stages of snag decay. Zack and others (2002) determined that nest excavation was more likely on snags with an extensive history of foraging activity by both beetles and woodpeckers. Additionally, Farris and others (2004) documented the influence of foraging woodpeckers on the decomposition of ponderosa pine. Snags that were used by foraging woodpeckers experienced significantly greater sapwood decay than unused snags. Additionally, more than half the woodpeckers sampled in this study carried wood-decaying fungi on their beaks. These findings suggest that woodpeckers may influence snag decomposition through their foraging behavior that can structurally degrade the wood and passively introduce the wood decaying fungi required for subsequent nest cavity excavation.

Snag longevity following insect attack is variable and is likely dependent on site-specific factors such as soil characteristics, snag size, and local microclimate (Landram and others 2002; Laudenslayer, this volume). On average, the half-life (amount of time it takes for half of the population to fall) of ponderosa and Jeffrey pine snags in the southern Cascades of California is 5 to 6 years for small (13-36 cm [5.2-14.3 in] dbh) snags and 7 to 8 years for large (> 38 cm [15 in] dbh) snags (Landram and others 2002). However, the authors don't distinguish between fall rates of snags of various origins (e.g. fire versus beetle kill). Mitchell and Preisler (1998) found that the half-life of lodgepole pine killed by the mountain pine beetle in central Oregon to be 8 years, with snags falling as early as 3 years after death.

In summary, the temporal patterns of snag use by woodpeckers following beetle kill events are concentrated within the first 1 to 8 years. Foraging is most intense within the first 1-3 years after tree death, while nesting activity begins around year 5. Snags begin to fall during this same period and may be an indication that snags killed by beetles don't stand long enough to serve as useful nest sites in some instances (*fig. 1*).

Fire

Historically, fire was likely the most common mortality agent in ponderosa pine forests, and is the most widely studied means of tree death in terms of evaluating woodpecker response. Fire can directly kill and weaken trees through crown scorch, cambial girdling, and root damage (Harmon and others 1986). The effects of fire are highly variable and typically depend on the type of burn, its intensity, topography, and forest type. In ponderosa pine forests, fires historically burned at frequent intervals and low intensities (Agee 1993). However, management activities common during most of the 20th century including timber harvest, livestock grazing, and fire suppression, have altered both the structure and function of many of today's forest stands, resulting in an increase in the number, size, and severity of fires (Allen and others 2002). Typically woodpeckers respond dramatically to both large and small scale fires where they find an abundance of insect prey and decaying snags suitable for nesting. However, the specific patterns of woodpecker use are likely dependent upon the severity of the fire and whether trees are killed directly or experience a prolonged weakening and death from other associated mortality agents such as bark beetles.

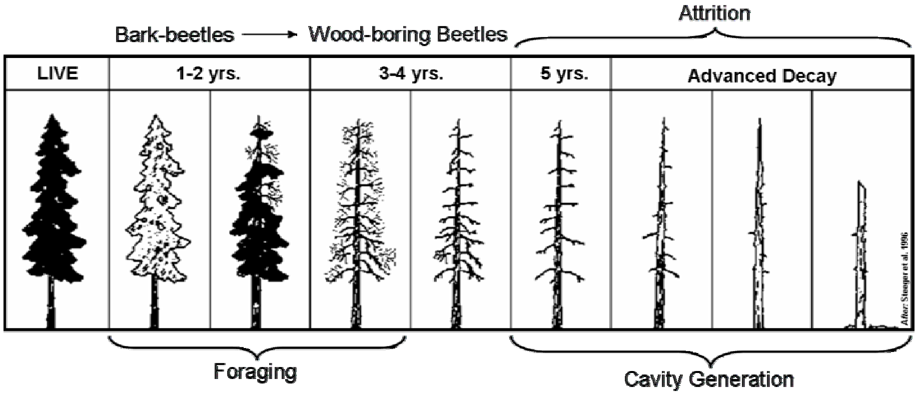

Figure 1—Generalized conceptual model of temporal woodpecker use and attrition of snags created by bark beetles in ponderosa pine. Foraging activity is most intense during the first 3 years following death and corresponds to the insect types active within the snag. Cavity generation and snag attrition both begin after approximately 5 years. The shading of the trees in this figure illustrates changes in foliage color from a healthy green (far left shaded black), fading to yellow (second panel shaded as light grey) and dead and shedding red needles (third panel shaded as black with gaps of needle loss). Schematic patterned after Steeger and others 1995).

Foraging woodpeckers, particularly those of the genus *Picoides* respond immediately to post-fire areas where they typically prey on bark and wood boring beetles that invade dead and dying trees (Farris and Zack in press, Murphy and Lehnhausen 1998). This foraging period is generally brief, lasting only 2-3 years after the fire, before most of the beetle prey has been exhausted (Farris and Zack in press, Murphy and Lehnhausen 1998). As decay progresses in subsequent years, insects such as carpenter ants and termites become important prey for other species of woodpecker, such as the pileated. In general, snags selected by foraging woodpeckers within burned areas are typically large in diameter (Kreisel and Stein 1999, Murphy and Lehnhausen 1998), and of moderate fire damage (less than 50 percent of the bole burned) (Murphy and Lehnhausen 1998). However, some studies have demonstrated woodpecker use of both small and large diameter snags in post-fire environments (Hutto 1995, Horton and Mannan 1988, Haggard and Gains 2001), suggesting that other variables, such as fire damage, tree species, bark thickness, timing, and insect activity may be important factors.

Nesting woodpeckers typically use post-fire environs relatively later than foraging woodpeckers. For example, models created by Lehmkuhl and others (2003) for ponderosa pine in the eastern Cascades of Washington predicted nest cavities to most likely occur between 5 and 25 years post-fire. Note that these were cumulative numbers across a chronosequence of wildfires so the exact timing of excavation and occupancy was unknown. In contrast, the models of Saab and others (2004) predicted nest occupancy to be highest during the first 5 years, with cavities showing up as early as 2 years after the burn. Snags selected as nesting sites are generally larger in diameter than the average available snag (Bull and others 1997, Lehmkuhl and others 2003, Saab and Dudley 1998, Saab and others 2004).

Snag longevity following fires is variable and likely dependent on numerous, site-specific factors such as fire severity, soil characteristics, snag size, and local microclimate (Landram and others 2002; Laudenslayer, this volume). Everett and others (1999) found that 50 percent of the small (<23 cm [9.1 in] dbh) ponderosa pine fell or broke to heights less than 1.5 meters (4.9 feet) during the first 7-12 years following fires in the eastern Cascades of Washington. Larger ponderosa pine (>41 cm [16.1 in] dbh) were scarce in their study areas, but of the few present, 79 percent remained standing 60 years post-fire. In Colorado, Harrington (1996) documented ponderosa pine snags falling as early as 3 years following fire. Morrison and Raphael (1993) found a 68 percent decline in snags 18 to 23 years following a fire event in the eastern Sierra Nevada of California, but do not distinguish between pine and fir. In northeastern California, Laudenslayer (2002) did not observe any snags fall during the 7 year study period, but did document top breakage after 5 years. Farris and Zack (in press) found 20 percent of Jeffrey pine snags fell 4 years following a fire event in the southern Cascades of California. There is conflicting evidence regarding the relative attrition and decay rates of ponderosa compared to other conifers such as white-fir. Landram and others (2002), Morrison and Raphael (1993) and Raphael and White (1984) all found that both ponderosa and Jeffrey pine fell at more rapid rates than white-fir. In contrast, one study in the southern Cascades of California noted greater persistence of Jeffrey pine as compared to white-fir following fire (Farris and Zack in press). Additionally, Lowell (1996) and Farris and Zack (in press) both noted more pronounced wood decay in white-fir compared to ponderosa and Jeffrey pine following fire, which may lead to more rapid attrition.

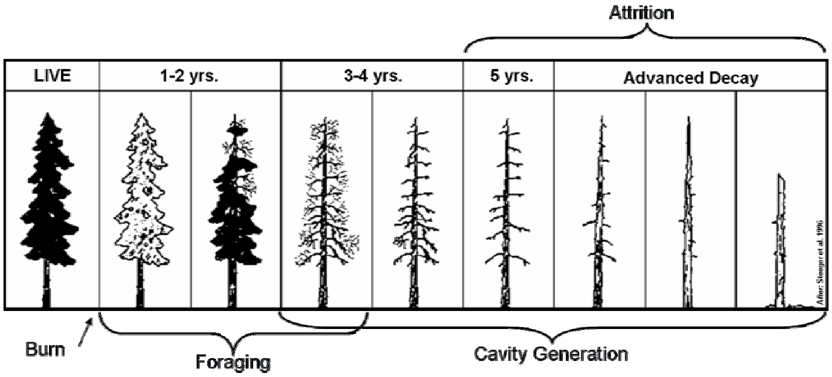

Figure 2—Generalized conceptual model of temporal woodpecker use and attrition of snags created by low to moderate intensity fire in ponderosa pine where most of the trees are weakened and eventually infested with bark and wood boring beetles. Foraging activity is most intense during the first 3 years following fire, while cavity generation can overlap this period and last for several years following the burn event. Fire-killed trees generally begin to fall about 5 years after death. Schematic adapted after Steeger and others 1995).

In summary, the temporal patterns of snag use by woodpeckers following fire events are concentrated within the first 1-5 years, but can extend to 15 years following wildfire. Foraging is most intense within the first 1-3 years after the burn and nesting activity overlaps the later stages of foraging and extends throughout the life of the snag. Attrition and advanced decay typically starts around 5 years after the burn, but can occur as early as 3 years (*fig. 2*). These generalizations are for low-moderate severity fires in which bark and wood-boring beetles play an active role.

Artificial Creation

Methods of artificial snag creation are becoming a tool to meet snag management guidelines in areas lacking suitable snag densities. Creation methods vary and include the use of insect pheromones (Shea and others 2002), mechanical girdling (Shea and others 2002, Hallett and others 2001), girdling through basal burning (Parks and others 1999), mechanical topping (Hallet and others 2001), and topping using explosives (Bull and others 1981). Typically, the motivation for creating snags is to provide nesting habitat, so few studies have evaluated the quantitative use of these artificially created structures as foraging substrates. In ponderosa pine 3 creation methods have been well documented: girdling, topping, and pheromone baiting.

In eastern Washington, foraging woodpeckers showed no preference between topped and girdled snags (Hallet and others 2001). Parks and others (1999) documented equal amounts of woodpecker foraging on ponderosa killed by basal burning and mechanical girdling in New Mexico. Shea and others (2002) documented a greater use of pheromone-baited trees as compared to girdled trees for two types of woodpecker foraging strategies aimed at two distinct types of insects. In their study, all trees that were baited with western pine beetle pheromone, exhibited woodpecker “flaking” (superficial removal of successively thin layers of bark to procure pupae and emerging adult bark beetles) within a few months after treatment, while none of the girdled trees showed evidence of flaking; suggesting that bark beetles were not present or were too rare to exploit in the girdled snags. In contrast, evidence of woodpecker foraging “excavations” (distinct holes created in the bark that often penetrate through to the tree’s sapwood created in pursuit of wood-boring beetles) were recorded on snags in both treatment types, but in much greater densities on the pheromone baited snags as compared to the girdled snags; suggesting that these structures provided a more productive foraging medium for woodpeckers.

Nesting woodpeckers tended to prefer topped trees versus girdled trees in Washington (Hallett and others 2001), while Parks and others (1999) documented greater nest use in mechanically girdled snags versus trees killed using basal burning methods, or a combination of girdling and burning. In the Southern Cascades, 44 percent of pheromone baited snags contained woodpecker nest cavities 6 years after treatment, while none of the girdled trees were used by nesting woodpeckers (Shea and others 2002). However, 2 years later, 14 percent of the girdled snags contained cavities and the use of baited snags increased to 50 percent (Shea unpublished data).

Decay and attrition of snags created by the three methods are variable and once again, likely dependant on local site conditions. Most authors cited “decay” as the breaking of branches off the bole, the loss of bark, or advancement between the standardized decay stages outlined by Thomas and others (1979) or Cline and others (1980). Hallet and others (2001) found that topped trees tended to decay slower than girdled trees, but have little data specific to ponderosa pine and do not speculate on fall rates. In New Mexico, 97 percent of the girdled trees remained standing while only 68 percent of the basally burned trees remained standing 4 years after treatment. After 6 years, standing girdled trees declined to 72 percent, while basally-burned trees declined to only 36 percent. In northeastern California, 16 percent of the pheromone baited trees had fallen, while only 12 percent of the girdled trees fell or broke to heights less than 1.5 meters (4.9 feet) 6 years after treatment (Shea and others 2002).

Summary and Implications

Several factors determine how and when snags are utilized by foraging and nesting woodpeckers. These include the mode of mortality, time since tree death, and snag size. In terms of foraging, woodpecker utilization of snags appears to be most concentrated within the first three years of tree death, especially on snags originating from bark-beetles and/or a combination of bark beetles and low/moderate severity fires. Snags created artificially through girdling or topping are used less intensively for foraging. Moreover, peak foraging activity in artificially snags typically occurs after the first two or three years of snag creation. This disparity is likely due to differences in the sequence of insect infestation. Greater insect diversity and abundance have been reported in naturally created snags compared to snags created artificially by girdling (Shea and others 2002).

In comparison to foraging woodpeckers, snag utilization by nesting woodpeckers appears to be concentrated in older snags with some level of decay. Importantly, intensive foraging by woodpeckers in young snags may facilitate future cavity excavation by promoting wood decomposition through a combination of structural damage to the wood and the transfer of tree decaying fungi. The manner in which a snag is created seems to have an effect on the timing of cavity excavation in ponderosa pine. Snags decay to cavity-bearing structures fastest in burned areas, as soon as 2 to 3 years after the fire in some cases. Bark beetle killed trees provide nesting habitat as early as 5 years after death, while girdled trees can take as long as 6 to 8 years. Snag size is another predictor of cavity generation with larger snags more likely to eventually contain nest cavities. However, it is important to note, that a very small fraction of available snags actually contain cavities excavated by woodpeckers; suggesting the need for more consideration of mortality agent.

Snag attrition rates are highly variable and tend to depend not only on the mode of mortality, but also on local site conditions. In general however, the fastest recorded attrition was in instances of fire-killed trees, which fell as early as 4 years after the fire. These patterns may be directly related to the severity of fire damage on individual trees and the history of post-fire beetle use. Generally, snags that experience greater use by both bark and wood-beetles attract foraging woodpeckers, which serve to further degrade the wood integrity and can lead to faster decay rates than trees that don’t experience these same use patterns. However, there are instances where trees that experience extremely severe fire damage stand for extended periods of time and are not used by either beetles or woodpeckers. Trees killed by insects, independent of fire, stand slightly longer than those affected by both fire and insects simultaneously. Insect-killed snags begin to fall between 5 to 6 years after death. This can be problematic, as these snags may fall before they can provide nesting habitat. Snags created artificially by topping or girdling tend to stand longer than either fire or insect killed trees. This may be due to the relatively less use of these trees by beetles and woodpeckers, which can contribute to expedited decay of these structures (Farris and others 2004).

These temporal trends in woodpecker use and snag decomposition patters are paradoxical. Trees killed by bark beetles alone, or in association with fire serve as high quality foraging habitat and seem most likely to contain future nest cavities. However, these same structures are likely to fall more rapidly than snags originating from sources of mortality that don't attract beetles and foraging woodpeckers in similar intensities (e.g. artificial creation methods such as mechanical girdling or topping).

Classically, snag management has been driven by the objective to provide nesting habitat for snag-dependant species such as woodpeckers. Many of the strategies have centered on the notion that snags simply needed to be large enough and in adequate densities in order to be useful for most wildlife species. As illustrated through this review, recent information suggests that managing quantity while ignoring quality may not be sufficient. In particular the following 5 principles should be considered:

1. Snag recruitment and attrition is a dynamic process that is dependent on characteristics of individual sites.
2. Snags are an ephemeral resource on the landscape (e.g. half of all ponderosa snags may fall within 8 years of death).
3. Not all snags are useful to wildlife (a small fraction of standing snags are actually used by cavity-nesting wildlife).
4. The manner in which a tree dies affects its subsequent use by beetles and woodpeckers.
5. Foraging use by both beetles and woodpeckers appears to play a significant role in the decay dynamics, cavity excavation patterns, and standing life span of a snag. These interactions likely influence future use by subsequent snag-dependent wildlife species.

Consideration of these important sources of variability in snag ecology and woodpecker use may help improve snag management in ponderosa pine forests.

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Fire Ecology of Ponderosa Pine and the Rebuilding of Fire-Resilient Ponderosa Pine Ecosystems [1]

Stephen A. Fitzgerald[2]

Abstract

The ponderosa pine ecosystems of the West have change dramatically since Euro-American settlement 140 years ago due to past land uses and the curtailment of natural fire. Today, ponderosa pine forests contain over abundance of fuel, and stand densities have increased from a range of 49-124 trees ha^{-1} (20-50 trees acre^{-1}) to a range of 1235-2470 trees ha^{-1} (500 to 1000 stems acre^{-1}). As a result, long-term tree, stand, and landscape health has been compromised and stand and landscape conditions now promote large, uncharacteristic wildfires. Reversing this trend is paramount. Improving the fire-resiliency of ponderosa pine forests requires understanding the connection between fire behavior and severity and forest structure and fuels. Restoration treatments (thinning, prescribed fire, mowing and other mechanical treatments) that reduce surface, ladder, and crown fuels can reduce fire severity and the potential for high-intensity crown fires. Understanding the historical role of fire in shaping ponderosa pine ecosystems is important for designing restoration treatments. Without intelligent, ecosystem-based restoration treatments in the near term, forest health and wildfire conditions will continue to deteriorate in the long term and the situation is not likely to rectify itself.

Introduction

Historically, ponderosa pine ecosystems have had an intimate and inseparable relationship with fire. No other disturbance has had such a re-occurring influence on the development and maintenance of ponderosa pine ecosystems. Historically this relationship with fire varied somewhat across the range of ponderosa pine, and it varied temporally in concert with changes in climate.

Over the last century this relationship has undergone an increasing amount of strain. Although wildfires continue to burn in ponderosa pine ecosystems, recent wildfires tend to be more intense and lethal, and consume large expanses of ponderosa pine forests. There is an impending need to better understand the role of fire and techniques land managers can use to emulate fire to restore fire-resiliency. I define fire-resiliency simply as the ability of ponderosa pine forests to survive wildfires relatively intact, as typically occurred during presettlement times.

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[1] An abbreviated version of this paper was presented at the symposium on Ponderosa Pine: Issues, Trends and Management, October 18-21, 2004, Klamath Falls, Oregon.

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This paper discusses the ecological role of fire in ponderosa pine ecosystems, changes in forest structure and fire behavior over the past century, and strategies for rebuilding fire-resilience in ponderosa pine ecosystems in western North America.

Adaptations and Morphological Characteristics Affecting Fire Resistance and Survival

Various adaptations allow vegetation to survive fire. Adaptations can either facilitate survival of species (e.g., fire-stimulated flowering, refractory seed buried in soils, etc) or individuals (e.g. thick bark, basal sprouting, etc) (Kauffman 1990). Ponderosa pine is considered one the most fire resistant conifers in the west, and fire resistance increases as the tree matures (Miller 2000). Ponderosa pine is well suited to survive low-intensity surface fires primarily due to its bark characteristics. Ponderosa pine develops a protective outer corky bark (0.3-0.6 cm) early in life when saplings reach a basal diameter of 5 cm allowing some young trees to survive very light-intensity surface fires (Figure 1a) (Hall 1980). Mature ponderosa pine trees possess thick, exfoliating bark (Figure 1b), which slough off when the bark is on fire. Presumably, this helps “take away” heat as flaming bark flecks flake off, thus reducing or preventing heat transfer and minimizing injury to cambial cells; however, this mechanism has not been well researched. Ponderosa pine bark on mature trees continues to flake off with or without fire, and over long time periods without fire a thick mulch layer of bark develops at the base of trees. When ignited, this mulch smolders for days, conveying heat directly to and through the bark to the cambial layer, often killing or severely stressing the tree. Bark beetles may then attack and kill weakened trees. Historically, frequent low-intensity surface fires prevented this bark mulch layer from accumulating around mature trees.

Ponderosa pine also has a deep rooting habit compared to other western conifer species (e.g., true firs, Engelmann spruce, and lodgepole pine). Although a surface fire may heat the soil and kill some surface roots, deeper roots remain intact and allow for continued uptake of water and nutrients. The amount and moisture content of surface fuels (needles and branches, saplings and herbaceous plant material) along with larger woody debris (downed logs) beneath or in contact with the tree affects the degree of injury to surface roots.

Ponderosa pine crown structure, branching pattern, and needle and bud characteristics also affect survival during fire. The open crown structure and branching pattern of ponderosa pine allows for better mixing of air and dissipation of heat within stands during a fire, thus reducing the potential for crown scorch. The open crown structure may also dampen fire-spread through tree crowns in less extreme fire conditions (Flint 1925, Agee 1993). Ponderosa pine has long needles with high moisture content that surround terminal buds. Although needles may be scorched and killed by heat, they help protect meristematic tissue within the bud, allowing branch tips to refoliate. Buds of ponderosa pine have thick outer scales that also help protect meristematic tissue from heat (Miller 2000).

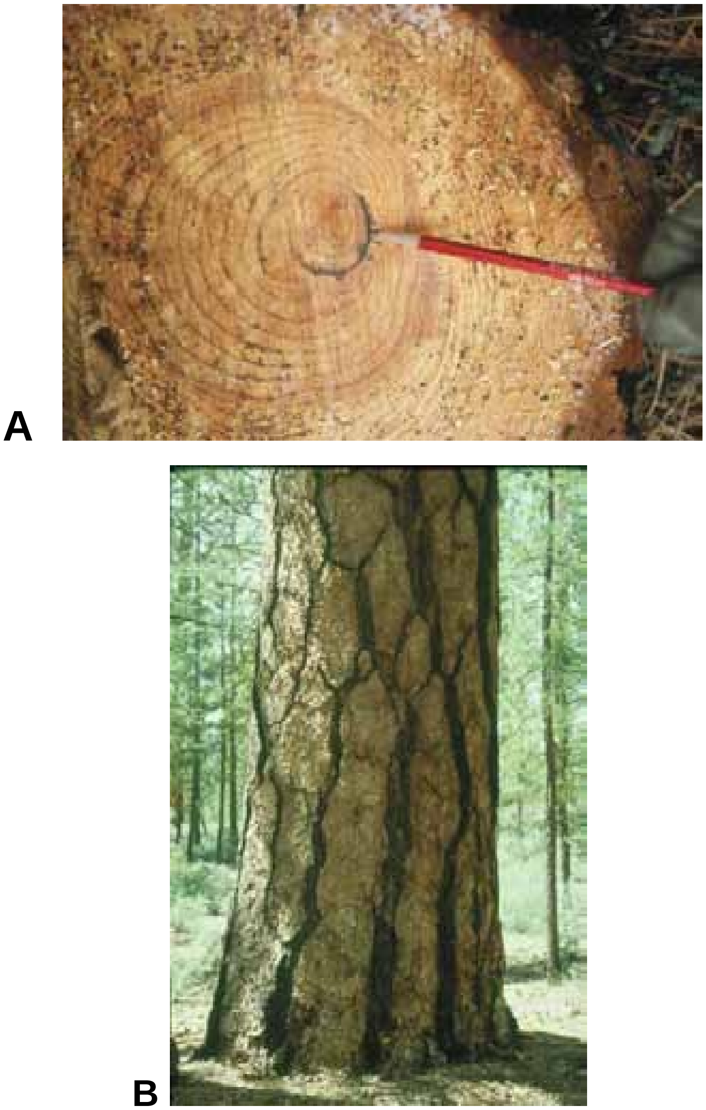

Figure 1—(A) This young ponderosa pine survived a light-intensity surface fire when the basal diameter was about 3-4 cm. Note that the cambium on approximately half the circumference was killed. The sapling recovered and greatly accelerated its diameter growth probably as a result of less competition and a flush of nutrients following the fire. (B)Typical fire-resistant bark of an old-growth ponderosa pine showing the platy bark surface.

Another factor reportedly affecting fire resistance is ponderosa pine's ability to self-prune (gradual shedding of lower branches) (Flint 1925, Starker 1934, Miller 2000), resulting in the clear bole that often characterizes large-diameter old-growth ponderosa pine. Presumably, this mechanism lifts the lower crown over time and prevents surface fires from moving up into the tree's canopy. However, there is no evidence that ponderosa pine self-prunes on its own. The clear boles are a result of either repeated surface fires, which scorch and kill lower branches when trees are young and lower branches have small diameters, and/or death of lower branches from competition (shading) from neighboring trees. In both cases, dead branches are shed and remaining stubs are grown over after decades or centuries of tree growth. Evidence that counters the notion of self-pruning is demonstrated by open-grown ponderosa pine in areas where fires were naturally excluded, such as in rocky areas where fuels are too scarce to carry fire. Often these trees have branches that are large, heavy, and located on the lower portions of the bole showing no propensity for self-pruning.

Fire Regimes in Ponderosa Pine Ecosystems

Fire regimes are influenced greatly by climate, vegetation types and by topographic and geologic features that either facilitate or restrict fire spread (Agee 1993, Camp and others1997, Taylor and Skinner 1998). Fire regimes are characterized by their frequency, intensity, severity, extent, and seasonality and have a great influence on vegetative recovery, plant succession, and forest and ecosystem structure (Agee 1993).

Frequency

Fire frequency, or the mean fire return interval, is a measure of how often fire returns, on average, to an area. There may be a wide range around this mean, which has important ecological implications for stand development and forest structure (Baker and Ehle 2001). The median fire return interval is also used to characterize fire return intervals in forest ecosystems.

Within ponderosa pine ecosystems, fire returned approximately every 2-47 years. This estimate of fire frequency is based on several studies that date fire scars on individual trees (point sample) or from several fire-scarred trees in an area (composite fire interval) (Table 1). The wide range in fire frequency is a reflection of current and past regional climate, plant association, aspect and slope, elevation, aboriginal burning and other factors. In the Front Range of Colorado, for example, ponderosa pine forests were subject to frequent surface fires at lower elevations, much like other ponderosa pine forests in the west. At higher elevations (2400 m), where ponderosa pine is mixed with Douglas-fir and lodgepole pine (more moist conditions), fires were less frequent and were a combination of both surface and stand-replacing fire (Veblen and others 2000). In western Montana, Arno and others (1997) observed the same change in fire frequency from pure ponderosa pine stands to more mesic mixed stands of ponderosa pine and western larch. Other studies have shown a link between regional climate patterns, such as periods of wet and dry (drought) conditions and fire occurrence (Swetnam 1988, Touchman and others 1996, Veblen and others 2000, Grissino-Mayer and others 2004, Wright and Agee 2004).

Lightning is thought to have been the primary source of presettlement fire ignitions. Some geographic areas are more prone to lightning than others due to prevailing summer weather patterns and topography. The zone that extends from northwest California up through Oregon, northern Idaho and northwest Montana, known as “lightning alley,” is a prime example of the relationship between lightning ignitions and fire frequency in ponderosa pine forests (Figure 2).

Aboriginal burning also affected fire occurrence in localized areas. Barrett and Arno (1982) compared presettlement fire intervals in forests known to have had heavy use by Native Americans and compared that to fire intervals on similar sites but in more remote areas. Fire intervals were twice as frequent in heavy use areas (MFI of approximately 5-6 years in heavy-use areas versus 12.5 years in remote areas for two sites). However, Indian-caused burning may have been much more wide spread, as documented by early explorer and pioneer writings, particularly after Native Americans acquired horses in the early 1700s (Barrett 1980, Barrett and Arno 1982).

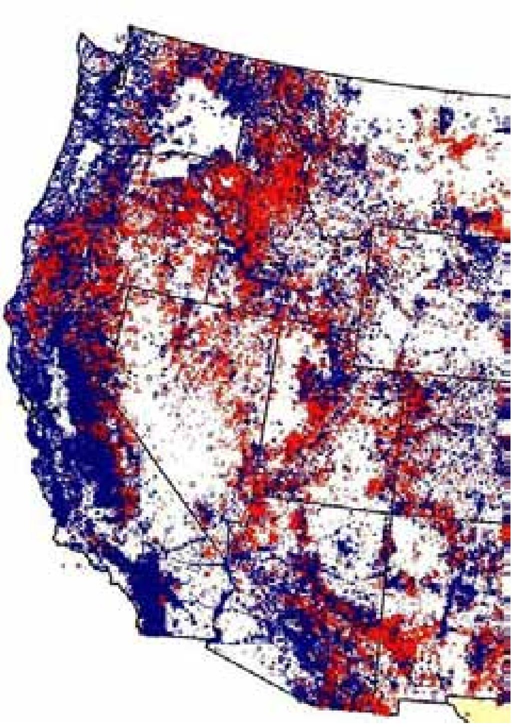

Figure 2 — Lightning occurrence in the western United States (from Schmidt and others 2002). Note the concentration of lightning from northern California extending northeast through Oregon, Idaho, and northwest Montana.

Intensity and Severity

Fire intensity and severity are often used interchangeably, but fire scientist distinguish between them. Fire intensity is a measure of heat or energy released (kW) per unit length (m) along the fireline, and can be estimated by measuring flame length as the flaming front passes a known point (Rothermal and Deeming 1980). Fire severity is determined by either a visual estimate or measured assessment of fire effects on soils and vegetation. High intensity fires (e.g., long flame lengths), for example, result in more consumption and charring of surface fuel, increased exposure of soil and alteration of soil properties, and more damage and mortality of trees and other vegetation.

Historically, ponderosa pine ecosystems were predominantly subjected to frequent, low-intensity fire (Agee 1993). These periodic fires would consume accumulated fuels, thin young seedlings and saplings, and consume shrubs and herbaceous plant material, leaving the large, fire-resistant trees intact. Some individual large trees or small groups of large trees may have been directly killed or stressed by fire and later attacked and kill by bark beetles (Munger 1917).

On more moist sites where ponderosa pine is seral (on Douglas-fir, dry and wet Grand fir series), fire intensity was often greater and resulted in a more mixed-severity fire regime composed of a combination of underburning (light intensity) and patch stand-replacement (high intensity) fire (Agee 1993, Arno and others 1995, Arno and others 1997, Veblen and others 2000, Baker and Ehle 2001, Wright and Agee 2004). In the ponderosa pine forests of the Black Hills of South Dakota, fires were of higher intensity and more of mixed-severity (Brown and Sieg 1996, Shinneman and Baker 1997), most likely because of longer fire intervals and abundant regeneration that establishes between successive fires.

Extent

The perimeter and extent of presettlement fires is often difficult to ascertain because fires of very low intensity may not scar trees, or subsequent fires may destroy fire scars in catfaces or kill and consume fire-scarred trees (Agee 1993). The extent of presettlement fires was highly variable and often associated with changes in inter-annual weather patterns (Norman and Taylor 2003). In central Washington, Wright and Agee (2004) found that most fires were small (< 1000 ha (2470 ac)) and that large fires (> 4000 ha (9880ac)) occurred every 27 years and were associated with annual periods of drought (Palmer Drought Severity Index and winter Southern Oscillation Index (El Nino)). In northern California, Taylor and Skinner (1998) found that average burn area was 350 ha, but documented 16 fires between 1627 and 1992 that were larger than 500 ha. In southwest Colorado, Grissino-Mayer and others (2004) found that historic fires in the lower elevation ponderosa pine stands were small and quite localized but at higher elevations, where ponderosa pine is mixed with other species, fires were less common but larger in size. In central Colorado and in the Southwest there is an association between large fires and periods of wet years that increase dry matter production (fuel), followed by dry spring and summer months (Swetnam and Baisan 1996, Touchan and others 1996, Veblen and others 2000, Donnegan and others 2001, Norman and Taylor 2003, Grissino-Mayer and others 2004). Norman and Taylor (2003) found that extensive burning of 1000s of hectare took place approximately every 20 years in northeastern California.

Table 1 – Fire return intervals by region and plant series for climax and seral ponderosa pine (adapted from Agee (1994) and Baker and Ehle (2001)).

Geographic Region	Mean Fire Return Interval	Median Fire Return Interval	Composite (C) or Point (P) Sample	Plant Series	Study
Central Washington (E. Cascades)	18.8	15.5	C	Douglas-fir	Wright and Agee (2004)
	20.6	17.4	C	Grand fir (dry)	Wright and Agee (2004)
	7	6.7	C	Douglas-fir	Everett and others (2000)
	7-11	--	C	Douglas-fir	Wischonfske and Anderson (1983)
	23.9	20.6	C	Grand fir (wet)	Wright and Agee (2004)
Northeast Washington	10-24	--	P	Douglas-fir	Finch (1984)
Northern Rocky Mountains:					
Western Montana	6-11	--	P	Ponderosa pine	Arno (1976)
	7-19	--	P	Douglas-fir	Arno (1976)
South Dakota (Black Hills)	20-23	--	C	Ponderosa pine	Brown and Sieg (1996)
	10-12	--	C	Ponderosa pine	Brown and Sieg (1999)
Central Oregon (E. Cascades)	16-38	--	C	Ponderosa pine (dry)	Bork (1985)
	7-20	--	C	Ponderosa pine (mesic)	Bork (1985)
	9-25	--	C	White fir	Bork (1985)
North-central Oregon (E. Cascades)	11-16	--	P	Ponderosa pine	Weaver (1959)
	3-36	--	P	Ponderosa pine	Soeriaatmadja (1966)
Northeast Oregon (Blue Mts.)	10-43	20	C	Ponderosa pine	Heyerdahl and others (2001)
	10		P	Douglas-fir	Hall (1976)
	12-53	5-15	C	Douglas-fir	Heyerdahl and others (2001)
	47	--	P	White fir	Weaver (1959)
	13-71	15	C	White fir	Heyerdahl and others (2001)
South-central Oregon (Cascades)	9-42	--	C	White fir	McNeil and Zobel (1980)

Table 1 **--** continued

Northern California (S. Klamath Mts.)	11.5		--		C	Ponderosa & limber pine	Taylor and Skinner (2003)
	13.0		--			Douglas-fir	Taylor and Skinner (2003)
	13.0					Douglas-fir/white fir	Taylor and Skinner (2003)
	12.5		--			Douglas-fir/ponderosa pine/incense-cedar	Taylor and Skinner (2003)
				C			
	13.5	--		C		Douglas-fir &limber pine	Taylor and Skinner (2003)
Northern California (N. Klamath Mts.)	12.0-15.5		--	C	C	Douglas-fir	Taylor and Skinner (1998)
Northern California (S.Cascades Mts.)	21.4	--	12.0	C	C	Ponderosa pine	Norman and Taylor (2003)
Central California (Westside Sierras)	9-18		--			White fir	Kilgore and Taylor (1979)
				C			
Central Rocky Mountains:							
San Juan Mts.	6-13		5-9			Ponderosa pine	Grissino-Mayer and others (2004)
	19-30		19-29		C	Mixed conifer	Grissino-Mayer and others (2004)
Northern Front Range	8.3-22.4		6.4-11.6[1]	C	C	Ponderosa pine	Veblen and others (2000)
	17.2-18.6		8.1-10.4[1]		C	Douglas-fir	Veblen and others (2000)
Southern Rocky Mountains:							
North-central Arizona	3.7				C	Ponderosa pine	Fulé and others (1997)
	7.4				C	Ponderosa pine	Fulé and others (2003)
	8.7				C	Mixed conifer	Fulé and others (2003)
Arizona	4-12	4				Ponderosa pine	Weaver 1951
Arizona	1.8	6				Ponderosa pine	Dieterich 1980
Southwest New Mexico	5-8	5	--			Ponderosa pine	Swetnam and Dieterich (1985)
		--		P			
		--		C			
				C			

[1] Weibull median probability interval

Seasonality

Fire in ponderosa pine systems varies somewhat by season and geographic region within the species range as a result of regional weather patterns. Seasonality of presettlement fires can be determined using dendrochronology techniques that examine the position of each fire scar relative to ring development, such as during earlywood and latewood development or after the cessation of latewood growth (Baisan and Swetnam 1990). In north-central Washington, Wright and Agee (2004) found that 80 percent of presettlement fires occurred late in the growing season (during late-wood formation) or in early fall (after cambial dormancy) for seral ponderosa pine growing on Douglas-fir, dry grand fir, and wet grand fir plant association groups (PAGs). Heyerdahl and others (2001) found that in the southern Blue Mountains a higher proportion (26 percent) of fires occurred during earlywood and latewood formation, while on more mesic sites in the northern Blue Mountains most fires occurred after cessation of ring growth. In northern California, 93 percent of fires occurred during the dry midsummer through early fall period (Taylor and Skinner 2003). In a study in southwestern ponderosa pine where season could be determined, Fulé and others (1997) found that approximately 40 percent of the presettlement fires occurred in spring (late April to June) and 60 percent in the summer (July to early September). There appears to be a north-south gradient of seasonality within the Medeterranean Climate zone of southwestern Oregon (Klamath Mountains) to the southern Sierra Nevada in California. Fire scars are mostly (>90%) at the ring boundary in the Klamath Mountains, indicating mostly late-summer/early-fall fires. In the southern Sierra Nevada, fire scars are mostly (>80%) within the rings, indicating many early- to mid-summer fires (Skinner 2002)

Historic Forest Structure

Presettlement ponderosa pine forests throughout its range were typified by open, park-like stands of large-diameter trees with few seedlings and saplings in the understory (Langille and others 1903, Munger 1917, Weaver 1943, Morrow 1985, Arno 1988, Fulé and others 1997, Youngblood and others 2004, Moore and others 2004). Stands were typically uneven-aged, with many stands containing a few large individual trees 400 to 600 years old (Munger 1917, Morrow 1985, Arno and others 1995, Youngblood and others 2004). Historic photos depict the open character of old-growth ponderosa pine on the Klamath Indian Reservation in south-central Oregon (Figure 3). Other turn-of-the-century photos depict similar forest conditions, although historic photos locations are not randomly located. In addition, the understory plant community may have also been influenced by livestock grazing not readily apparent in most historic photos. Many rangelands and adjacent ponderosa pine forests were heavily grazed during settlement from the late 1860s through the early 1900s (Hessburg and Agee 2003).

Figure 3 -- Harold Weaver standing in an open park-like stand of old-growtth ponderosa pine on the Klamath Indian Reservation in south central Oregon in the 1930s. Note the open understory. (BIA photo)

In an early survey of ponderosa pine forests in Oregon, Munger (1917) found ponderosa pine forests contained approximately 30-99 (12-40 per acre) large-diameter (> 30.5 cm (12 in) DBH) trees per hectare across several sites (Table 2). In both a climax ponderosa pine site in central Oregon and a ponderosa pine-dominated mixed conifer (dry) site in northeast Oregon very few seedlings and saplings are present (Figures 4a,b), and both stands have a relatively low stand density index (SDI) (Reineke 1933) compared to ponderosa pine stands today. Even on the more productive mixed conifer site, there are few seedlings and saplings of ponderosa pine and "other" species, which are likely grand fir and Douglas-fir. Because of frequent fire, presettlement ponderosa pine stands probably had low levels of snags and large woody debris (Skinner 2002). The following quote taken from Munger (1917, page 17) sums up his observations on the historic character of ponderosa pine forests:

> *"In most of the pure yellow-pine forests of the State the trees are spaced rather widely, the ground is fairly free from underbrush and debris, and travel through them on foot or horseback is interrupted only by occasional patches of saplings and fallen trees."*

Table 2 -- Historic trees per hectare of trees greater than 30.5 cm DBH for several sites in Oregon (from Munger 1917).

Location	Hectares Surveyed	Trees per Hectare >30.5 cm
Looking Glass Creek	109	99
Austin & Whitney	638	77
Winlock	49	42
Embody	74	79
LaPine	99	30
Klamath Lake	393	86

Other reconstructive studies have shown similar low stand densities in presettlement ponderosa pine forests across its range (Weaver 1943, Cooper 1961, Morrow 1985, Habeck 1985 (cited in Arno 1988), Taylor and Skinner 2003, Youngblood and others 2004).

The stems of trees within ponderosa pine stands also show a distinct spatial pattern. Historically, many old-growth pine stands had a groupy-patchy stem pattern (Munger 1917, Weaver 1943, Cooper 1961, Biswell and others 1973, Habeck 1985 (cited in Arno 1988), Morrison 1985, Arno and others 1995, Kaufmann and others 2001,Youngblood and others 2004). For example, Youngblood and others (2004) examined the spatial stem pattern of mature trees in old-growth ponderosa pine stands in central Oregon and northern California and found that trees on 24 out of 27 large plots were moderately to strongly grouped (Figure 5), most likely the result of past fire, insect, and regeneration patterns. The open and groupy nature of old-growth ponderosa pine forests affords additional fire resistance because the canopy is broken up by gaps between groups of different sizes and ages of trees, leaving the overall stand less vulnerable to crown fire.

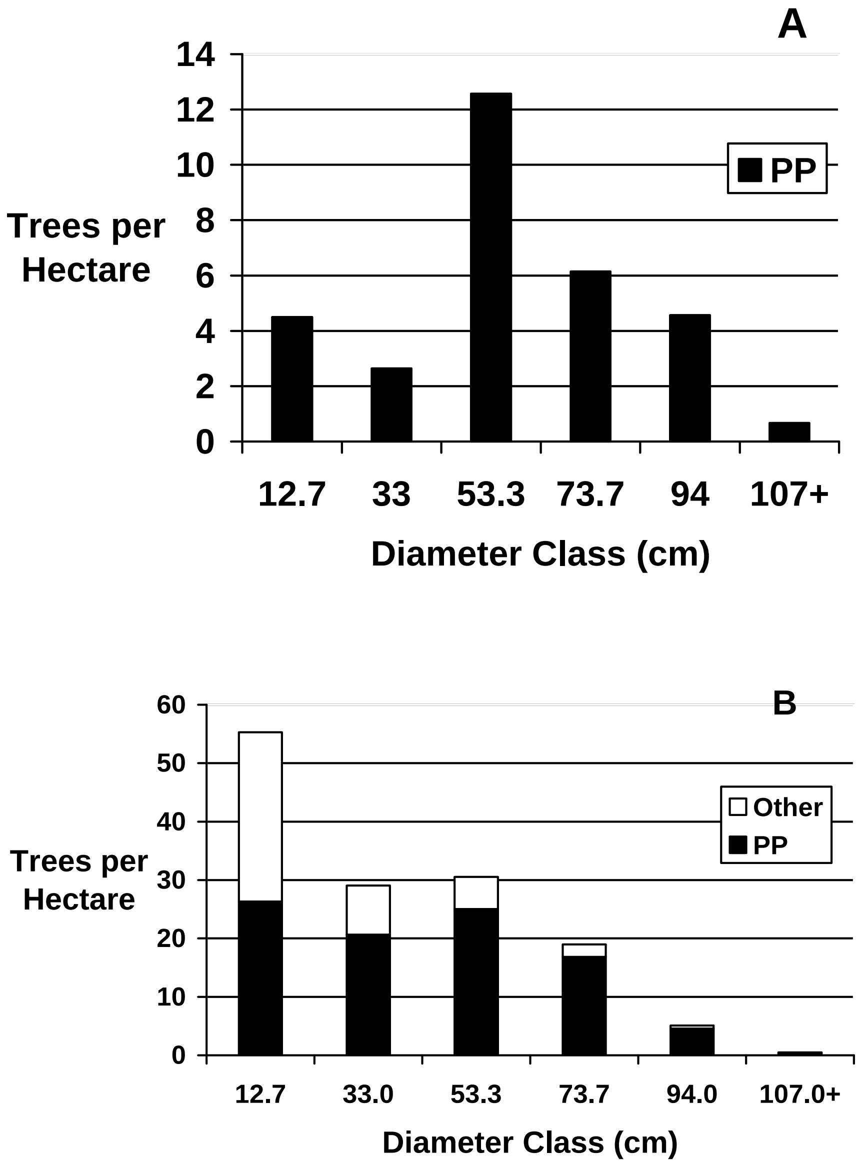

Figure 4 – (A)Trees per acre by diameter class on a climax ponderosa pine site in central Oregon and (B) on a ponderosa pine-dominated mixed-conifer site in northeastern Oregon (from Munger 1917).

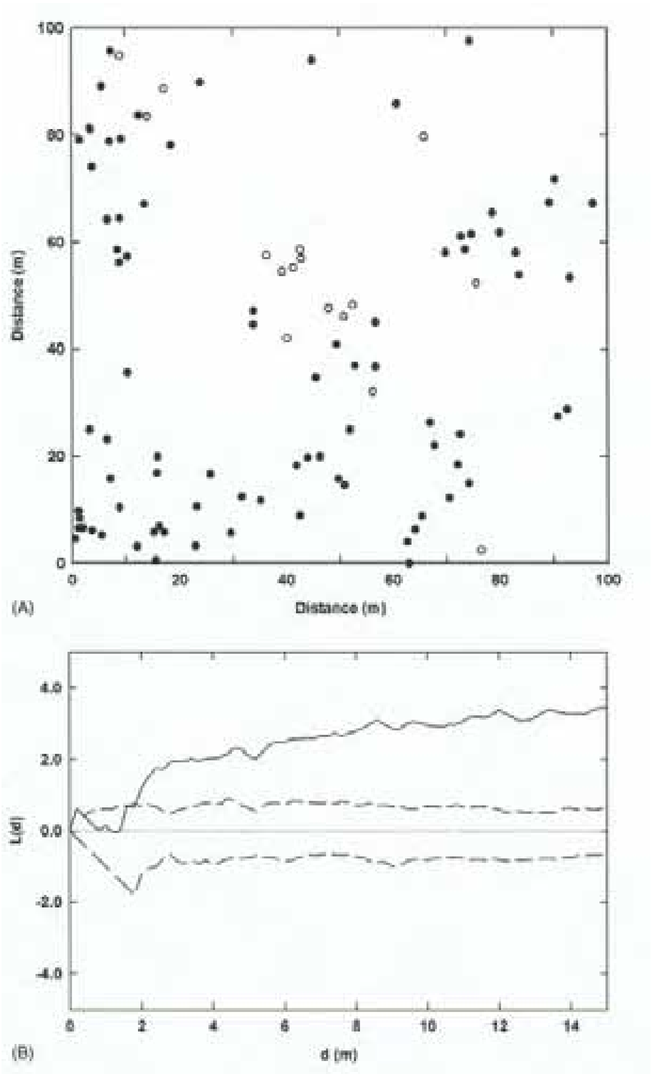

Figure 5 – (A) Spatial pattern of large (•) and dead (o) trees in the upper canopy of old growth plot 23 at Metolius study area and (B) the results of Ripley's *K(d)* analysis on these 99 trees, with empirical cumulative distribution of *L(d)* shown as a solid line, the expected random distribution as a dotted line with mean equal to zero, and point-wise 95% confidence envelop around the expected distribution of complete spatial randomness shown by dashed lines (from Youngblood and others 2004).

Changes in Forest Structure, Fuels and Fire Regime

Ponderosa pine forests have undergone great ecological change in the last 140 years since settlement (Hessburg and Agee 2003), and not for the better from a fire and forest health perspective. Heavy grazing in the late 1800s and early 1900s, active fire suppression after 1910, and other land uses have disrupted the natural fire regime in ponderosa pine ecosystems. This has allowed succession to proceed unchecked resulting in above-normal fuel accumulations and abundant tree regeneration. In addition, early selective logging removed the large, fire-resistant ponderosa pine, western larch, and Douglas-fir, creating openings that favor the development of shade-tolerant Douglas-fir and true firs in the understory and mid-canopy layers (ladder fuels) on more mesic sites.

These past land use activities have produced cascading ecological effects that are manifested today in altered ecological processes (Weatherspoon 1996, Keane and others 2002). Forest stand structure has changed from open park-like stands dominated by large, fire-resistant trees to over-dense even-aged stands (Weaver 1943, Covington and Moore 1994, Moore and others 2004) that are more susceptible to crown fire (Weatherspoon and others 1992, Skinner and Chang 1996), and contain trees that are less likely to survive fire because of their smaller diameter, thinner bark and low hanging crowns (Agee 2002). In ponderosa pine stands in the southwest, forest density increased from 148 trees (65 ponderosa pine and 80 oaks) per hectare in 1883 to 1265 trees per hectare in 1995 (Fulé and others 1997). Similar changes in stand density have been documented in the northwest and elsewhere (Weaver 1943, 1959, Morrison 1985, Arno 1988, Agee 1993, Adams 1995, Weatherspoon 1996, Taylor and Skinner 2003, Youngblood and others 2004). In some cases these changes in succession and stand density have been documented from long-term study plots and time-series photography (Smith and Arno 1999).

Recent coarse-scale assessments have documented changes in the potential fire regime from frequent, low-intensity fires confined mostly to the understory to more lethal high-intensity fires that have the potential to consume entire stands across large landscapes (Figure 6) (Quigley and others 1996, Schmidt and others 2002). On more mesic sites, forest composition has changed from stands dominated by ponderosa pine and western larch to those dominated by Douglas-fir and true firs both of which are more susceptible to insects, disease, and drought (Filip 2002).

Remaining old-growth ponderosa pine forests are in jeopardy as well, due to elevated risk to insect attack, some diseases, and stand-replacement fire (Harrington and Sackett 1992, Weatherspoon 1996, Fitzgerald and others 2000, Keane and others 2002, Moore and others 2004). Management action is needed in these stands to maintain this important structure on the landscape for wildlife that depend on mature, old-growth forest conditions and for human benefits. Because many ponderosa pine forests lie adjacent or in close proximity to the wildland-urban interface, homes and communities are increasingly at risk from high intensity wildfires starting in and spilling over from dense, fire-prone forest conditions. Without efforts to restore ponderosa pine forests to some semblance of natural forest conditions, large intense wildfires will continue, producing detrimental ecological effects and continued deterioration of forest and watershed health (Wickman 1992, Mutch and others 1993).

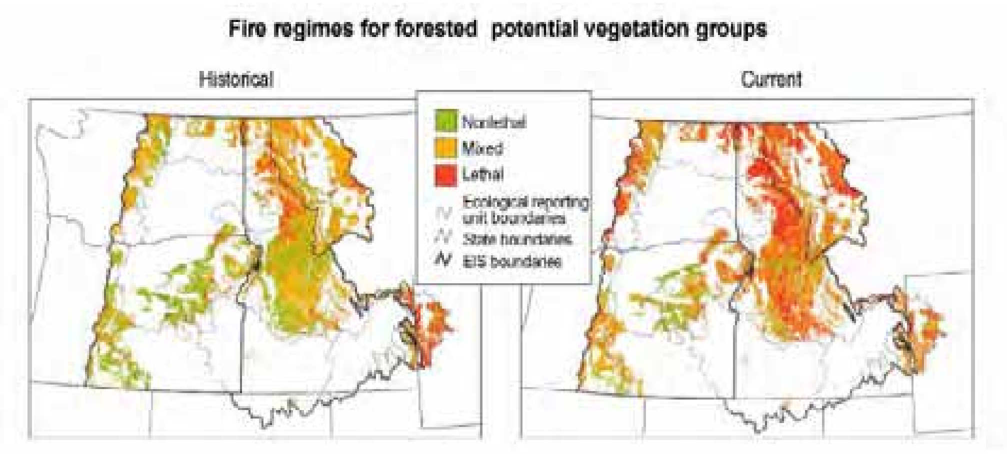

Figure 6 – Historic and current fire regime for the Inland Northwest (from Quigley and others 1996). The increase darker grey areas indicate a change to more lethal fire regimes across the region under current conditions.

Rebuilding Fire Resilient Ponderosa Pine Ecosystems

Fire Behavior and Forest Structure

Fire behavior is affected by the interaction of fuel, weather, and topography, known as the fire behavior triangle (Figure 7). Aspects of fire behavior that managers are primarily interested in include how fast a fire moves (rate of spread), how hot it burns (intensity), torching of tree crowns (passive crown fire), crowning (active crown fire), firewhirls, and spotting. Spotting occurs when fire brands (glowing embers) are lofted up and out ahead of the main fire front, igniting spot fires that then feed back into the main fire front, creating very extreme (and dangerous) fire conditions. A change in any one of these three main factors during the course of combustion will influence a fire's behavior. For example, a surface fire will move (spread) faster upslope because flames are tilted toward the slope and more efficiently dry and preheat fuels in front of the fire. Other topographic features such as saddles and draws affect wind patterns by funneling air and intensifying fire behavior. Weather, too, is a dominant factor in fire behavior. A change in weather from hot and dry to cooler and more moist conditions will reduce fire intensity and rate of spread. Often firefighters have to wait for a change to cooler and wetter weather before they can safely attack a wildfire head on. We have no control over topography and weather, but fire managers often take advantage of favorable weather conditions or topographic features to attack and suppress wildfires.

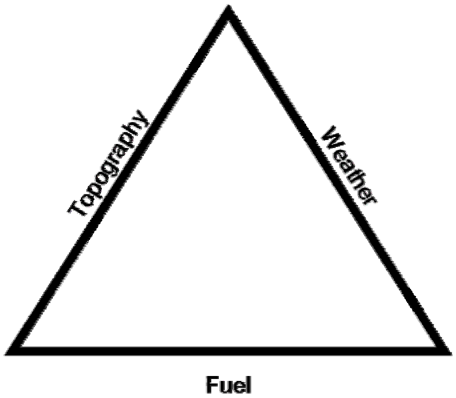

Figure 7 – The fire behavior triangle.

Fuel comprises the third leg of the fire behavior triangle. How much fuel (fuel loading in tons per hectare), and its vertical (within stands) and horizontal (across landscapes) arrangement affects fire intensity and the ability of surface flames to begin torching tree crowns or support active crown fire for a given set of weather and topographic conditions. Fuels can be comprised of dead biomass (needles, fallen branches, dried herbaceous material, and coarse woody debris) or of live trees and other vegetation, such as shrubs. Some shrubs, such as bitterbrush, contain volatile oils and have higher heat content and, because of their air-to-volume ratio, produce long flame lengths when ignited, initiating torching. Torching is the movement of surface flames up into individual tree crowns or into the crowns of tree groups. Torching is the precursor to active crown fire.

Fire exclusion over the last century in ponderosa pine forests has allowed fuels to build up on the forest floor (surface fuels) and shrub cover and tree regeneration to increase. This buildup has created “fuel ladders” where surface fuels are now connected to the overstory canopy by dense understory and mid-story saplings and medium-sized trees, making it easier for surface fires to move up and torch tree crowns and, under the right weather conditions and topographic setting, support active crown-to-crown fire spread. Crown fuels are comprised of needles, twigs, and small branches. Crown fuels are quantified by *crown bulk density*, which is the weight of needles, twigs, and small branches in kilograms per cubic meter of crown volume. Dense, even-aged ponderosa pine stands with crown bulk densities above 0.10 kg/m^3 are more vulnerable to active crown fire because fire can easily spread from tree crown to tree crown under weather and topographic conditions conducive to crown fire initiation and spread (Graham and others 1999, Agee and others 2000, Graham and others 2004).

In short, the structure of the forest and the fuels contained within have a major influence on fire behavior and severity (Agee and others 2000, Graham and others 2004, Peterson and others 2005). The amount and arrangement of fuel is the only element in the fire behavior triangle that managers have some influence over, and it is this concept that drives all fire control efforts from direct fire suppression tactics to proactive or pre-fire fuel reduction treatments.

Treatments to Reduce Fire Intensity and Severity

Keeping wildfire on the surface is important for reducing fire intensity and excessive damage to vegetation and watersheds. Factors that affect a surface fire's transition to a crown fire include foliage moisture content, surface flame length, and height to the base of the canopy (Agee 1993, Agee and others 2000). Moisture content of foliage at the beginning of the summer can be as high as 300 percent in new foliage, but declines to less than 100 percent as the summer progresses and is more easily ignited by surface flames (Agee and others 2002). In years of drought, foliage moisture content declines earlier in the season. We have no influence over foliage moisture.

Surface flame lengths depend on the amount, energy content, and moisture content of surface fuels. Removing accumulated surface fuels, or targeting the removal of specific fuels such as bitterbrush because of its high energy content, reduces flame lengths making it more difficult to initiate torching of tree crowns. In addition, the higher the base of tree crowns, the more difficult it is for surface flames to combust and torch tree crowns.

Once a fire begins torching and moving up into the canopy, the rate of spread (a function of wind speed) and crown bulk density determine the likelihood for development of an actively moving crown fire. Increasing the space between tree crowns reduces the opportunity for fire to spread from tree crown to tree crown, and allows a crown fire to transition back to a surface fire.

Following the principles of Agee (2002) (Table 3), four actions will improve fire-resilience in ponderosa pine ecosystems: reducing surface fuels, removing ladder fuels, leaving large, fire resistant trees, and spacing tree crowns (in that order). These conditions can be achieved with a variety of methods including prescribed burning, mowing, pruning and thinning.

Prescribed Burning

Prescribed burning is used in ponderosa pine stands to remove accumulated surface fuels, consume slash generated from thinning activities, kill and thin out encroaching trees in the understory, and rejuvenate herbaceous plants and shrubs (Ffolliott and others 1977, Sackett 1980, Walstad and others 1990). Prescribed burning also scorches and kills lower branches of trees, which, in the long run, results in lifting the canopy much like pruning, increasing the height from the forest floor to the lower canopy and increasing fire resistance. Periodic burning can prevent the development of ladder fuels and can be used as to maintain stands in a fire-resilient condition over time.

However, in most ponderosa pine stands prescribed burning is limited as a first-entry fuels treatment because of heavy accumulations of surface and ladder fuels. In most cases, other mechanical treatments are needed prior to prescribe burning in order to reduce fuels to a level that prescribed burning can be used in subsequent treatments without undue damage to the residual stand. Preparatory treatments, such as mowing, pruning, and thinning, improve fire control and safety, reduce the risk of escape, reduce damage to residual trees, and significantly reduce the level of smoke production and effects on air quality and human health in nearby communities.

Mowing and Mechanical Fuel Reduction

Mowing involves using a 4-wheel drive tractor or a tracked-vehicle outfitted with a mowing head. The operator essentially mows the understory shrubs and small trees (< 7 cm (3 in) diameter) reverting back to a grass-dominated understory. Mowing cuts and grinds up surface fuels and the smaller ladder fuels to small particle sizes, which decay rapidly when in contact with the forest floor. This mechanical treatment is limited to gentle topography and has been implemented on the Deschutes National Forests near wildland-urban interface areas. Mowing costs are approximately $40 per acre. Stands to be underburned can be first treated with mowing to reduce surface fuels and improve fire safety and control. Because many shrubs species resprout, mowing is a short-term fuel reduction treatment.

There are several other mechanical methods for reducing shrubs and small trees. These include various kinds of excavators outfitted with a rotating head mounted on a hydraulic arm. The operator moves the head back and forth along the ground to mulch up shrubs or smaller trees (saplings). On larger trees (10 to 20 cm dbh (4 to 8 inches)) the tree can be ground up by mulching from the top of the tree down to the base. These treatments can cost up to $350 to $400 per acre, depending on terrain and density of shrubs and small trees. However, they cut and mulch all in one pass, thus eliminating subsequent costs for piling and burning slash often associated with manual hand cutting.

Pruning

Pruning removes the lower branches of trees and lifts the crown, creating more distance between potential surface flames and the bottom of the tree canopy. Pruned branches need to be piled and burn. This technique is particularly useful in young stands where crowns are low and close to surface fuels (grass/shrubs).

Thinning

Thinning can be used to change stand and fuel characteristics (e.g., ladder and crown fuels) and lessen the chance of passive and active crown fires (Graham and others 1999, Scott 1998, Agee and others 2000, Fulé and others 2001, Graham and others 2004, Peterson and others 2005). Thinning from below, also referred to as low thinning, removes trees in the subordinate lower crown classes (Figures 8 and 9) leaving the larger, more vigorous trees. Thinning from below removes ladder fuels, reduces canopy bulk density, and leaves trees that have higher crowns, thicker bark, and better ability to survive fire. In mixed conifer stands, thinning should leave the most fire resistant species, such as ponderosa pine, western larch, and Douglas-fir. Thinning can be done incrementally such that the stand is progressively opened up over time or, if fire risk is high, thinning more heavily to a wider spacing in one operation. The latter situation would be appropriate for ponderosa pine stands adjacent to the wildland-urban interface and where the risk of wind throw is low.

Following thinning, the amount of remaining surface fuel should be assessed (Weatherspoon 1996). Where excessive slash is found to exist, slash must be removed either by piling and burning or with prescribed underburning to prevent high-severity surface fire (Brown 1980). Wildfire in thinned stands and in stands thinned in combination with other fuel treatments experience reduced fire intensity, lower rates of spread, less severe tree damage and lower overall fire severity

(Graham 2003). However, thinning also opens up the stand and changes the microclimate, allowing surface fuels to dry out more completely and within-stand wind speeds to increase (Weatherspoon 1996). These changes can increase both the rate of spread and intensity of subsequent surface fires. However, the increased intensity and spread rates are why the original forests had frequent fires. Thinning in this manner creates stands similar to pre-settlement conditions. Thus, the reduction in fire hazard with thinning generally more than makes up for potential increases in fire spread and intensity. This also makes fire suppression, when deemed necessary, more efficient. Thus, if heavy fuels are removed, the residence time (or duration) of the fire is reduced, often resulting in a non-lethal surface fire.

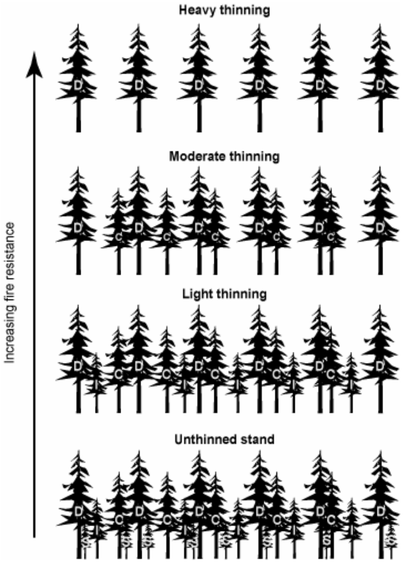

Figure 8 – Thinning to improve fire resistance. Thinning from below removes smaller trees in the stand leaving the larger, more fire resistant trees (adapted from Graham 1999).

Figure 9 – A 90-year old ponderosa pine stand thinned from below on the Sun Pass State Forest, Oregon. Note the more open canopy and high crowns. The fire resistance in this stand has been significantly improved.

Thinning from below can also benefit large, old-growth ponderosa pine trees in stands that have developed a dense understory and are under increasing competitive pressure. Thinning from below not only reduces ladder fuels and the risk of torching, but by reducing stand density tree vigor is improved and risk to bark beetle attacked reduced (Figure 10) (Fitzgerald and others 2000, Latham and Tappeiner 2002, McDowell and others 2003). Because the extent of old-growth ponderosa pine forests is limited across western North America, thinning from below to restore and maintain existing old-growth forests can better insure their survival over the long term, until other areas develop old-growth conditions and serve as replacement old-growth habitat in the future.

Figure 10 -- This old-growth ponderosa pine stand was thinned from below and later underburned to reduce competition and restore a more historic stand structure.

Thinning ponderosa pine stands (from below) to improve fire resistance typically involves removing non-commercial trees (<23 cm dbh (9 in)) and trees containing small to medium-size sawlogs (25-46 cm dbh (10-18 in)). In most cases the value and volume removed in small to medium-size sawlogs is not enough to cover all the costs of removing non-commercial trees and to conduct follow up treatments. Net costs to treat stands have ranged from $247 to $1976 per ha ($100 to $800 per acre). Thus, subsidies are needed to accomplish fuel treatments at the landscape level. However, some thinning treatments can show a modest profit, depending on stand conditions, thinning diameter, basal area limits, and markets (Scott 1998, Barbour and others 2004, Larson and Mirth 2004). This profit could then be used to offset costs in stands that do not produce any net revenues, thus increasing the overall acreage treated. Although there is resistance to paying for these treatments on federal lands through Congressional appropriations, improving fire-resilience in ponderosa pine ecosystems through subsidies is far less expensive than the cost of wildfire suppression that can approach $7410 per ha ($3000 per acre), not to mention the loss in timber value, cost of watershed rehabilitation, the risk and loss of homes, and the loss of ecosystem values and services (Butry and others 2001, Fairbanks and others 2002, Morton and others 2003, Lynch 2004, Mason and others 2004).

Table 3 -- Principles for creating fire-resilient forests (after Agee 2002).

Principle	**Effect**	**Advantage**	**Concerns**
Reduce surface fuels	Reduce potential flame length	Less torching, control easier	Surface disturbances, less with fire than with mechanical techniques
Increase height to live crown	Requires longer flame length to begin torching	Less torching, control easier	Opens up understory, may allow surface winds to increase
Decrease crown density	Makes tree-to-tree crown fire less probable	Reduces crown fire potential, control easier	Surface wind may increase and surface fuels may be drier
Keep larger trees	Thicker bark and higher crowns	Increase the survivability of trees	Removing smaller trees is less economically profitable

Summary

Over the last 140 years ponderosa pine ecosystems have changed immensely and bear little resemblance to their presettlement condition. The original old-growth ponderosa pine forests were once considered an endless resource to early pioneers and settlers, and the vast “yellow pine” forests were utilized to fuel economic growth and the development of western North America. Past and current land use activities along with active fire suppression eliminated natural surface fires from these forests and the disturbance patterns that controlled their development and helped sustained them over the millennia.

This elimination of fire has profoundly changed the structure of the original ponderosa pine forests, and not for the better. Today, ponderosa pine forests contain an overabundance of fuel, high stand densities across large landscapes and few old growth trees. These conditions have contributed to declining tree health and have helped sustain increases in large, uncharacteristic wildfires across the west. The ponderosa pine ecosystems are in trouble, and the problem will not go away or take care of itself. In the Pacific Northwest timber stand improvement activities, such as thinning, are down 60 percent compared to over a decade ago (FY 1988), and the level of funding for silvicultural treatments has declined over the last decade, resulting in a huge backlog of forest requiring some level of treatment (Powell and others 2001). However, doing nothing will result in forests that continue to deteriorate over time because wildfire today no longer operates in its historical fashion, that of frequent low-intensity surface fires.

Restoration treatments should return ponderosa pine forests to within their natural range of variation for both stand and landscape structure where possible. Ongoing research to determine reference stand conditions (density, tree size, tree pattern, gaps, etc) should establish conditions across broader landscapes, which would provide a “blueprint” for restoration activities (Covington and Moore 1994, Fulé and others 1997). Restoration also needs to re-introduce processes, like fire, to maintain stands and promote the sustainable development of younger ponderosa pine stands.

Craig and others (2002) outline in detail 16 principles to consider for the restoration of southwestern ponderosa pine. These same principles could be adapted and applied to most ponderosa pine forests of western North America. The first of these principles, and probably the most important in the near term, "reduce the threat of crown fire," is needed to *first* stop the cycle of uncharacteristic wildfires to prevent loosing critical forest structures, important wildlife habitat, and genetic reservoirs, like old-growth. Treatments that move stands closer to conditions of pre-European settlement (Table 3) are likely to reduce the chance of crown fires and improve fire-resiliency. Treatments to reduce fire intensity and severity have been shown to work (Agee and others 2000, Graham 2003, Martinson and Omi 2003, Graham and others 2004). To make a real difference at the landscape level, however, will require a suite of treatments (prescribed fire, thinning, and combinations) that are prioritized. In addition, long-term Congressional investments will be needed to treat the millions of acres of ponderosa pine forests on federal lands in need of restoration (U.S. General Accounting Office 1999).

Finally, without intelligent, science-based intervention in the near term to restore fire-resiliency, we cannot expect ponderosa pine forests of western North America to continue to produce all the ecological and social values that the public desires in the long term.

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Diseases as Agents of Disturbance in Ponderosa Pine[1]

Gregory M. Filip[2]

Abstract

Several diseases affect the growth and survival of ponderosa pine in the Pacific Northwest and serve as agents of disturbance. Probably the most widespread and damaging class of disease agents is dwarf mistletoe, which causes serious growth loss and mortality of ponderosa pine. Dwarf mistletoes (*Arceuthobium* spp.) are seed plants that can parasitize all age classes of pine. Decades of research and observation have shown that although dwarf mistletoes spread slowly within and among trees, their localized effects can be quite spectacular. Root diseases caused by the fungi *Armillaria ostoyae* (Romagn.) Herink, *Heterobasidion annosum* (Fr.:Fr.) Bref., and *Leptographium wageneri* var. *ponderosum* (Harrington & Cobb) Harrington & Cobb cause localized mortality resulting in gaps in affected forests. Root diseases may spread by root contact, airborne spores, or insect vectors depending on the species of root disease fungi. Stem decays caused principally by *Phellinus pini* (Thore.:Fr.) A. Ames and *Fomitopsis officinalis* (Villars:Fr.) Bond. & Sing. result in single-tree gaps if trees break as a result of advanced decay. Decay fungi infect wounds on living trees, and decay may take decades to develop to where tree structural integrity is compromised. Stem diseases caused by rust fungi such as *Cronartium comandrae* Peck and *C. coleosporiodes* Arth. can kill the tops of trees or result in whole-tree mortality. Decayed and cankered trees can serve as habitat for a wide diversity of wildlife. Disturbances in ponderosa pine forests caused by forest diseases can affect forest succession, insect outbreaks, fire frequency and severity, and both animal and plant diversity.

Introduction

Diseases caused by fungi and parasitic plants are important fine-scale agents of disturbance in ponderosa pine ecosystems. In the absence of periodic wildfire, the resulting dense stands of ponderosa pine often are attacked by several classes of forest diseases including root disease, stem decay, stem disease, foliage disease and dwarf mistletoe. These diseases have a tremendous effect on the structure and ecology of ponderosa pine stands. Root diseases increase windthrow of live trees, live stem breakage, top-killing, bark beetle attack, tree mortality, down wood, wildlife habitat, wildfire, stand opening and regeneration, and shifts in stand composition. Stem decays cause hollows in living stems and branches, and increase tree mortality, amount of down wood, number of hollow logs, diversity of wildlife habitat, stem breakage, and intensity and incidence of wildfire. Dwarf mistletoes are

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parasitic plants that infect conifers and result in increases in live-crown ratios, abnormal branches (witches' brooms), dead branches, top-killing, branch and stem breakage, tree mortality, down wood, wildlife habitat, and intensity of wildfire. Fuel ladders caused by dwarf mistletoe make fires more likely to crown. These fine-scale disturbances created by forest diseases can affect forest succession and alter both plant and animal diversity.

Root Diseases of Ponderosa Pine

Major root diseases of ponderosa pine in the Pacific Northwest include Armillaria root disease caused by *Armillaria ostoyae* (Romagn.) Herink, annosum root rot caused by *Heterobasidion annosum* (Fr.:Fr.) Bref., and black-stain root disease caused by *Leptographium wageneri* var. *ponderosum* (Harrington & Cobb) Harrington & Cobb. Infection may result in tree growth loss, windthrow, or mortality. Root diseases spread primarily by root to root contact between infected trees or between residual stumps and healthy trees. Root diseases are difficult to detect and require removal of soil and bark at root collars to detect signs and symptoms of pathogenic root-disease-causing fungi. The preferred tool for examining roots is often the pulaski. There is no cure for root disease. Management strategies include prevention of infection, and favoring tree species that are less susceptible or immune to root diseases on the site.

Armillaria root disease is probably the most common and damaging root disease of ponderosa pine in the Pacific Northwest, although pine is considered tolerant to Armillaria when compared to white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.) or grand fir (*A. grandis* (Doug. ex D. Don) Lindl.). Major mortality centers occur where a particularly virulent clone of the fungus is present or where pine is mixed with infected true firs. Some virulent clones of Armillaria are reported to be very old and cover over 2000 acres of infected forest in northeastern Oregon (Ferguson and others 2003). Symptoms of Armillaria root disease include chlorotic and shortened leaders and lateral branches, and basal resinosus (pitching), although the latter symptom is less common in ponderosa pine. Signs include mycelial fans under the bark at the root collar and mushrooms produced at the base of infected trees in autumn. Armillaria management involves maintaining good tree health and vigor, avoiding soil disturbance, and thinning of overstocked young stands.

Annosum root rot is most damaging in ponderosa pine in southern Oregon and northern California, usually on low-quality sites (Schmitt and others 2000). The disease is associated with tree harvesting where freshly cut stumps are susceptible to infection by windborne spores. Infection results in tree mortality or windthrow. Stumps can be protected from annosus spore infection by treating freshly cut surfaces with a boron-containing product with 48 hours of cutting. There are actually two types of the fungus: one type infects primarily true firs (S-type) and the other infects primarily pine (P-type). Usually only the drier pine sites are seriously affected by annosum root rot.

Black-stain root disease is most common in central and eastern Oregon and in the Sierras of California. It spreads underground but may also be vectored long distances by insects. More research is needed to identify the vectors and clarify their roles. Best management practices also are being identified, especially the use of prescribed fire (Filip 1999). Symptoms and signs of the disease include a basal resinosus and a black stain in the wood that occurs only in the first few annual rings.

Black-stain root disease management may include thinning or pruning pine in August and September to avoid insect flights and disease spread. More research is needed to confirm this. There are two types of the fungus: one that infects the 2- or 3-needle pines and one that infects Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco). Clearcutting of diseased areas and a buffer may help to sanitize black stain-affected ponderosa pine stands. Any resistant-appearing trees should be retained. Treated areas can be regenerated immediately since the fungus dies when the trees are cut. Prevention strategies are especially important with black stain, since alternative conifer species are often lacking on sites where this disease occurs.

Stem Decay of Ponderosa Pine

Stem decays are less serious in ponderosa pine than in other conifer species (Filip 2001). Because of the resinous nature of pine wood, most decay-causing fungi are not able to successfully colonize wound surfaces and deeper tissue. A few species, however, such as *Phellinus pini* (Thore.:Fr.) A. Ames, cause of red ring rot or white pocket rot, can infect and decay ponderosa pine. Early stages of decay result in reduced wood quality, and advanced decay results in cull logs and loss of merchantable material. Hollow logs and decayed trees, however, serve an important role as wildlife habitat for birds and mammals. Living pines with internal decay may stand for centuries and provide habitat longer than dead and decayed trees (Bull and others 1997).

Other less common stem decays of ponderosa pine include red-brown butt rot or velvet top fungus caused by *Phaeolus schweinitzii* (Fr.:Fr.) Pat., and brown trunk rot or quinine fungus caused by *Fomitopsis officinalis* (Villars:Fr.) Bond. & Sing. On sites with a high incidence of conks, management recommendations include harvest rotations less than 150 years and preventing bole wounds that give decay-causing fungi easy access for infection, while saving some decayed trees for wildlife.

Stem Diseases of Ponderosa Pine

Several species of fungi called rusts cause stem disease of ponderosa pine. The most common are western gall rust caused by *Endocronartium harknessii* (J. P. Moore) Y. Hirat., comandra blister rust caused by *Cronartium comandrae* Peck, and stalactiform blister rust caused by *Cronartium coleosporiodes* Arth. Unlike the well-known and introduced white pine blister rust (*Cronartium ribicola* J. C. Fisch.), all of the ponderosa pine rusts are native. Rust diseases can cause tree growth loss, top-kill, and sometimes tree death. Seedlings, saplings, and mature trees can be affected. Rust fungi have complex life cycles that often require two hosts to complete their life cycle (Allen and others 1996). Western gall rust, however, does not require an alternate host: pine to pine spread can occur. Although more common on lodgepole pine (*Pinus contorta* Dougl. ex Loud.), galls do occur on ponderosa pine, especially in riparian communities. Stalactiform blister rust also is more common on lodgepole than ponderosa pine. Alternate hosts for stalictiform blister rust include several species of Scrophulariaceae including giant red Indian paintbrush (*C. miniata* Dougl.).

The most damaging rust disease of ponderosa pine is comandra blister rust. It gets its common name from its alternate hosts, bastard toadflax (*Comandra umbellata* (L.) Nutt.) and other species in the genus *Comandra*. The disease can kill seedlings, but it is most commonly seen as stem cankers on young trees or top-kill on older trees. The fungus will girdle the stems of trees resulting in tree mortality or top-killing. If tops are killed, the fungus will grow down the living stem and kill branches. This is diagnostic and distinguishes top-killing by comandra blister rust from top-killing by bark beetles or animals. Management for stem rusts includes selective removal of trees with infections, favoring lightly infected trees or disease-free trees, or pruning of infected branches.

Foliage Diseases of Ponderosa Pine

There are three primary diseases of ponderosa pine foliage in the Pacific Northwest: Elytroderma needle disease, dothistroma needle blight or red band needle disease, and diseases caused by several species of fungi that are loosely termed pine needle diseases. Foliage diseases of ponderosa pine are caused by fungi that cause the loss of 2-5-year-old needles. Infection results in some growth loss, but tree mortality may occur if disease is severe. Infection severity is keyed to proper environmental conditions for the fungi and host genetics that regulate infection, colonization, and disease expression. Control is usually not warranted in forest settings.

Elytroderma needle cast or blight is caused by the fungus, *Elytroderma deformans* (Weir) Darker. Infection begins in the needle from windborne spores, then progresses to the twig. Twig infection becomes perennial and results in misshapen and upturned branches resembling an “arthritic hand.” Infections in pine resemble those caused by dwarf mistletoe, but branches with live mistletoe plants present are green or yellow rather than red or brown as with Elytroderma infections. Severely infected pines have poor form, experience some growth loss, and can be thinned to leave alternative conifer species and pines with the least infection.

Red band needle disease is distinguished by the red or brown banding pattern on infected needles rather than the whole needle being discolored. The disease is caused by the fungus *Mycosphaerella pini* Rost. in Munk. Severely infected trees have thin crowns, and when environmental conditions are favorable for infection, the disease can spread rapidly. Repeated infections over many years can result in tree death.

Other pine foliage diseases are caused by several species of fungi including *Lophodermella concolor* (Dearn.) Darker, *L. morbida* Staley & Bynum, and *Lophodermium pinastri* (Shrad.:Fr.) Chev. These fungi periodically cause discoloration of 2-3-year-old needles if environmental conditions are favorable for infection. Some tree growth loss occurs. Off-site trees are more severely affected.

Dwarf Mistletoe of Ponderosa Pine

Infection by dwarf mistletoe (*Arceuthobium campylopodum* Engel.) is probably the most serious disease affecting ponderosa pine in the Pacific Northwest. Dwarf mistletoes are higher seed plants that have stems, leaves, seeds, and a root system that colonizes branch and stemwood of living trees. Male and female plants occur with pollination in the spring, seed development throughout the summer, and seed dispersal in the fall. Seeds are explosively shot from ripe fruit and adhere to needles of the same tree or adjacent trees. Infection from overstory to understory trees commonly occurs in infested pine stands. Seeds adhere to needles and slide to the needle fascicle with fall rains. Seeds then germinate to produce a “penetration wedge" that enters the bark and subsequently becomes embedded in successive layers of xylem (Hawksworth and Wiens 1996). Once established, shoots are formed and then seeds on female plants. The entire life cycle from twig infection to seed production takes about five years.

Dwarf mistletoe infection severity (DMR) is rated on a scale from 1 to 6. DMR 1 trees have light infections only in one third of the crown, whereas DMR 6 trees have severe infections throughout the entire crown. Tree growth is reduced proportional to infection severity. DMR 6 trees can experience growth impacts such that growth is 20% of similar uninfected pine. Severe infections can cause tree mortality especially in sapling to pole-size understory trees. Because dwarf mistletoes are obligate parasites (require a living host), once the host tree has died or has been killed, the mistletoe also dies.

The abnormal branches caused by dwarf mistletoe infections (witches' brooms) are used by birds, owls, and mammals for hiding, roosting, and nesting. Although this is more common in tree species that form large, dense brooms like Douglas-fir, old infections in ponderosa pine cause brooms that can serve as wildlife habitat. Girdling trees with brooms will kill the tree and prevent mistletoe spread, but the function of the broom as wildlife habitat ends earlier than a living tree.

Although dwarf mistletoe is one of the more damaging diseases of ponderosa pine, it is one of the easiest to manage. Silvicultural techniques such as thinning, sanitation harvesting, and clearcutting have been used for decades to remove infected trees and reduce the spread of the parasite. In young pine, removing moderately to severely infected trees and creating gaps of 18 feet between residual trees have been shown to significantly reduce spread and impact of dwarf mistletoe (Barrett and Roth 1985, Roth and Barrett 1985).

Summary

Several diseases affect the growth and survival of ponderosa pine in the Pacific Northwest and serve as agents of disturbance. These include root diseases, stem decays, stem diseases, foliage diseases, and dwarf mistletoes. Forest diseases serve as fine-scale disturbance agents by causing host trees to form structures that are infrequent in disease-free stands: dead trees, down wood, abnormal branches (witches' brooms), dead branches, dead tops, and broken stems. Disease also promotes insect attack, diversifies wildlife habitat, and increases risk of wildfire.

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Bugs in the System: Development of Tools to Minimize Ponderosa Pine Losses from Western Pine Beetle Infestations[1]

Christopher J. Fettig[2]

Abstract

The western pine beetle, *Dendroctonus brevicomis* LeConte, is a major cause of ponderosa pine, *Pinus ponderosa* Dougl. Ex Laws., mortality in the western USA and particularly in California. Under certain conditions, the beetle can aggressively attack and kill apparently healthy trees of all ages and size classes. The average loss is substantial, and has been estimated at 1 billion board feet annually. The availability of pest management techniques for preventing and suppressing infestations is rather limited. Currently, we are conducting several studies on the development of chemical, silvicultural and semiochemical-based management tactics for minimizing the negative impacts associated with *D. brevicomis* infestations. This paper describes the status of four such studies. Efficacy of the insecticide bifenthrin for protecting individual, high-value trees, and of the anti-aggregation pheromone verbenone for protecting small stands, is described. In a trapping bioaasay, nine nonhost angiosperm volatiles significantly augmented the effect of two release rates of verbenone, reducing trap catches to levels significantly below that of either release rate of verbenone alone. These results suggest that the addition of nonhost angiosperm volatiles to verbenone could be important for developing successful semiochemical-based management techniques for *D. brevicomis*. In another study, we examined and described the effect of mechanical fuel reduction treatments on the activity of bark beetles in ponderosa pine stands. Many of the results presented here are preliminary in nature.

Keywords: western pine beetle, *Dendroctonus brevicomis*, ponderosa pine, *Pinus ponderosa*, insecticides, verbenone, nonhost angiosperm volatiles, semiochemicals, fuel reduction, forest health

Introduction

The western pine beetle (*Dendroctonus brevicomis* LeConte) is a major cause of ponderosa pine mortality in the western USA and particularly in California. Under certain conditions, the beetle can aggressively attack and kill apparently healthy trees of all ages and size classes. The total annual loss ranges from 500 million to 3.5 billion board feet. The average loss is substantial, and has been estimated at approximately 1 billion board feet annually (Miller and Keen 1960). Much of this mortality occurs within the larger diameter classes. Currently, the availability of pest management techniques for preventing or suppressing infestations is limited to prevention thinning or insecticide treatments.

Severe droughts, such as those occurring in California from 1975 to 1977 and from 1988 to 1993, are often accompanied by excessive amounts of *D. brevicomis*-

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caused tree mortality in both managed and unmanaged stands. High-value trees, such as those located in residential, recreational, or administrative sites, are particularly susceptible to attack as a result of increased amounts of stress associated with drought, soil compaction, mechanical injury, or vandalism (Haverty and others 1998). Regardless of landowner objectives, tree losses generally have a catastrophic impact. For example, the value of a mountain home may be severely reduced by the mortality of adjacent shade and ornamental trees (McGregor and Cole 1985). The mortality of trees located in campgrounds or other administrative sites can have long-range management impacts. The value of these individual trees, the cost of removal, and the loss of aesthetics may justify protecting individual trees with insecticides until the main thrust of an infestation subsides. This situation emphasizes the need for assuring that effective insecticides are available for individual tree protection in the future.

Several formulations of carbaryl have been evaluated and found effective for protection of individual trees from attack by western bark beetle species (Gibson and Bennet 1985, Hall and others 1982, Haverty and others 1985, Shea and McGregor 1987). These and other studies (McCambridge 1982, Smith and others 1977) led to registration of 2 percent Sevimol® for this use. In 2003, only two formulations of carbaryl, permethrin, and injected metasystox-R were registered for protection of ponderosa pine from the western pine beetle. Metasystox-R has been shown to be rather ineffective and is not recommended for use (Haverty and others 1997). Given the uncertain future availability of an effective preventative spray, I believe it is important to develop alternative insecticides for this important forestry use.

Verbenone (4,6,6-trimethylbicyclo[3.1.1]-hept-3-en-2-one) is the primary antiaggregation pheromone of the mountain pine beetle (*Dendroctonus ponderosae* Hopkins), southern pine beetle (*Dendroctonus frontalis* Zimmermann), and western pine beetle. It was first identified in males of the southern and western pine beetles, and subsequently in the hindgut of emergent and feeding female mountain pine beetles. Verbenone is naturally derived from three sources: (1) the beetles themselves, (2) auto-oxidation of α-pinene and subsequently of cis- and trans-verbenol to verbenone (Lindgren and Borden 1989), and (3) auto-oxidation from cis- and trans-verbenol to verbenone by certain microorganisms that are associated with bark beetle species. It is assumed that verbenone reduces the negative impacts of intraspecific competition by reducing the overcrowding of developing brood within the host tree.

In recent years, verbenone has been evaluated as a tool for mitigating stand losses due to bark beetle infestations. In the southern USA, verbenone is used once an infestation (termed "spot") has formed to reduce future spot growth (Clarke and others 1999). In general, by placing elution devices containing synthetic verbenone on all freshly attacked trees and a suitable buffer of uninfested trees at the active margin (termed "head") of the spot, emerging and reemerging beetles are forced to disperse in search of unoccupied hosts. Spot growth is often reduced and sometimes completely suppressed. Verbenone is now available commercially in a slow-release polyethylene pouch (Phero Tech Inc., Delta, BC) that has received U. S. Environmental Protection Agency (EPA) registration for use in forest stands containing southern pines (EPA Reg. No. 56261-CN-1 (1999)).

In the western USA, efforts have concentrated on using verbenone for small-scale stand protection. Most field evaluations have concentrated on *D. ponderosae* populations, and results have often been favorable. Verbenone released from multiple points in stands of lodgepole pine significantly reduced infestation levels in some studies (Amman and others 1989, 1991; Lindgren and others 1989, Gibson and others 1991, Shea and others 1992, Shore and others 1992, Lindgren and Borden 1993), but not others (Gibson and others 1991). In general, the application of verbenone has been unsuccessful in reducing the number of *D. ponderosae*-attacked trees in ponderosa pine (Bentz and others 1989, Lister and others 1990, Gibson and others 1991). To my knowledge, no studies have been published on the efficacy of verbenone for prevention or suppression of *D. brevicomis* infestations in ponderosa pine stands.

Volatile stimuli associated with host and non-host trees are important in mediating behavioral responses of phytophagous insects (Visser 1986). Bark beetles are believed to use a combination of host kairomones and aggregation pheromones to locate suitable host material (Borden 1985). Rejection may occur on the basis of absence of host cues or presence of non-host cues such as green leaf volatiles or angiosperm bark volatiles collectively termed nonhost angiosperm volatiles (NAVs). NAVs have been the focus of several recent studies which show that these compounds are capable of reducing aggregation in several scolytid species including *D. frontalis*, *D. ponderosae*, and *D. pseudotsugae* Hopkins (Zhang and Schylter 2004). In general, NAVs (except conophthorin) have only been effective for reducing aggregation if presented as combinations of two or more volatiles. One study has examined the effects of NAVs on *D. brevicomis* attraction, but was limited in scope to green leaf volatiles only (Poland and others 1998). I believe the system has promise for prevention and suppression of *D. brevicomis* infestations in ponderosa pine stands, and merits further investigation.

Under the National Fire Plan, the hazardous fuel treatment program has, and likely will, continue to increase in the future. One of the key goals of this program is the reduction of hazardous fuels within the wildland urban interface. In FY 2002, the goal was to treat 1.0 million hectares, and in consideration of the recent fire activity throughout the West, the protection of these communities and management of adjacent fuel loads has become paramount. At present, much of the biomass that has been removed is not merchantable as markets have yet to be developed for small dimensional timber. On many Forest Service districts in the Southwest, this material is chipped, and/or cut and lopped, and distributed on site. As a result increased amounts of host material (slash), and host volatiles (slash and chips) can attract and concentrate bark beetles within these areas. Populations can then reach high enough densities to threaten adjacent, apparently healthy trees. Trees that are killed by bark beetles after fuel reduction treatments increase fuel loads in areas not likely to be re-treated for many years. Furthermore, the cost of removal, regeneration, and long-term aesthetic losses justify the development of guidelines to mitigate such losses.

This paper describes the status of four studies conducted to develop tools for mitigating the amount of bark beetle-caused tree mortality in ponderosa pine stands. The results presented here are largely preliminary in nature.

Methods

Insecticide study.

This study was conducted on the eastern slope of the Sierra Nevada between 1400-1700 m elevation in Lassen County, California. Four treatments (insecticides) were analyzed for protecting ponderosa pine from *D. brevicomis*-attack: bifenthrin at

0.03, 0.06 and 0.12 percent AI (Onyx™); carbaryl at 2.0 percent AI (Sevin SL®); and two separate untreated controls. One control group was used to assess beetle pressure during the first summer, and the second used to assess pressure during the second summer post-treatment.

All insecticides were applied in mid-May 2003 using a trailer-mounted hydraulic sprayer (300 psi), which allowed treatment of the entire bole of each tree, until runoff, to a height of >10m (Haverty and others 1983). Two tank samples were collected and returned to the laboratory to be analyzed for concentration and purity. Treatments were applied between 0600 and 1100 when wind speeds are diminished.

Test trees were located in areas with recent beetle activity and isolated from other sample trees. Trees selected were 28 to 52cm dbh, and within 75m of an access road to facilitate treatment. Experimental trees were separated by >160m to ensure that a sufficient number of beetles would be in the vicinity of each tree to rigorously test the efficacy of these treatments (Shea and others 1984). The surrounding cover type was Ponderosa-Jeffrey (Yellowpine) Series. Forest composition, in order of decreasing abundance, was ponderosa pine, Jeffrey pine, *Pinus jeffreyi* Grev. & Balf., white fir, *Abies concolor* (Gond. and Glend.) Hildebr., and incense-cedar, *Calocedrus decurrens* Torr.

All test trees and the first set of untreated check trees were baited with *D. brevicomis* aggregation pheromone (*exo*-brevicomin, frontalin, and myrcene; WPB Tree Bait®; Phero Tech Inc., Delta, BC) for a period of 4 weeks in August 2003. The surviving treated trees in each treatment (if there are no more than 7 killed by the bark beetle challenge), and the second set of check trees were baited again in June 2004. Untreated check trees were monitored biweekly until 60 percent exhibit signs of successful attack, at which time the baits were removed from all trees.

Each treatment was randomly allocated to 35 trees (n = 210). The only criterion used to determine the effectiveness of the insecticide treatments was whether or not individual trees succumbed to attack by western pine beetle. Tree mortality was assessed in June 2004 (first baiting) and October 2004 (second baiting). The period between pheromone removal and mortality assessment was sufficient for crowns to "fade", an irreversible symptom of pending mortality. Treatments were considered to have sufficient beetle pressure if >60 percent of the untreated control trees die from beetle attack. Insecticide treatments were considered efficacious if <7 treated trees die as a result of bark beetle attack. These criteria were established based on a sample size of 30-35 trees/treatment and the test of the null hypothesis, H_O:S (survival ≥ 90 percent). These parameters provided a conservative binomial test ($\alpha = 0.05$) to reject H_O when more than six trees die (Shea and others 1984).

Verbenone study

This study was conducted over a 3 yr-period. The experimental design was a randomized complete block with two treatments (verbenone-treated and untreated) and six replicates (3 reps/treatment/block). Plots were 2 ha, square in shape, and predominately (>60 percent BA) ponderosa pine. Adjacent plots were separated by >400 m. Three replicates of each treatment were evaluated on the McCloud RD, Shasta-Trinity National Forest, and three on the Placerville RD, Eldorado National Forest.

Verbenone pouches (Phero Tech Inc., Delta, BC) were deployed twice annually on June 1 and July 1 (2002-2004) on a grid, approximately 9.1 by 9.1 m. Individual pouches were stapled at approximately 2 m in height on the bole of 250 trees per plot. Pouches from the June application were removed when the second application was made in July. On weekly intervals, one verbenone pouch per plot was removed, and stored in a deep freeze for verification of chemical purity and determination of release rates. Each pouch that was removed was immediately replaced with a fresh pouch. Western pine beetle tree baits (Phero Tech Inc., Delta, BC) were placed at the center of each plot (verbenone-treated and untreated) to create additional beetle pressure and a robust examination of this treatment (Amman and others 1989).

A 100 percent cruise was conducted pre- and post treatment on each plot in May and August (2002-2004) for the purpose of locating bark beetle infested trees. For all ponderosa pine >15 cm dbh, the number of *D. brevicomis*-attacked and -unattacked trees was recorded. All trees with evidence of bark beetle activity were geo-referenced by UTM coordinate.

Nonhost angiosperm volatile study

Twenty-seven (27) 16-unit multiple-funnel traps were deployed in a ponderosa pine forest on the McCloud RD, Shasta-Trinity NF, northern California in June 2004. Trap locations were separated by >20 m to avoid interference among adjacent treatments. Traps were hung on 3-m metal poles with collection cups approximately 1 m from the ground. A 3 x 3-cm time-released insecticidal Prozap Pest Strip (2,2-dichlorovinyl dimethyl phosphate (DDVP), Loveland Industries Inc., Greely, CO) was placed in the trap cup to kill arriving insects. Each trap location was randomly assigned one of nine treatments: 1) untreated control, 2) frontalin, *exo*-brevicomin, myrcene (WPB bait), 3) verbenone (4 mg/24h) + WPB bait, 4) (*E*)-2-hexen-1-ol , (*E*)-2-hexenal, (*Z*)-2-hexen-1-ol (**GLV**) + WPB bait, 5) **GLV** + verbenone + WPB bait, 6) benzyl alcohol, benzaldehyde, conopthorin, guaiacol, nonanal, salicylaldehyde (**BV**) + WPB bait, 7) **BV** + verbenone + WPB bait, 8) (*E*)-2-hexen-1-ol , (*E*)-2-hexenal, (*Z*)-2-hexen-1-ol + benzyl alcohol, benzaldehyde, conopthorin, guaiacol, nonanal, salicylaldehyde (**NAV**) + WPB bait, and 9) **NAV** + verbenone + WPB bait (Table 1). Treatment locations (trap and treatments) were re-randomized daily for 10 consecutive days.

Specimens were tallied, sexed, and identified using available keys (Wood 1982) and voucher specimens. The experimental design was completely randomized with nine treatments and 30 replicates per treatment. A test of normality was performed, and appropriate transformations were used when the data deviated significantly from a normal distribution. We performed a one-way analysis of variance on the pooled data (exclusive of the untreated control) collected from each trap using α=0.05 (SigmaStat Version 2.0, SPSS Inc.), and Tukey's HSD for separation of treatment means.

A second experiment was conducted in August 2004 using similar protocols. Each trap location was randomly assigned one of 12 treatments: 1) untreated control, 2) WPB bait, 3) verbenone (4mg/24h) + WPB bait, 4) verbenone (50mg/24h) + WPB bait, 5) NAV+C + Verb4 + WPB bait, 6) NAV+C + Verb50 + WPB bait, 7) NAV-C + Verb4 + WPB bait, 8) NAV-C + Verb50 + WPB bait, 9) ALD + Verb4 + WPB bait, 10) ALD + Verb50 + WPB bait, 11) ALC + Verb4 + WPB bait and 12) ALC + Verb50 + WPB bait (NAV+C includes conophthorin; NAV-C excludes conophthorin; ALD (aldehydes); ALC (alcohols)). The NAV alcohols were benzyl alcohol, guaiacol, (*E*)-2-hexen-1-ol, and (*Z*)-2-hexen-1-ol, and aldehydes were benzaldehyde, nonanal, salicylaldehyde, (*E*)-2-hexenal. The experimental design was completely randomized with 12 treatments and 30 replicates per treatment. Thirty-six (36) 16-unit multiple-funnel traps were used in this experiment.

Table 1*— Nonhost angiosperm volatiles used in Experiments 1 and 2.*

Volatile	Source	Release Device	Release (mg/24h)
benzyl alcohol	Fischer Scientific	Phero Tech bubblecap	1.3 (20 °C)
benzylaldehyde	Fischer Scientific	Phero Tech flexlure	3.5 (20 °C)
conophthorin	Phero Tech	0.4 ml polyethylene vial	3.0 (28 °C)
guaiacol	Sigma Aldrich	Phero Tech bubblecap	5.0 (20 °C)
nonanal	Sigma Aldrich	Phero Tech flexlure	3.5 (20 °C)
(*E*)-2-hexenal	Bedoukian Research	Phero Tech bubblecap	3.5 (20 °C)
(*E*)-2-hexen-1-ol	Bedoukian Research	Phero Tech bubblecap	3.8 (20 °C)
(*Z*)-2-hexen-1-ol	Bedoukian Research	Phero Tech bubblecap	3.8 (20 °C)
salicylaldehyde	Sigma Aldrich	Phero Tech bubblecap	5.0 (20 °C)
verbenone	Phero Tech	Phero Tech pouch	50.0 (30 °C)
frontalin *exo*-brevicomin myrcene	Phero Tech	250µL Eppendorf vial 250µL Eppendorf vial 1.8 mL X 2 Eppendorf	3.0 (24 °C) 3.0 (24 °C) 18.0 (24 °C)

Hazardous fuel reduction study

This study was conducted over a period of two years at three locations, and was not limited in scope to *D. brevicomis* as in the previous three studies. The experimental design was a randomized complete block with three blocks, six treatments, and 2 replicates/treatment/block. Blocking occurred on the basis of location. Plots were 0.4 ha, square in shape, and predominately (> 60 percent BA) ponderosa pine. Adjacent plots were separated by >100 m. Two replicates of each treatment were evaluated on the Tahoe National Forest, CA, Kaibab National Forest, AZ, and Prescott National Forest, AZ.

The following treatments were evaluated:

1) May chips, randomly dispersed within plot
2) May chips, raked 2 m from the root collar of all live residual trees
3) August chips, randomly dispersed within plot
4) August chips, raked 2 m from root collar of all live residual trees
5) May slash, scattered according to standard practices
6) August slash, scattered according to standard practices
7) Untreated control

A 100 percent cruise was conducted on each plot in 2003 and 2004 for the purpose of locating all bark beetle infested trees. For all pines >22.8 cm dbh, the number of bark beetle-attacked trees was recorded. Any tree containing successful bark beetle attacks was tagged, and the species and dbh were recorded. The primary variable of interest is the number of bark beetle-attacked trees occurring per unit area during the two-year period.

Results and Discussion

Insecticide study

A rigorous examination of the insecticide treatments occurred during the first baiting period as 22 of 35 untreated trees were killed by *D. brevicomis* attacks. This mortality rate exceeded the criterion established for a viable experiment (60 percent or 21 trees). Only the 0.03 percent AI bifenthrin treatment failed to meet the criterion for efficacy (12/34). The remaining treatments, specifically 0.06 percent AI bifenthrin (0/31), 0.12 percent AI bifenthrin (0/29), and 2.0 percent AI carbaryl (1/34), were effective for protecting ponderosa pine from *D. brevicomis* attacks during the first field season. Investigations are continuing to determine if efficacy remains during the second field season. This study has led to the registration and recommendation of 0.06 percent AI Onyx™ as a preventative spray.

Verbenone Study

No significant differences were observed between the number of *D. brevicomis*-killed trees in untreated and verbenone-treated plots, respectively (P = 0.84). In 2002, (mean ± sem) 6.8 ± 2.6 and 9.0 ± 3.3 trees were killed in untreated and verbenone-treated plots, respectively (P = 0.84). In 2003, 7.8 ± 2.7 and 4.8 ± 1.2 were killed in untreated and verbenone-treated plots, respectively (P = 0.33). Analysis of the 2004 data is currently ongoing and not available at this time. At present, we must conclude that the high release rate of verbenone (50 mg/24h) examined here, which represents a 6X increase over conventional release devices, does not increase the efficacy of this treatment for prevention and suppression of *D. brevicomis* infestations in ponderosa pine.

Analysis of the 2003 data from Placerville as a separate, completely randomized design results in a significant treatment effect (P = 0.07). In fact, 2X more mortality occurred on untreated plots than verbenone-treated plots. These data support our initial hypothesis that efficacy will likely vary with differences in stand structure, composition and topographic features. Forest conditions which include mixed stands of susceptible and non-susceptible species (such as Sierra-mixed conifer), increased crown cover, and vertical stratification will likely result in higher efficacy.

Nonhost angiosperm volatile study

In Experiment 1, there was no significant interaction between treatment and the gender of *D. brevicomis* ($F_{7, 464} = 0.52$; $P = 0.82$). A significant treatment effect was observed ($F_{7, 232} = 9.35$; $P < 0.001$; Fig. 1), and no significant departures from residual normality or equal variance were apparent. Significantly more female than male *D. brevicomis* were collected (paired T-test, $P < 0.001$). The ratio of males to females was 0.52.

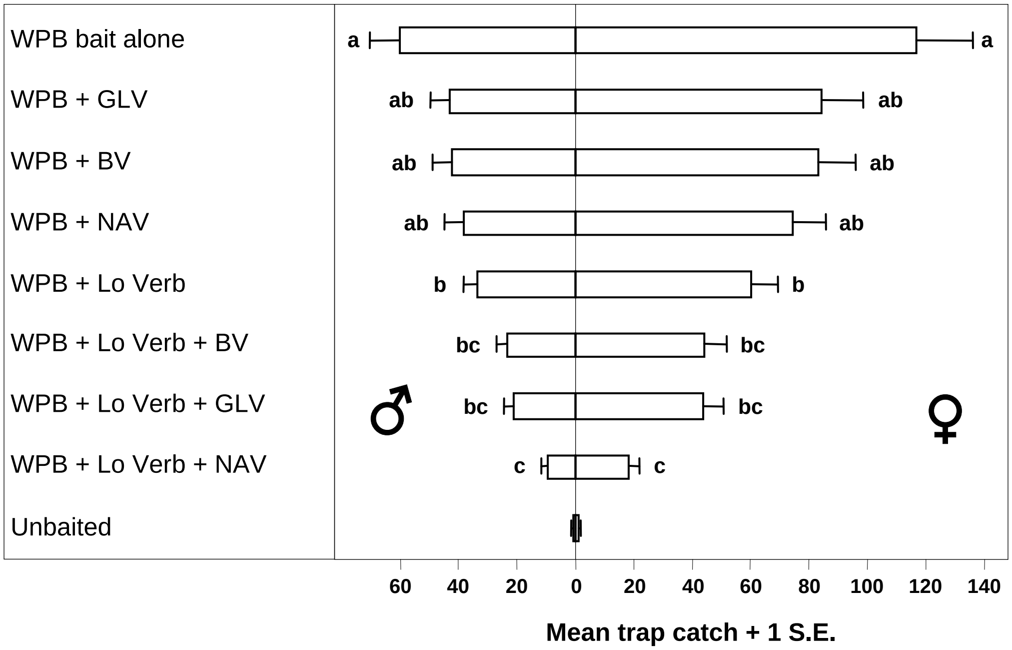

Figure 1—Disruption of western pine beetle, *Dendroctonus brevicomis* LeConte, attraction to baited (frontalin, *exo*-brevicomin, and myrcene) multiple-funnel traps during Experiment 1, McCloud Flats, Shasta-Trinity National Forest, California, June 2004. Means followed by the same letter are not statistically different (*P* > 0.05; Tukey's HSD (Honest Significant Difference) for means separation).

Based on these results, we chose to conduct a second experiment to determine: 1) if the NAV signal was overwhelmed by higher release rates of verbenone, 2) if components from the NAV blend could be removed without significantly affecting efficacy, and 3) if differences exist in the response of *D. brevicomis* to the alcohols and aldehydes in the NAV blend.

The results from Experiment 2 suggest that conophthorin can be excluded from the NAV blend without suffering a reduction in efficacy. This result disagrees with observations by other authors regarding the response of congenerics to NAV blends (Fig. 2; Zhang and Schlyter 2004). This observation is extremely important in that conophthorin is difficult to procure and is rather costly compared to the other NAVs. Future studies are planned to determine the effects of these treatments for protecting individual trees.

Hazardous fuel reduction study

The following results pertain to the Tahoe block specifically. Several bark beetle species appeared to be preferentially attracted to chipped sites. Most notable were the red turpentine beetle, *D. valens* LeConte, *D. brevicomis*, and *Ips paraconfusus* Lanier. Bark beetle activity was greater in chipped plots (19 percent of residual stems) than chipped and raked plots (12 percent). This was particularly evident in regard to the spring treatments where we observed a 48 percent reduction in tree mortality associated with raking the chips away from the base of residual trees. There was relatively little subsequent bark beetle activity in lop and scatter (3.5 percent) and untreated control (4 percent) plots. Most of the attacks in the untreated control were confined to rust-infested sugar pine, *P. lambertiana* Dougl.

Our preliminary results suggest that chipping may result in unacceptable levels of tree mortality in the Sierra Nevada, particularly when associated with hazardous fuel reduction treatments in the wildland urban interface.

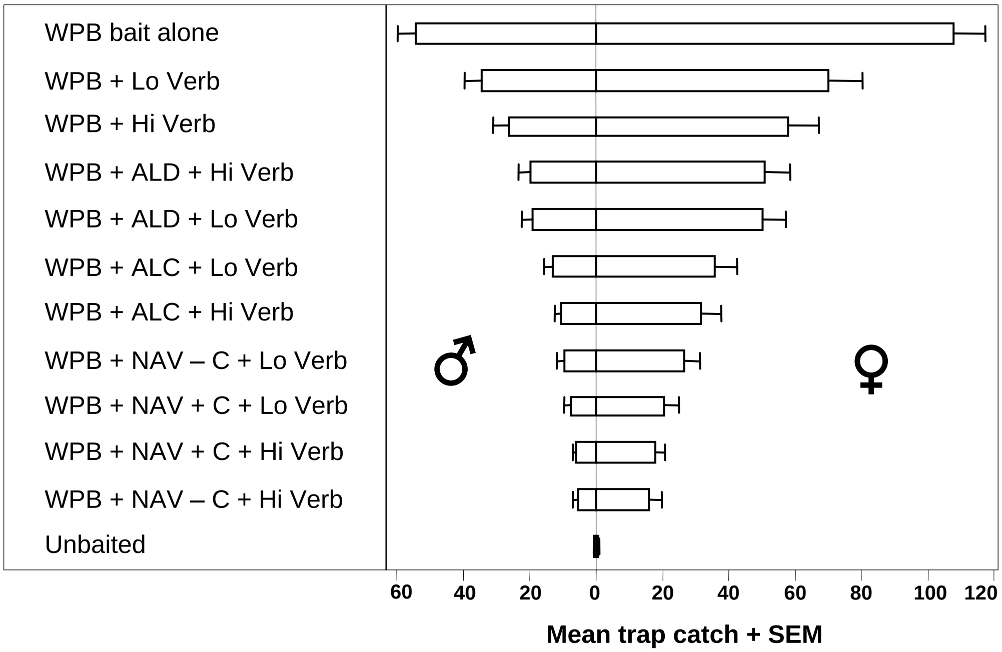

Figure 2—Disruption of western pine beetle, *Dendroctonus brevicomis* LeConte, attraction to baited (frontalin, *exo*-brevicomin, and myrcene) multiple-funnel traps during Experiment 2, McCloud Flats, Shasta-Trinity National Forest, California, August 2004. Final statistical analyses are not yet available.

Summary

This paper discusses four studies associated with the development of tools to reduce undesirable levels of tree mortality. At present, management options are rather limited in this regard. It is our hope that continued development and refinement of these techniques will eventually result in useful products for minimizing ponderosa pine losses from western pine beetle infestations.

Acknowledgments

I thank Robert R. Borys, Christopher Dabney, Stephen R. McKelvey, and L. Maria Patterson (Pacific Southwest Research Station) for technical assistance. My collaborators, Michael I. Haverty, Dezene Huber, Steven J. Seybold, Patrick J. Shea (Pacific Southwest Research Station), Joel McMillin and Sheri L. Smith (Forest Health Protection) provided valuable insight. Bayer Inc., FMC Corp., Fruit Grower's Supply Co., Univar Inc., and Sierra Pacific Industries provided valuable resources. These studies would not have been possible without the help of the Eldorado, Lassen, Shasta-Trinity and Tahoe National Forests, California.

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Reintroducing Fire Into the Blacks Mountain Research Natural Area: Effects on Fire Hazard[1]

Carl N. Skinner[2]

Abstract

Frequent, low-intensity, surface fires were an integral ecological process in the Blacks Mountain Experimental Forest (BMEF) prior to the 20th Century. With rare exception, fires have been successfully excluded from BMEF since the early 1900s. The Blacks Mountain Research Natural Area (BMRNA) covers approximately 521 acres of BMEF in 5 compartments of approximately 100 acres each. With the help of the Lassen National Forest, we have begun to reintroduce fire to BMRNA using prescribed fire. Two compartments have been burned – one in fall of 1997, the other in fall of 2000. Stand conditions and responses are being compared to two compartments where fire has continued to be excluded. The fifth compartment – mostly meadow – is not being studied at this time. Although fire hazard reduction was not a primary goal of this project, the usefulness of prescribed fire treatments for fire behavior modification is of interest to many. This paper compares the ability of the prescribed fire to alter wildfire behavior through computer simulation of expected wildfire behavior and effects for treated and untreated stands. Though the application of prescribed fire initially reduced expected fire behavior, expected fire behavior was again quite high within a few years (~ 4-6 yrs). This is due to ensuing accumulation of dead fuel from the many small trees killed in the initial burns and the inability of prescribed fire to sufficiently thin the stands for a more lasting effect. We estimate it may take up to three applications of prescribed fire to achieve a level of fire behavior modification that is similar to a single application of mechanical treatment followed by a single prescribed fire.

Key words: *Cascade Range, fire effects, ponderosa pine, prescribed fire, research natural areas, white fir*

Introduction

Frequent, low-intensity, surface fires were originally an integral ecological process in the development of relatively open ponderosa pine (*Pinus ponderosa* Laws.) and Jeffrey pine (*P. jeffreyi* Grev. & Balf.) dominated forests of northeastern California (nomenclature follows Hickman 1993). The frequency of fire occurrence varied with the spatial scale of interest. For sites < 100 acres, mean fire intervals varied from ~5 to 17 years (Taylor 2000; Norman 2002). In contrast, years of widespread fires (10,000+ acres) across broad landscapes occurred with a median fire interval of 20.5 years (range 7-49) (Norman 2002).

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[1] An abbreviated version of this paper was presented at the symposium on Ponderosa Pine: Issues, Trends, and Management, October 18-21, 2004, Klamath Falls, Oregon.

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The introduction of sheep grazing in the 1800s, followed by fire suppression and other forest management activities in the 20th Century (Taylor 2000; Norman 2002), led to great changes in the structure of ponderosa pine forests of northeastern California, similar to that described for other forests throughout the western USA (Agee 1993; Taylor 2000; Norman 2002; Youngblood and others 2004). Higher stocking densities and increased fuel accumulations have led to conditions that more readily support high-intensity fires than is likely to have been the case historically (Dolph and others 1995; Taylor 2000; Norman 2002; Youngblood and others 2004).

Reduced stand density and reduced surface fuel loads often are required to enhance forest sustainability and ecological function, and to lower the likelihood of large, high-intensity wildfires (Weatherspoon and Skinner 1996). Within wilderness areas, many parks, and other natural areas intended to be managed primarily for natural processes, the tools for achieving these objectives over large areas are largely limited to using prescribed fire – either human or lightning ignited. However, after nearly a century of increasing stand densities and surface fuel loads, reintroducing fire to large areas will not be easy nor will it likely mimic the historical fires that burned in more open stands with lighter fuel loads.

Published simulation exercises have suggested that prescribed fire should be effective at reducing fire hazard (potential for high-intensity wildfire) immediately following treatment (van Wagtendonk 1996; Stephens 1998). This reduction in fire hazard is primarily due to the reduced surface fuels – needles and small woody material. There have been few studies on the effectiveness of using prescribed fire alone to achieve more lasting fire hazard reduction (e.g., Fulé and others 2002; Fernandes and Botelho 2003). The length of years of effective fire hazard reduction will depend not only upon the longevity of reduction in surface fuels, but also on the degree to which stand density and canopy conditions have been modified. These studies indicate that the care with which prescribed fire must be applied following the many years without fire makes it difficult to both restore historical structure and reduce fire hazard in an initial application of prescribed fire.

The Blacks Mountain Research Natural Area (BMRNA), within the Blacks Mountain Experimental Forest (BMEF) in the Cascade Range of northeastern California was set aside to study natural processes in interior ponderosa pine dominated forests (Cheng 2004). The team of scientists that designed the Blacks Mountain Interdisciplinary Ecological Research Project included four compartments of the BMRNA to study the reintroduction of fire to a forest that had experienced many decades of fire suppression and accompanying ecological changes (Oliver 2001). The goal of treatments in the BMRNA is to achieve a more open stand condition, dominated by larger trees, where most of the larger trees would survive fires burning under typical summer conditions. The team reasoned that the existing stand density and surface fuel loads would require that fire be reintroduced in stages – at least two, and likely three, applications of prescribed fire – before desired conditions were likely to be achieved. Attempting to achieve desired conditions more quickly would likely result in excessive damage to the surviving stands.

The intent of this paper is to describe the changes in stand density and simulate the effects on fire hazard resulting from the first application of prescribed fire in the BMRNA. Additionally, the logic for selection of appropriate fuel models to describe expected fire behavior under the different stand conditions is described.

Methods

Study Area

The Blacks Mountain Research Natural Area is made up of five disjunct compartments of approximately 100-125 acres (40-51 ha) each within the Blacks Mountain Experimental Forest and covers a total of 521 acres (211 ha). These compartments were set aside to receive no management manipulation shortly after the BMEF was established in 1934 and were later included in the U.S. Forest Service's Research Natural Area Program (Cheng 2004).

Elevation varies from 5600 to 6400 ft (~1700-1950 m). The topography is a gentle volcanic landscape with mostly shallow soils overlaying basalt. There are no perennial streams due to rapid percolation of runoff into the porous soils. Average annual precipitation is 22.55 inches (573 mm), most of which falls as snow in winter. Infrequent thunderstorms occur in summer. Temperatures range from – 15 to 85 ^{o}F (0.6-29.4 ^{o}C). The lower elevation compartments (A, C, and E) are mostly ponderosa and Jeffrey pine stands. Compartments B and D at higher elevation have stands dominated by ponderosa and Jeffrey pine in association with white fir (*Abies concolor* [Gordon & Glend.] Lindley) and incense-cedar (*Calocedrus decurrens* [Torrey] Florin) (Cheng 2004).

The four mostly forested compartments of the BMRNA were selected for this study – A, B, C, and D. Compartment E was not included as it is mostly meadow. Compartments B and C were randomly selected for reintroduction of fire through the periodic use of prescribed fire. Initial burns were conducted in 1997 (C) and 2000 (B). Compartments A and D will not be purposely burned and will serve as long-term control comparisons to the two burned compartments.

Data Collection

Canopy Fuels

Data to describe stand structural conditions were collected on nested, fixed-area plots centered on the established grid points. As part of the Blacks Mountain Ecological Research Project, the PSW Redding Lab has established a permanently monumented 328 x 328 ft (100 x 100 m) grid in each of the four forested compartments to facilitate interdisciplinary study of spatial relationships of a variety of environmental variables (Oliver 2000). Live trees and snags > 11.5 inches (29.2 cm) dbh were measured on .2-acre (.08 ha) plots. Trees 3.5 to 11.5 inches (8.9-29.2 cm) dbh were measured on .05 acre (.02 ha) plots while seedlings (>.5" [1.3 cm] base and <4.5' [1.37 m] high) and saplings (<3.5" [8.9 cm] dbh and >4.5' [1.37 cm] high) were tallied on .01 acre (.004 ha) plots (Oliver 2000).

Pre-treatment data were collected in all four compartments to describe existing stand conditions. Following application of prescribed fire in B and C, data were again collected at the same grid points at post-fire year 1 (both B and C) and post-fire year 5 (C only).

Tree height and height to crown base data were not collected in the pre-treatment measurements in the BMRNA compartments. Therefore, these values were estimated by regressing height and height to crown base on diameter using data collected from compartments A and D that were not burned. Regressions were run for ponderosa/Jeffrey pine combined, incense-cedar, and white fir. Tree heights were constrained so that no individual live tree was estimated to be higher at pre-treatment than it was measured to be post-treatment.

Surface Fuels

The fuel inventory to describe the <3" diameter surface woody fuel conditions for fire behavior simulation has not yet been completed for the BMRNA compartments. For this paper, surface fuels were estimated by comparing fuel conditions in the compartments to available photo series (Maxwell and Ward 1980; Blonski and Schramel 1981) and then subjectively selecting an appropriate standard fire behavior fuel model (Anderson 1982; Rothermel 1983). Although there was variation in the fuel conditions within and between compartments, fire behavior fuel model 10 (Anderson 1982; Rothermel 1983) appeared to be the best fit in all four compartments for estimating pre-treatment fire behavior.

Biomass equations supplied by Powers (unpublished data at PSW Silviculture Lab, Redding, CA) were used to estimate the amount of fuel that has been added to the surface fuels from the trees killed by prescribed fire and had fallen by the 5th year post-fire measurement in compartment C.

Data Analysis

Canopy fuel conditions were estimated using program FUELCALC (Reinhardt 2004). FUELCALC uses the tree data by plot for each stand and calculates several variables necessary for estimating crown fire potential and behavior. Output includes the following variables: canopy bulk density (CBD, lbs/ft^3), plot averaged canopy base height (CBH, ft), plot averaged stand height (SH, ft)[2], total canopy fuel weight (TFW, tons/acre), plot canopy cover (COV, percent), plot basal area of trees (BA, ft^2/acre), and trees per unit area (TPA, trees/acre). The 25th percentile of canopy base height was used for predicting passive crown-fire behavior (e.g., Fulé and others 2001).

Fire weather and fuel moisture conditions that existed during a recent wildfire at BMEF were used in the fire behavior simulation. Though it did not burn within the RNA compartments, the Cone Fire burned nearly 2000 acres of BMEF in September of 2002. Fire weather conditions during this fire were as follows, 1-hour fuel moisture (%) = 1, 10-hour fuel moisture (%) = 2, 100-hour fuel moisture (%) = 2, wind speed = 15 mph (range 9 – 20) (USDA Forest Service 2002). A wind reduction factor of 0.3 was used to reduce the 20 ft (6.1 m) measured wind speed to mid-flame wind speed to account for the influence of stand structure and canopy on wind. Canopy foliage moisture content was estimated to be 75% since the Cone Fire burned under dry, north wind conditions following the long, dry summer of the area.

Fire behavior was simulated using the spreadsheet program NEXUS (Scott 1999). NEXUS quantifies fire hazard by coupling existing models of surface and crown fire behavior to simulate overall fire spread and intensity as described by (Scott and Reinhardt 2001). For this study, the NEXUS estimates of Fire Type (surface fire, passive crown fire, active crown fire), Crown Percent Burned, Rate of Spread (ROS, chains/hour), Heat/Unit Area (btu/ft^2), Flame Length (ft), Scorch Height (ft), Torching Index (wind speed when torching begins), and Crowning Index (wind speed when active crown fire begins) were used for comparing pre- and post-treatment fire behavior. *Fig. 1* shows the general steps used in providing data to NEXUS to generate the estimated fire behavior parameters.

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[2] Stand height is the average height of the five tallest trees on each plot (Scott and Reinhardt 2005).

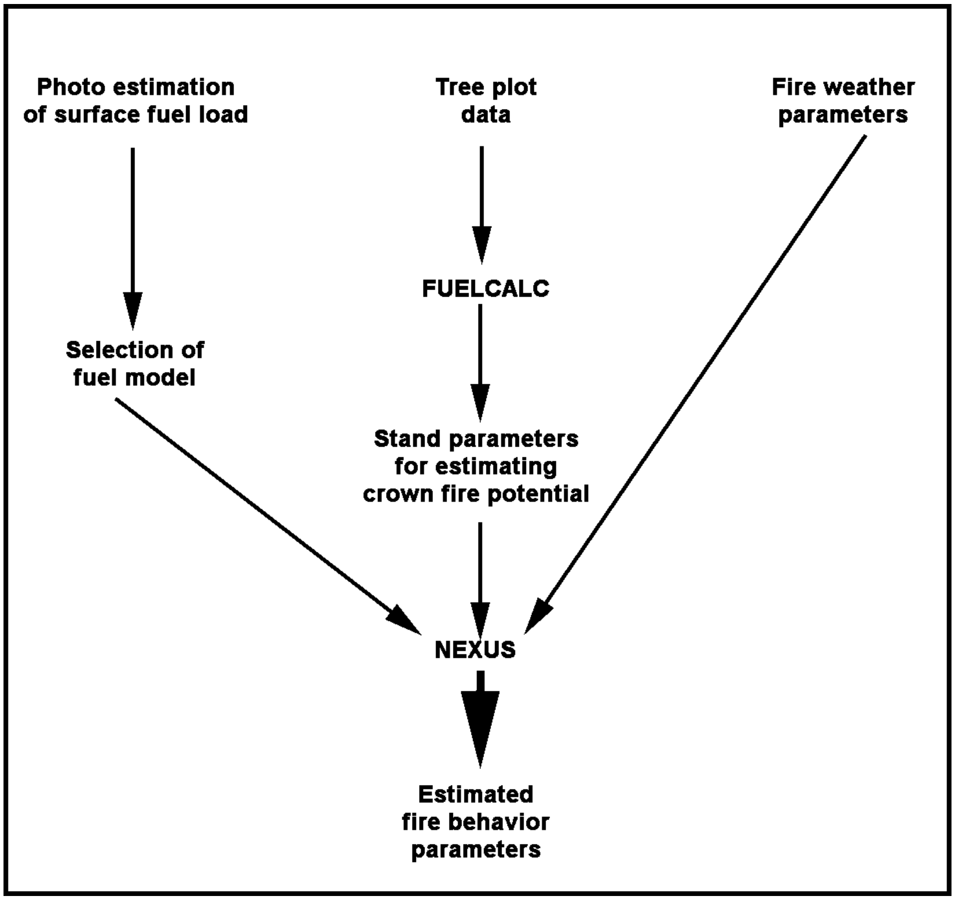

Figure 1— Steps used to provide data to NEXUS to estimate fire behavior.

Pre-treatment predictions of fire behavior were made for all four compartments. Post-treatment predictions of fire behavior were made for the first year following treatment for both of the treated compartments. Because the years of treatments were staggered, the Cone Fire occurred in the second year following treatment for compartment B and the fifth year following treatment for compartment C. Therefore, to estimate potential fire behavior had the Cone Fire been able to burn into the two treated compartments, fire behavior was predicted for the second year following treatment in compartment B and the fifth year following treatment in compartment C.

Results

Pre-treatment Stand Conditions

After many decades without fire, all four compartments had developed relatively dense stands with considerable accumulations of duff and litter (*table 1, fig. 2*). The median (25th - 75th percentile) density (trees/acre)[trees/hectare] of seedlings and saplings (<3.5” dbh [8.9 cm]) in each compartment were: A=1,050 (625-1,675) [2595 (1544-4139)], B=1,100 (600-1,800) [2718 (1483-4448)], C=750(450-1225) [1853 (1112-3027)], and D=1,150 (725-2,075) [2842 (1792-5127)].

Table 1— *Pre-treatment median (25th -75th percentile) values of plot-level characteristics for trees >3.5" DBH for the Blacks Mountain Experimental Forest RNA compartments.*

Compartment	A	B	C	D
Trees/Acre	269.9 (126.2-542.9)	439.9 (280.0-715.0)	312.4 (192.6-472.7)	497.4 (196.3-612.3)
Basal Area (ft^2) / Acre	140.3 (113.6-177.2)	160.0 (126.5-202.7)	138.6 (117.9-178.4)	158.5 (134.12-612.3)
Canopy Cover %	52.2 (33.0-60.5)	64.1 (57.5-77.4)	60.4 (48.7-72.8)	59.8 (44.4-67.0)
Stand Height (ft)	75.5 (49.2-85.3)	72.0[1] (45.0-87.0)	69.0[1] (58.5-54.0)	64.0 (46.8-86.1)
Canopy Bulk Density (lbs/ft^3)	.0021 (.0017-.0027)	.0034 (.0022-.0048)	.0018 (.0011-.0026)	.0036 (.0022-.0054)
Canopy Base Height (ft)	16.4 (13.1-32.8)	13.1[2] (13.1-16.4)	16.4[2] (13.1-16.4)	13.1 (7.0-15.6)
Canopy Fuel Load (tons/acre)	7.3 (5.7-9.6)	8.3 (5.1-9.6)	4.3 (3.7-5.3)	7.1 (6.0-10.8)

[1] Post-treatment heights were used since pre-treatment heights were not measured.

[2] Canopy base height was estimated by regressing canopy base height against dbh from trees in compartments A and D for each species.

After considering the stand conditions and fuel accumulations, fire behavior fuel model 10 (Anderson 1982; Rothermel 1983) appeared to be the best fit in all four compartments. Using the weather and fuel moisture conditions that prevailed during the first day of the Cone Fire, the NEXUS simulation predicted passive crown fire in all compartments (*table 2*).

Figure 2— A typical view of pre-treatment conditions in the Blacks Mountain RNA showing dense understory of small trees. (Photo by John Anstead, USDA Forest Service).

Table 2— *Estimates of fire behavior under Cone Fire weather conditions for the Blacks Mountain RNA compartments under pre-treatment stand conditions (Wind 15 mph).*

Compartment	A	B	C	D
Fuel Model	10	10	10	10
Fire Type	Passive Crown	Passive Crown	Passive Crown	Passive Crown
Crown % burned	14	36	11	32
ROS (ch/hr)	24	42	21	27
H/A (btu/ft^2)	2056	2048	1745	2485
Flame Length (ft)	11	18	10	15
Scorch Height (ft)	93	139	76	117
Torching Index (mph)	12	10	12	5
Crowning Index (mph)	32	22	37	31

Figure 3 – One year post-treatment. Typical scene showing retention of dead needles in standing trees that were killed by the prescribed fire. All trees in the fore- and middle-ground are dead. (Photo by Carl Skinner, USDA Forest Service)

Post-Treatment Stand Conditions

One year following prescribed-fire treatments, tree density had been reduced by 14.7% in B and 16% in C (*fig. 3, table 3*). There were corresponding decreases in basal area, canopy cover, and canopy bulk density. The relatively small proportions of change are due to mostly small trees being killed by the prescribed fire. Among small trees, seedlings and saplings were only partly reduced as can be seen by comparing the pre- and post-treatment median (25th percentile - 75th percentile) density (trees/acre)[trees/hectare] of seedlings and saplings: B was reduced by 55% one year post-treatment to 500 (300-1,300)[1236 (741-3212)] and C was reduced by 80% to 150 (75-425)[371 (185-1050)] five years post-treatment.

Canopy bulk density is reduced in the first year following prescribed fire. Yet, canopy fuel load increases in B the first year and in C by the fifth year. This is counterintuitive and may be an artifact of having to estimate the pre-treatment tree heights.

Table 3–*Comparing pre-treatment and post-treament median (25th -75th percentile) values of plot-level characteristics for trees >3.5” DBH for the Blacks Mountain Experimental Forest RNA compartments B and C.*

Compartment	B	B	C	C	C
Time	Pre-burn	Burn+1yr	Pre-burn	Burn+1yr	Burn+5yrs
Trees/Acre	439.9 (280-715)	375.1 (230-510)	312.4 (192.6-472.7)	262.4 (162-330)	285.1 (140-325)
Basal Area (ft^2) / Acre	160.0 (127-203)	149.8 (110-204)	138.6 (117.9-178.4)	125.1 (103-164)	120.4 (106-168)
Canopy Cover %	64.1 (57.5-77.4)	58.8 (52.2-71.2)	60.4 (48.7-72.8)	45.3 (35.4-50.3)	45.1 (37.0-49.8)
Stand Height (ft)	72.0[1] (45.0-87.0)	68.9 (49.2-85.3)	69.0[1] (58.5-54.0)	69.0[1] (58.5-54.0)	68.9 (50.9-86.1)
Canopy Bulk Density (lbs/ft^3)	.0034 (.0022-.0048)	.0032 (.0027-.0043)	.0018 (.0011-.0026)	.0012 (.0008-.0016)	.0017 (.0014-.0023)
Canopy Base Height (ft)	13.1[2] (13.1-16.4)	13.1 (13.1-16.4)	16.4[2] (13.1-16.4)	19.7 (19.7-25.4)	19.7 (19.7-23.8)
Canopy Fuel Load (tons/acre)	5.3 (3.8-5.8)	8.3 (5.2-9.7)	4.3 (3.7-5.3)	2.8 (2.4-3.8)	6.3 (5.7-8.7)

[1] Post-treatment heights were used since pre-treatment heights were not measured. Differences are due to trees that existed in first measurements that were down by the last measurement.

[2] Canopy base height estimated by regressing dbh from trees in compartments A and D for each species.

Fuel model 8 (Anderson 1982; Rothermel 1983) was used to predict fire behavior in the first year following treatment for both compartments because initial changes in available fuels were primarily due to reduced amounts of needles and small fuels on the surface that easily ignite and carry fire. Fuel model 8 may over-predict fire behavior immediately post-treatment. However, of all of the available standard fuel models, model 8 predicts the lowest fire behavior. Additionally, fuel model 8 was considered appropriate because post-treatment data indicated that approximately 60% of the surface was still covered by fairly compact litter and duff, even though nearly all of the young, loose, highly flammable needles had been consumed. The changes in fuel conditions due to the prescribed-fire treatments initially reduced the predicted fire behavior in both treated stands considerably (*table 4*). Predicted behavior in both B and C was reduced from a high-intensity, passive crown fire to a low-intensity surface fire. Predicted spread rate, energy released, flame length, scorch height, and potential for passive crown fire (torching index) were all reduced in both compartments. However, it is important to note that the potential for the stands to carry a crown fire once it is initiated (crowning index) was generally not affected. This is because canopy bulk density was not appreciably affected by the treatments (*table 3*).

Table 4–*Comparing estimates of fire behavior under Cone Fire weather conditions pre- and post-treatment for Blacks Mountain RNA compartments B and C (Wind 15 mph).*

Compartment	B	B	B	C	C	C
Time	Pre-burn	Burn+1 yr	Burn+2 yrs	Pre-burn	Burn+1 yr	Burn+5 yrs
Fuel Model	10	8	9	10	8	11[1]
Fire Type	Passive	Surface	Surface	Passive	Surface	Surface
Crown % burned	36	0	0	11	0	0
ROS (ch/hr)	42	3	13	21	3	10
H/A (btu/ft^2)	2048	263	536	1745	263	1128
Flame Length (ft)	18	2	4	10	2	5
Scorch Height (ft)	139	2	20	76	2	30
Torching Index (mph)	10	119	24	12	192	35
Crowning Index (mph)	22	33	33	37	90	53

[1] Had fuel model 10 been used here instead of 11, the post-treatment predicted fire behavior would have been equal to that of pre-treatment by the fifth year.

Since the initial reduction in predicted fire behavior was mostly due to the reduction of needles and small twigs and not due to significant reduction of stand density or raising the canopy base height, fire hazard began to return quickly. Five-year re-measurement data were not yet available for compartment B. In order to account for changes over time in expected fire behavior due to accumulating needles and small twigs, fire behavior model 9 (Anderson 1982; Rothermel 1983) was used to predict expected fire behavior for two years post-treatment. Two years post-treatment was chosen since few of the small trees that were killed by the prescribed fire had fallen to become part of the surface fuels and was the condition of compartment B in the year of the Cone Fire. The dead, small trees began falling in large numbers in year four. Thus, this gives a prediction of the fire behavior that would have been expected had the fire been able to burn into compartment B.

The second year prediction of fire behavior for compartment B suggests that the initial dramatic reduction in fire behavior will likely not last long. The Torching Index, which is most sensitive to the surface fuel load, was reduced from 119 mph (192 km/hr) immediately following treatment to 24 mph (39 km/hr). The second year figure is close to the maximum wind speeds measured during the Cone Fire and suggests that a danger of passive crown fire is likely to return as more fuel is added to the surface when the small, dead trees fall. The stand density was not reduced sufficiently to affect the Crowning Index which is most sensitive to canopy bulk density once the crowning phase of the fire has begun.

Fuel model 11 (light logging slash) was selected for simulating fire behavior for the fifth year post-treatment in compartment C for several reasons:

- Model 8 (slow spreading fires in compact litter beds) no longer applied due to the accumulation of needle cast and the fallen, small, dead trees.
- Model 10 (timber stands with large, down, dead fuels and understory ladder fuels) did not apply due to the effects of the prescribed fire – a) the prescribed fire consumed much of the heavy dead, down woody fuel, b) broke up continuity of surface fuels, and c) reduced the live, woody ladder fuels.
- Model 9 (long needle litter with limited down dead and no live understory ladder fuels) was not appropriate because a) fuel continuity was reduced by the prescribed burn and b) the accumulation of woody fuel resulting from the fallen small, dead trees killed by the prescribed burn exceeded that appropriate for the model 9.
- Model 11 (light logging slash) represents a) the accumulation of woody fuel over the 5 years since the prescribed burn and b) the limited number of years of accumulation of litter and needle cast since the prescribed burn. Model 11 predicts slower rate of spread but higher intensity than model 9.

Figure 4 – Four years post-treatment. Typical scene showing that many of the smaller trees killed by the prescribed fire have fallen and added to the surface fuel bed. (Photo by Carl Skinner, USDA Forest Service)

Fire behavior predictions based on using fuel model 11 to represent the fifth year post-treatment conditions for compartment C indicate that the initial dramatic reduction in fire hazard has largely been lost by the fifth year after the first application of prescribed fire. Several factors contributed to the rapid rebound of fire hazard. First, the needles, twigs, and small branches killed (scorched) in the burn have mostly dropped to the forest floor by the fifth year. Second, the many live trees remaining in the stand continue to annually contribute needles and small twigs to the forest floor. Thus, the ability for the fuel bed to ignite and carry fire is quickly regained. Third, many of the smaller trees killed outright by the prescribed fire had fallen and become part of the surface fuel load by the fifth year (*fig. 4*). Thus, an average of 8 (range 0-33) tons/acre [18 (0-82) Mg/ha] were estimated to have been added to the fuel bed. Fourth, stand density was not affected sufficiently to have a lasting effect on the potential for crown fire once surface fuels began to rebuild.

Conclusion

This simulation exercise indicates that fire hazard (the potential for high-intensity wildfire) was immediately reduced following an initial treatment with prescribed fire compared to pre-treatment conditions in the Black Mountain Research Natural Area. However, the reduction in fire hazard from a single burn appears to be short-lived for the following reasons: a) the prescribed fire did not appreciably affect canopy bulk density because only a portion of the smaller trees in the understory were killed while most intermediate and larger trees survived, b) though surface fuels were initially reduced, the scorched material in dead and living trees and annual shedding of needles and small twigs by surviving trees rebuilt the surface fuel bed within a few years, and c) the small trees that were killed subsequently fell and added to the surface fuel bed within five years of treatment. As field data become available for a more accurate portrayal of surface fuel condition, the results are likely to change somewhat. However, the relative differences over time are likely to remain.

Additional prescribed fires will be necessary to achieve reduction of fire hazard lasting more than a few years.

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Three Studies on Ponderosa Pine Management on the Warm Springs Indian Reservation: Stocking Control in Uneven-aged Stands, Forest Products from Fire-damage Trees, and Fuels Reduction[1]

John V. Arena[2]

Abstract

Over 60,000 acres of ponderosa pine (*Pinus ponderosa* P. and C. Lawson) forest on the Warm Springs Indian Reservation (WSIR) in Oregon are managed using an uneven-age system. Three on-going studies on WSIR address current issues in the management of pine forests: determining levels of growing stock for uneven-age management, fire effects on wood suitability for engineered wood products, and mechanical forest fuel reduction alternatives. To evaluate various levels of growing stock, WSIR installed twelve 1.01-hectare plots in four areas of the ponderosa pine forest to test three basal area density levels: 8.0 m^2ha^{-1} (35 ft^2ac^{-1}), 11.0 m^2ha^{-1} (48 ft^2ac^{-1}) and 14.0 m^2ha^{-1} (61 ft^2ac^{-1}). All plots were measured four times over a fifteen-year period. Preliminary results show the variation in periodic basal area growth. Future treatments are planned for 2005 to impose new density levels for the next 20 years. A study of fire effects on wood quality was initiated to compare the strength of recently burned small-diameter ponderosa pine to the strength of unburned, green, small-diameter ponderosa pine. Small logs were shipped to Mississippi State University to test their performance as engineered wood products. A mechanical forest fuel reduction study was designed to compare the effectiveness of two methods for reducing understory conifers, shrubs, and downed fuel. Soil disturbance (both visual classification and soil compaction), machine productivity, system cost, and their ability to treat certain various fuel types will be compared in an 32 ha. (80 ac.) ponderosa pine unit.

Introduction

The Warm Springs Indian Reservation (WSIR) in Oregon has over 100,000 ha (250,000 ac) of commercial forest with approximately 24,000 ha (60,000 ac) of ponderosa pine (*Pinus ponderosa* P. and C. Lawson) type. These forests are managed to provide sustainable yield and associated income, as well as wildlife and cultural amenities, to the Confederated Tribes of Warm Springs. Timber harvesting and fire suppression have changed the structure of these ponderosa pine forests from stands dominated by large, old ponderosa pine (at least 200 years old) to stands dominated by a mixture of scattered old trees and dense groups of young trees (Weaver 1959). This new forest structure presents several new challenges: (1) choice of level of growing stock and stand structure for uneven-age management; (2) utilization of small-diameter ponderosa pine trees; and (3) development of economically efficient

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methods to reduce the potential of stand-replacement catastrophic fire in ponderosa pine forests.

To begin to address these challenges the following three studies were initiated on the WSIR in the ponderosa pine forests: determining levels of growing stock for uneven-age management, fire effects on wood suitability for engineered wood products from small-diameter trees, and mechanical forest reduction alternatives. In this paper a brief overview of each study is presented.

Determining Levels of Growing Stock for Uneven-age Management

Early selection harvests in the ponderosa pine forest occurred in the 1950s and 1960s (Logan 1981). These harvests selectively removed the oldest, large-diameter ponderosa pine of low vigor based on the Keen tree classification (Keen 1943). These selection harvests evolved into treating all diameter classes as prescribed in the Forest Management Implementation Plan 2002 – 2011 (Arena 2003) to meet the target levels of growing stock. These areas are in the ponderosa pine management group, which includes four plant associations (Marsh and others 1987).

In 1964 Earle Wilcox with the Bureau of Indian Affairs (BIA) stressed the need to determine levels of growing stock for the fully regulated forest under future intensive management of ponderosa pine (Wilcox 1964). His focus was on enhancing growth of young ponderosa pine and improving growth predictions for the allowable cut. The focus changed in 1982 from growth rates to optimum levels of growing stock for younger ponderosa pine. Wilcox met with forest managers of large reservations in the Northwest to determine interest in installing permanent stocking study plots (Sassaman 1982). The data from these plots were intended to help managers advise the tribes about alternatives for managing their ponderosa pine forests.

In response to this need, the WSIR installed stocking study plots in 1984. Four sites were selected (*fig.1*). Initially, there was some difficulty locating uneven-aged stands with three age classes that were less than 120-years-old, as well as areas with similar site quality that were disease free. Dwarf mistletoe reconnaissance plots found infection from western dwarf mistletoe (*Arceuthobium campylopodum* Engelmann) scattered through the ponderosa pine forest. Three plots were established at each of the four study sites. Each plot was 1.01 ha (2.5 ac) or 100.58 by 100.58 m (330 by 330 ft) with an exterior buffer 20.12 m (66 ft) wide. The plots were divided into 25 square subplots of 20.12 by 20.12 m (66 by 66 ft). Each plot was randomly assigned a different basal area stocking level yielding three basal area stocking levels at each site. The levels of basal area per acre were: 8.0 m^2ha^{-1} (35 ft^2ac^{-1}), 11.0 m^2ha^{-1} (48 ft^2ac^{-1}) and 14.0 m^2ha^{-1} (61 ft^2ac^{-1}). The selection of these levels was based on work from the Flathead Indian Reservation, where a similar study was installed in 1970. Unfortunately, with the exception of the Beachkomb and Upper Tenino plots, initial levels of growing stock did not match the planned levels (*table 1*).

Warm Springs Forest Products Industries harvested the stocking study plots and buffers between the fall of 1984 and the spring of 1985. The logging slash was piled and burned. The initial measurement was completed in the fall of 1985, and subsequent remeasurements were scheduled every 5th year. All trees greater than 1.5 inch in diameter at breast height (dbh) were tagged and numbered. Tree heights, 5-

year dbh growth, age, and growth basal area were recorded for every tree greater than 30 cm (12 in) dbh, every third tree in the 15-25 cm (6-10 in) dbh class, and every fourth tree in the 5-10 cm (2-4 in) dbh class. Growth basal area (GBA) is the basal area at which dominant trees grow 2.5 cm (1 in) in diameter per decade at age 100. (Hall 1987)

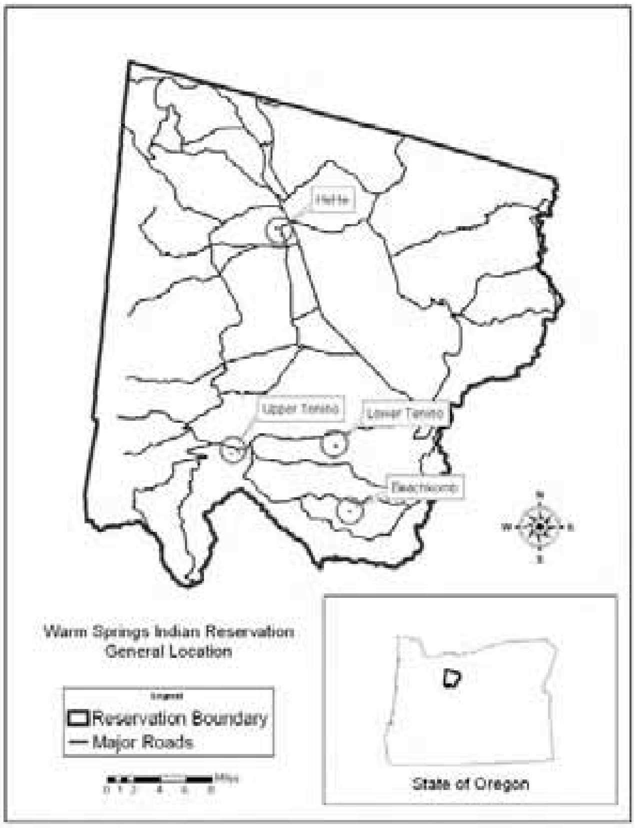

Figure 1— The location of the stocking study plots on the Warm Springs Indian Reservation.

Additional measurement variables that were collected since 1991 included all tree heights, height to lowest live limb, tree problem and severity codes, and crown class as well as a regeneration plot in the center of all sub-plots. In addition, all tagged trees were stem-mapped on each plot. The spatial data were entered into the Warm Springs Geographic Information System database. The University of Idaho remeasured the plots in 1994 as part of a Global Warming Project with WSIR and the Bureau of Indian Affairs Northwest Regional Office. The last remeasurement was done in l999-2000.

In 1991 each plot was classified by plant association (Marsh and others 1987). Two of the four study sites were located in areas of relatively high site quality; the HeHe site was in the ponderosa pine-Douglas-fir/snowberry plant association, and the Upper Tenino site was in the mixed conifer/snowbrush plant association. In contrast, Lower Tenino and Beachkomb sites were of lower site quality and were in the ponderosa pine/bitterbrush – greenleaf manzanita plant association and ponderosa pine/bitterbrush plant association, respectively.

Growth monitored over 15 years suggests that periodic rates of basal area growth differed among plots (*table 1*). This difference could be seen when comparing plot basal area growth at the Beachkomb and Upper Tenino sites, the sites at which actual basal areas were closest to the planned targets. At the Beachkomb site, basal area periodic annual increments were: 0.243 m^2ha^{-1} (1.06 ft^2ac^{-1}) for plot 1 (residual basal area 7.34 m^2ha^{-1} (32 ft^2ac^{-1})); 0.321 m^2ha^{-1} (1.40 ft^2ac^{-1}) for plot 2 (residual basal area 13.77 m^2ha^{-1} (60 ft^2ac^{-1})); and 0.259 m^2ha^{-1} (1.13 ft^2ac^{-1}) for plot 3 (residual basal area 10.10 m^2ha^{-1} (44 ft^2ac^{-1})). In contrast, at the Upper Tenino site, basal area periodic annual increments were: 0.734 m^2ha^{-1} (3.20 ft^2ac^{-1}) for plot 1 (residual basal area 8.72 m^2ha^{-1} (38 ft^2ac^{-1})); 0.613 m^2ha^{-1} (2.67 ft^2ac^{-1}) for plot 2 (residual basal area 13.08 m^2ha^{-1} (57 ft^2ac^{-1})); and 0.659 m^2ha^{-1} (2.87 ft^2ac^{-1}) for plot 3 (residual basal area 15.14 m^2ha^{-1} (66 ft^2ac^{-1})).

Table 1–*Basal area (ft^2 ac^{-1}) of each plot at each remeasurement. Target basal areas are in the 1982 column.*

	Basal area, m^2ha^{-1} (ft^2ac^{-1})				
Plot	**1982 Target**	**1985 Actual**	**1991 Actual**	**1994 Actual**	**2000 Actual**
HeHe					
1	8.0 (35)	9.9 (43)	12.4 (54)	14.0 (61)	17.7 (77)
2	11.0 (48)	9.9 (43)	12.2 (53)	13.8 (60)	17.2 (75)
3	14.0 (61)	9.4 (41)	11.5 (50)	12.6 (55)	14.9 (65)
Upper Tenino					
1	8.0 (35)	8.7 (38)	12.4 (54)	15.1 (66)	19.7 (86)
2	11.0 (48)	13.1 (57)	16.1 (70)	17.9 (78)	22.3 (97)
3	14.0 (61)	15.1 (66)	18.4 (80)	20.2 (88)	25.0 (109)
Lower Tenino					
1	14.0 (61)	8.3 (36)	10.3 (45)	11.5 (50)	14.2 (62)
2	8.0 (35)	6.9 (30)	8.3 (36)	8.9 (39)	11.0 (48)
3	11.0 (48)	8.3 (36)	10.1 (44)	11.0 (48)	13.3(58)
Beachkomb					
1	8.0 (35)	7.3 (32)	8.7 (38)	9.6 (42)	11.0 (48)
2	14.0 (61)	13.8 (60)	15.8 (69)	16.8 (73)	18.6 (81)
3	11.0 (48)	10.1 (44)	11.9 (52)	12.6 (55)	14.0 (61)

The next phase of this study will be to remeasure the trees, tag and measure the ingrowth, and then harvest the plots and their buffers to the new basal area targets: 8.0 m^2ha^{-1} (35 ft^2ac^{-1}), 11.5 m^2ha^{-1} (50 ft^2ac^{-1}) and 14.9 m^2ha^{-1} (65 ft^2ac^{-1}). These levels are similar to the original targets but will expand the range of densities to test the degree of variation in growth by regeneration and merchantable trees, especially at the higher density. The higher level (14.9 m^2ha^{-1} (65 ft^2ac^{-1})) is just above the prescribed level used in the WSIR Forest Management Implementation Plan 1992 – 2001 (Donaghu 1993).

Besides the new density levels, the silvicultural prescription for each plot will project a flatter reverse J-shape curve, shifting basal area from the lower 10-cm (4-in) dbh classes (sub-merchantable classes) to the small sawlog classes. Using tree data formatted for the Forest Vegetation Simulator (FVS) (Wykoff et al. 1982) and the spatial information from the stem-mapped plots, several marking scenarios can be assessed with respect to how well they meet the planned initial stocking levels, as well as establish an expected yield after 20 years. After treatment, the plots will be remeasured every 5 years and harvested every 20 years.

Unfortunately, the HeHe plot #3 burned in a wildfire in 1999 and only four trees remain in the plot. This plot will be remeasured every 5 years to assess post-fire recovery. The number and intensity of recent fires are a major concern on the WSIR in the ponderosa pine forest and, specifically, the stocking study plots.

The maintenance and remeasurement of these plots are long-term investments by the tribes. As the responses to different stocking levels are analyzed and compared, the desired stocking levels and stand structure can be prescribed to ultimately meet the tribal goals and objectives in these ponderosa pine forests. The process of remeasurement, harvest and analysis will continue since more questions will inevitably arise, as they did in 1964 during Earle Wilcox's silvicultural assessments. These plot data when entered into FVS can also be used to calibrate growth projections to the Warm Springs Indian Reservation.

Fire Effects on Wood Suitability for Engineered Wood Products from Small-diameter Trees

Overstocked stands of small-diameter conifer trees in areas that historically carried frequent fires have created problems for landowners in central Oregon and on the WSIR. The cost of thinning and slash treatment necessary to release and protect these stands is very high. If left untreated, these stands may contribute to and be consumed by catastrophic wildfires, resulting in subsequent loss of any wood value, as well as possible site degradation. The solution to this problem might be to develop new products or add value to products manufactured from small-diameter trees. At present, there are limited markets for either burned or green small-diameter trees. This study will test samples of recently burned and green small-diameter ponderosa pine and lodgepole pine for their utility in manufacturing engineered wood products at Mississippi State University (MSU). Eini Lowell and Susan Stevens Hummel from the USDA Forest Service Pacific Northwest Research Station are the principal investigators of this study[3]. Both burned and green ponderosa pine samples came from the southern portion of the WSIR where the B & B Fire burned in 2003. The lodgepole pine samples came from the Deschutes National Forest, Oregon.

In the fall of 2004, trees were selected, cut, and bucked into 2.4-m (8-ft) lengths by two Warm Springs engine crews. Each butt log was tagged, measured, loaded onto a trailer and shipped to MSU. The size of the trees ranged from 7.6 to 20.3 cm (3 to 8 in) dbh. The green trees were selected from a stand near the B & B Fire.

If engineering wood products from presently submerchantable ponderosa pine and lodgepole pine removed in thin-slash-burn treatments meet performance standards, forest managers will have new utilization and silvicultural opportunities. Depending on the cost of extraction and available manufacturing facilities, land managers could produce revenue or at least partly offset the cost of treatment, enabling them to increase the total area that can be treated. Utilization of this wood

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also creates an opportunity to develop scale-appropriate technology for employment through value-added wood products for the WSIR.

Mechanical Forest Fuel Reduction Alternatives for Forest Managers: Non-commercial Component

The ponderosa pine forests of the Warm Springs Indian Reservation are overstocked with young trees. The understory is mostly trees of non-commercial size. This condition predisposes forests to stand-replacement catastrophic wildfire. An Oregon State University, College of Forestry, research study is assessing mechanical options for fuels reduction through mulching and masticating small, severely suppressed, understory trees and surface fuel. The principal investigators are Chad Bolding and Loren Kellogg from the Department of Forest Engineering[4]. Two types of equipment are being studied: (1) a flexible tracked skidding machine with a masticating head and, (2) an excavator based swing-boom machine equipped with a rotary disk mulching head.

The objectives of this study were to assess: (1) effects of treatments on the soil, (2) cost and production rates of the equipment, and (3) apparent efficacy of treatment in altering future fire behavior. Visual soil disturbance classification, and soil strength (compaction) will be compared before and after treatment. The level of productivity per scheduled machine hour will be evaluated through both shift-level and detailed time studies. Productivity observations will determine operating costs per unit volume and area for each machine. The study will also attempt to determine the type or category of forest fuel that each machine can effectively treat (range of dbh classes, dead and down material, shrub component, etc.). The effectiveness of the machines for implementing the silvicultural prescription will be compared by simulation evaluating pre- and post-treatment data with the fire and fuels extension (FFE) to the forest vegetation simulator (FVS) (Reinhardt and Crookston 2003).

The 32-ha (80-ac) study area is divided into 16 2-ha (5-ac) subunits in the southeast portion of the ponderosa pine forest on the WSIR. Each piece of equipment will treat approximately 16 ha (40 ac), comprising eight randomly selected subunits. The specific silvicultural prescription outlines the density level by diameter class for the residual stand, as established in the Warm Springs Forest Management Implementation Plan 2002-2011 (Arena 2003). Tree groups are scattered throughout the study area, ranging from patches of old-growth ponderosa pine with an understory of bitterbrush (*Purshia tridentata* (Pursh) DC.) and greenleaf manzanita (*Arctostaphylos patula* Greene) to patches of young ponderosa pine within a narrow range of diameters. The diameter class of each group will determine the tree spacing guideline, but all trees greater than 28 cm (11 in) dbh are designated as leave trees. The shrub layer of bitterbrush and greenleaf manzanita will also be treated. Operators are not permitted to use any other equipment besides the two being compared to complete their specified subunits.

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[4] Chad Bolding, PhD Graduate Student, and Dr. Loren Kellogg, Oregon State University, College of Forestry, Department of Forest Engineering, Corvallis, OR

Summary

These three studies address the need for more information concerning management in the ponderosa pine forest. As mentioned earlier, the long-term investment in the stocking study plots should yield significant returns in the form of better stocking guidelines that the WSIR can apply in managing their ponderosa pine forest. These plots will help the Confederated Tribes identify options that best meet the diverse needs of their members. Any innovative value-added wood products may offer potential for both new manufacturing jobs and cost-effective ways to reduce overstocking of small-diameter conifer trees. The mechanical fuel reduction study examines treatment options and associated soil impacts, productivity, and costs in different fuel types, and should thereby help forest managers select the best treatment option for various stand and fuel conditions in these ponderosa pine stands.

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Lessons Learned on 50,000 acres of Plantation in Northern California[1]

Jeff Webster and Ed Fredrickson[2]

Abstract

Many lessons have been learned during reforestation of large wildfires and clearcuts in interior Northern California, a region of low rainfall and summer drought typical of a Mediterranean climate. Challenges appeared from time of establishment right up to commercial thinning. Establishment issues included procurement of improved seed, site preparation, soil mitigation, seedling performance, season of planting, and timing of vegetation management. Important decisions also had to be made with regard to mechanical vs. hand pre-commercial thinning (PCT), achieving a proper balance between reducing fire risk and compacting the soil, and prescribing spacing guidelines for thinning operations. On some sites damage from *Eucosma sonomana* (Kearfott) (western pine shoot borer) was a factor.

Introduction

From 1992 to 2004, Roseburg Resources Company in interior Northern California accumulated considerable experience while practicing intensive silviculture on company lands covering approximately 315,000 ac (128,000 ha) (*fig. 1*). Of the approximately 50,000 ac (20,000 ha) of plantation on these lands, two-thirds was replanted after wildfire. The remaining plantation acreage was the result of a shift in management emphasis over the last decade towards even-age management.

Northern California has a Mediterranean climate, with hot dry summers that present numerous silvicultural challenges. The climate also fosters a high fire frequency, with historical fire intervals of eight to twelve years (C. Skinner, USDA-FS Pacific Southwest Research Station, pers. comm.). The elevation of Roseburg Resources property ranges from 1,500 to 7,000 ft (500-2100m), rainfall ranges from 20 to 100 inches (50-250 cm) per year, and site class ranges from one to five, with the average being site 3 (Dunning, 1942). The vast majority of the ownership is Sierra Mixed Conifer, with smaller areas of Eastside pine, Ponderosa pine, Douglas-fir and White fir (Mayer and Laudenslayer, et al., 1988).

Reforestation practices on private land in Northern California have evolved over the last 25 years in response to experience gained on burned over areas. On Roseburg ground massive reforestation started with the Pondosa Burn in 1977. Other burns included the Day fire (1989), Fern fire (1990), and Finley Fire (1990). The lessons learned from these fires were aggressively applied to the Fountain Fire (FF) that started on August 20, 1992. This fire burned 64,000 ac (26,000 ha), 99 percent on private property, and included 41,000 acres on industrial ownership (Roseburg, Sierra

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[1] An abbreviated version of this paper was presented at the symposium on Ponderosa Pine: Issues, Trends and Management, October 18-21, 2004, Klamath Falls, Oregon

[2] Jeff Webster is Forest Manager for Total Forestry in Anderson, CA 96007

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Pacific and Fruit Growers Supply Co.). One percent of the industrial land base in California was consumed in the Fountain Fire, and 50,000 ac (20,000 ha) burned in two days with an intensity rarely seen before. On Roseburg property, the planting mix was 70 percent ponderosa pine (*Pinus ponderosa* P.& C. Lawson), 20 percent Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco) and 10 percent White fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.). The rest of this paper will describe the many lessons learned from these reforestation efforts and subsequent silvicultural activities.

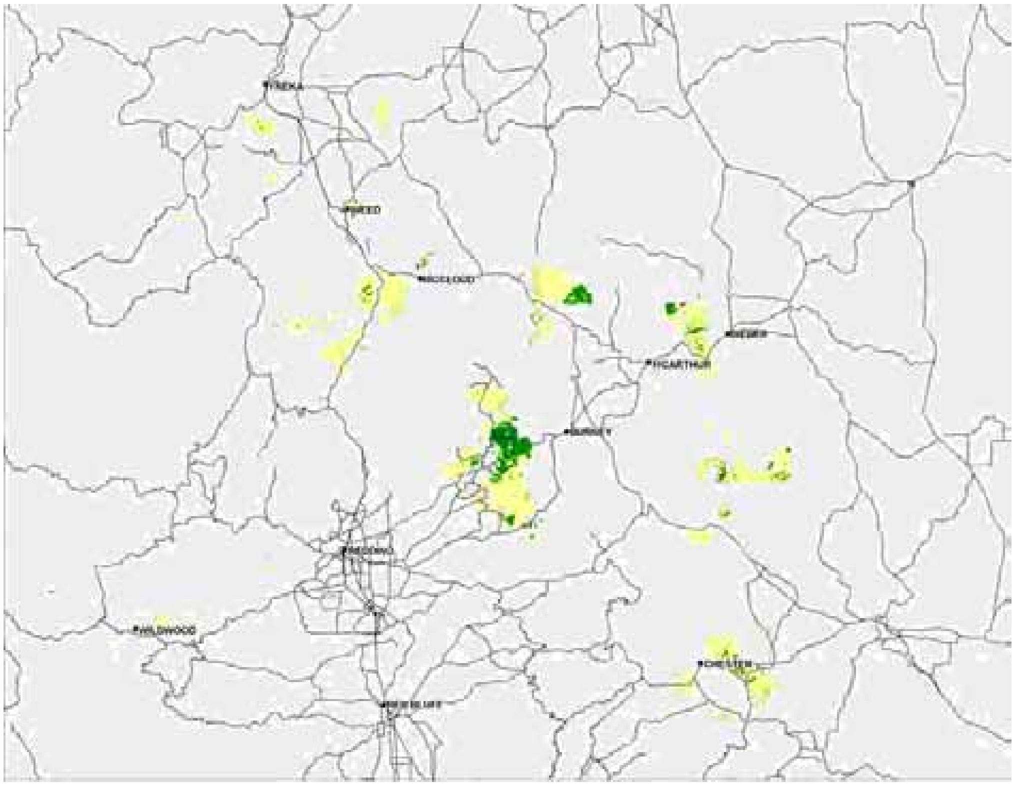

Figure 1—Vicinity map of Roseburg Resources ownership and plantations in Northern California.

Planning

The key to success of any reforestation project is proper planning. The temptation on large projects is to get started immediately with activities on the ground. However, the first step should be to break the project up into ecological units based on soils, aspect, elevation, vegetation type, access, slope, operability, and other attributes. Defining units of workable size can help focus efforts on achievable management objectives rather than on the overwhelming size of the undertaking. Knowledge of Geographic Information Systems (GIS) and spreadsheet software is invaluable for managing detailed information. Successful reforestation must be done in a series of steps and requires paying much attention to detail. Failing on any one of these steps often spells failure for the whole effort.

Units were prioritized on the basis of many factors relevant to maximizing success. The primary factor was the need for vegetation management. Competing vegetation from resprouting woody plants (mostly chinquapin) requires 2-3 years to grow sufficient crown volume to absorb enough chemical to completely kill the root system. Units with this type of competing vegetation were therefore deferred until the end of the reforestation effort. Areas where we could not use herbicides (along any perennial surface water) were planted immediately to take advantage of the site preparation from the fire. Beyond that the priorities were set by operational issues, such as timing of logging and a range of elevations to ensure as long a planting season as possible.

Planning does not take years; it can be done in a short period of time. Plans were completed in a matter of weeks on the Fountain Fire (FF). Logs were rolling into the mill by September 9, less than three weeks after the fire started and before the fire was even declared under control. It is important to take advantage of the site preparation created by the fire rather than see what competing vegetation re-occupies the site. Where woody vegetation was present before the fire, it will vigorously re-sprout after the fire. Use of herbicides within stream buffers was limited, so it was particularly important to take advantage of the site preparation achieved by the fire and to reforest without delay. Look for speculation stock in the nurseries that would be available to plant the following year.

Even though logging started immediately after the fire, rules set forth in the California Forest Practices Act (CFPA) provide a set of guidelines that ensure that environmental issues are not ignored. Stream buffers were established, archaeological surveys were completed, erosion mitigation measures were taken, and approximately 100 miles (160 km) of roads were improved with rock and drainage facilities. Although not required by the CFRA, Roseburg prepared Timber Harvest Plans (THP's) to ensure protection of the environment.

Seed Procurement and Inventory

Large landowners and governmental agencies normally maintain a seed inventory for small emergencies and unanticipated management needs. To meet the demand created by large fires, normal inventories are often inadequate. In this case, foresters must look on the open market for available seed from the correct seed zones and elevations. State agencies usually have seed available, as do private cone processors. A general rule of thumb is to move seed upon no further than 500 ft (150 m) up in elevation and 1000 ft (300 m) down, relative to the elevation of origin (J. Kitzmiller, retired from USDA-FS, pers. comm.). Likewise, seed should not be moved any further than the adjacent seed zone within the same series (climate). A standard inventory should be enough to reforest 10 percent of your land base, but probably more in areas of high fire risk. The more infrequent the cone crops, the larger should be the maintained reserve.

Superior seed programs can help reduce the amount of seed maintained in inventory. Seed orchards usually produce a more predictable and consistent source of seed. In addition to the reliability of seed production, improved seed generally has been developed to increase growth rates. On average, the first generation of improved seed is expected to increase growth by 10 to 15 percent, with an additional 10 percent from the second generation (J. Kitzmiller, retired from USDA-FS, pers. comm.). Improved seed is available for small landowners in California from the California Department of Forestry and Fire Protection. This state agency is a member of the North Sierra Tree Improvement Association (NSTIA), the most successful tree improvement cooperative on the west coast for ponderosa pine. The seed zones covered by the cooperative include 521, 522, 523, 524 and 525, on the west slope of the Sierra Nevada ranging from the Pit River in the north to the American River in the south.

A sufficient seed inventory also must cover the full suite of species required for potential reforestation projects. Ponderosa pine seed stores for a long time (over 20 years) as does Douglas-fir, but white fir (and most other *Abies* species) does not store as long (10 years or less).

Seedlings and Nurseries

After large fire events, it is important to reserve space in preferred nurseries to assure an adequate supply of quality seedlings. Criteria for selection of nurseries include the type of seedling desired (bareroot vs. container stock), species, storage, growing climate, lifting season, and others.

An effective and clear line of communication with the nursery is vital. Each nursery has its specialty, and not every nursery is good at growing all species. A solid grasp of the physiology of each species is helpful for asking important questions and making sure the nursery is growing the correct tree for your needs. Knowing the growth cycle of each species is very important. Quiescence is an important physiological state to understand, and deals with the transition period between growth and dormancy. The nursery must apply specific cultural practices to ensure that the appropriate size and physiological state at the right time. Cultural activities will change depending on whether you plant in the fall or spring. It is important to visit the nurseries at least once each year, and to communicate regularly by phone to emphasize your need for a very specific type of seedling.

Many important conditions can be observed on your visits. Did you get the germination that you planned for? Is the facility organized? How is the irrigation working? Is water distributed evenly? How is the fertilizer being calibrated? Do the trees look as good as the nursery manager says they are? Are you really going to get the trees you ordered? If they look poor, then you may want to consider a visit during lifting and packing to make sure they are culling weak trees properly. Are your trees clearly identified? If not, how can you be sure that you are getting your trees? Showing up and calling shows that you care and will be monitoring the nursery's performance closely.

Another factor to consider when planning reforestation is the number of nurseries you order seedlings from. Even the best nurseries can have unexpected problems, so it is a good rule to have several nurseries growing your seedlings for any given project. That way, if one nursery has a problem then your whole program will not be affected.

Seedling Trends

The trend in interior California is toward a higher proportion of container stock. Some interesting facts emerge from studies in Douglas-fir and ponderosa pine[3]. Three years after planting, initial stock size had no effect on the seedling volume of ponderosa pine or Douglas-fir. However, stock size did affect survival, with larger sizes experiencing less mortality. Larger containers result in lower bed densities and, hence, yield shorter trees with large caliper and lower limbs. The lower limbs shade the stem and prevent sun scalding. The same results were obtained for white fir. The

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[3] Fredrickson, unpublished data from Roseburg Resources field trials

best container sizes for ponderosa pine are stryro 5s or 8s; however; styro 8s and 10s worked best for Douglas-fir. If ungulate browsing is a potential problem, larger trees perform better (stryro 15s and 20s or plug-1 transplants). For surviving the summer drought, a relatively high root to shoot ratio (preferably 1:1, max.2:1) is essential, in contrast to the coastal region where taller stock is preferred.

The current trend in reforestation is to plant a mix of species. Conventional wisdom suggests that monocultures may invite insect problems, but even if they do occur the mix of species lowers the risk of total plantation loss. A big challenge in mixed species silviculture has been successful establishment of white fir with Douglas-fir. Planting the appropriate stock type is critical, and site preparation must be done to provide the specific types of microsites that Douglas-fir and white fir require. In the early stages of development these species are quite susceptible to sun scald and insufficient moisture. Providing dead shade and retaining soil organic matter to enhance water holding capacity are very important. The more organic matter that is retained during site preparation, the better is survival and growth. Organic matter retention includes minimizing the movement of organic matter across the unit. Maximizing organic matter retention, however, must be balanced against minimizing fire hazard.

Frost can be limiting to the success of high-elevation interior plantations. Fir planted on flats and low areas are susceptible to frost damage. The fir species are best planted on slopes (especially north aspect) with good air drainage.

Harvest Planning

Combining site preparation with logging activities can save a lot of money. In recent years our most effective tool has been a pre-harvest application of herbicide (mostly Chopper and Round Up, either separately or as a tank mix), preferably a year ahead of logging for maximum efficacy and greatest tolerance by conifer seedlings. We observed a 35 percent reduction in the need for release treatments as a result of pre-harvest applications[3]. Pre-harvest site-preparation provides more effective control of tenacious species like tanoak, snowbrush and other evergreen broadleaved shrubs and trees. Pre-harvest control is more effective in part because crowns and roots of undesirable species are not disturbed prior to logging and, allowing for better translocation and maximum efficacy of the herbicide. Pre-harvest site preparation also offers other benefits: lower labor costs, increased soil moisture retention, greater tolerance of planted conifers to treatment, and more options on choice of chemicals.

By spraying competing vegetation prior to logging, the dead material often disintegrates during felling and yarding. Chipping and removing sub-merchantable material before or after logging can also eliminate brush. This chipping operation adds another opportunity to run over dead woody material and break it down. Another benefit of chipping is the removal of unmerchantable material and reduction in the need to pile, especially if the trees are whole-tree yarded to the landing, where tops can be chipped. Even though chipping can be costly, if the market for biomass chips is weak, the chipping often is cheaper than piling, and it leaves more forest floor organic matter distributed across the site. Sub-soiling may be helpful depending on soil characteristics.

The importance of proper planning is particularly evident when considering access. To mitigate possible adverse effects of roads water quality, fish, and wildlife, the temptation is to bloc roads off. Closed roads create significant logistical problems for subsequent silvicultural treatments (planting, spraying, thinning, burning, piling, etc.). Early coordination of the reforestation effort with the harvesting schedule can reduce long-term costs of silvicultural activities considerably, but this coordination of logging and reforestation activities is too often an after thought.

Mechanical Site Preparation

Many more tools are available now for mechanical site preparation than was the case just a few years ago. The brush rake is still a favorite and is used frequently in California on slopes up to 30 percent. Ponderosa pine responds well to this type of site preparation. With the trend toward more mixed species plantations and reducing cost, sub-soiling and planting or just planting through the slash is increasingly common. These practices increase the risk of severe wildfire from the higher fuel loadings, and increase the probability of animal damage due to the better habitat that down wood provides. In addition, the cost of planting, spraying and thinning increases.

Sub-soiling is often done in combination with tractor piling of logging slash. Due to previous logging and site preparation, compaction is often a problem, especially with clay soils (*fig. 2*). After clearcutting or stand-replacing fire, the opportunity exists to mitigate soil conditions. Subsoiling can also be used to mitigate soils conditions, including hydrophobic soil resulting from extremely hot wildfires. The lower soil strength after subsoiling usually results in cheaper planting costs and higher quality planting, since the planters do not tire as quickly. **Error!**

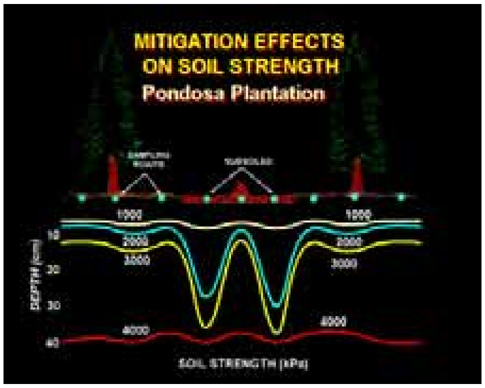

Figure 2—Soil compaction before and after mitigation in the Pondosa Burn.

The design of the shank on subsoiling equipment is important. The wings should be essentially flat, because the idea is to lift and fracture the soil, not to plow it. The shank should have a long leading tip, a sloped and tapered leading edge, and wings that feather behind the shank. Accumulation of roots and wood on the shank needs to be avoided because this accumulation acts as a plow. The operator needs to pay attention and lift the shank periodically. The target depth to sub-soil is a minimum of 18 in (46 cm) (R. Powers, UDSA-FS Pacific Southwest Research Station, pers. comm.). Due to the irregularities of the ground a target of 24 in (61 cm) is desirable, providing a range of treatment from 18 to 24 in (46 to 61 cm). Going deeper only slows the operation down and increases the cost.

The use of excavators for site preparation is becoming more common in California. Excavators are the preferred tool west of the Cascades in Oregon and Washington because they provide cleaner piles and increase organic matter retention (including large woody debris) across the site. However, the greater retention of organic matter again increases the risk of fire, and the more continuous fuel load increases the risk of escape from burning piles. An excavator with a 4-ft rake will increase the quality of planting and reduce the cost relative to an excavator with bucket and thumb.

The VH Mulcher mounts on an excavator or Timpco frame and works well for site preparation in slash or for making mounds. This tool was designed in Canada for shallow soils and high water tables. On such sites planting mounds are the preferred practices, contrary to accepted practices in dry Mediterranean climates. This tool works particularly well for making planting spots in slash, eliminating the need for piling. The VH Mulcher spins at a relatively high speed, so if the operator is not careful the mulcher can throw topsoil out of the hole, leaving only the C horizon to plant in. Depending on site conditions and management goals, several alternative microsites can be planted in the hole. The thumb on the head is used to move slash, and the bottom rotates to essentially drill a hole. By working the head back and forth you can prevent the removal of soil from the hole to some extent.

The Spyder is another tool that is relatively new, but is very effective for piling on steep slopes (*fig. 3*). This machine essentially walks on four legs, so is capable of working on slopes up to 100 percent. The Spyder has been used as an alternative to broadcast burning. In California, the maximum clearcut size allowed by the CFPA is 40 ac (16 ha), but the operational average is 15 to 20 ac (6 to 8 ha). Broadcast burning extremely expensive under these conditions. Increasing restrictions on burning due smoke management and air quality also limit opportunities for broadcast burning. Pile burning has a wider range of permissible conditions (larger burning window) than broadcast burning. A larger window is available for burning piles because they can be covered and burned later during wet conditions when risk of escape is low. The large amount of organic material left behind makes this escape risk a serious consideration.

Figure 3—Spyder is used to prepare sites for planting.

As mentioned earlier, chipping is a valuable tool that can be used to reduce the need for piling, but opportunities for chipping depend heavily on local demand for chips, the availability of an infrastructure to process and transport the chips, and a price that makes removable of this material economically feasible. The key factor determining economic viability is the number of dry tons removed per acre. For example, if 10 tons per ac (22 Mg per ha) must be removed, if the market will pay $40 per ton ($44 per Mg), and if hauling costs $45 per ton ($50 per Mg), then fuel reduction by chipping and removal will cost $50/acre ($132 per ha).

Masticators are another group of machines that mulch brush very well. One disadvantage is that they do nothing to disturb the root system, and another is that the machines are expensive. Without some follow up chemical treatment, sprouting shrubs and hardwoods will return within three to five years.

To summarize, the predominant trend in mechanical site preparation is to leave more organic material on site. This increased organic retention is a benefit to Douglas-fir and white fir survival, but it increases fire risk and drives up the cost of subsequent silvicultural treatments.

Broadcast Burning

The cost of broadcast burning is becoming prohibitive, except as a last resort, due to the increasingly small clearcut units in California. Burning is inherently risky and requires extensive training and experience to implement successfully. Increasingly strict smoke management guidelines designed to meet air quality standards also make it difficult to obtain permits and increases the costs.

Vegetation Management

Vegetation management is essential to the establishment of fully stocked plantations in northern California and many other regions. Without vegetation control, many plantations are doomed to failure (*fig. 4*). Chemical treatment is the safest and most effective means of ensuring reforestation success. Research in vegetation management has led to safer chemicals and lower application rates with the intent to minimize environmental impacts.

The objectives of vegetation management are: 1) to maximize site resources available to planted trees and thereby ensuring survival and enhancing growth, 2) to achieve the highest level of control with the least amount of chemical, 3) to obtain desired results while protecting and enhancing other resources and 4) to continually improve results through research and development.

Our vegetation management strategy has relied on well-planned initial site preparation. The current strategy involves pre-harvest chemical treatment, followed by logging, mechanical site preparation, and a residual herbicide to control the grass and forbs. The goal is to maintain a weed-free environment for the first two years to ensure successful establishment. Release treatments are scheduled as necessary to increase growth (*fig. 5*). Twenty five percent aboveground brush cover equates to root systems covering 100 percent of the area.

Figure 4—A demonstration area in the Pondosa Burn of northeastern California showing an established ponderosa pine plantation where competing vegetation was controlled and a brushfield resulting from no treatment.

One important key to success is application of the correct product on the target species, even if that means treating the same unit twice with two different chemicals. This approach will be cheaper in the long run because it is more difficult to kill the unaffected species after one or more growing seasons. Applications of the proper chemical will kill the undesirable species rather than simply brown them.. Growth regulators should not be mixed with a systemic chemical, e.g., Garlon and Roundup. The growth regulator destroys the cell structure in the plant before the systemic chemical gets translocated to the roots. Chemical application beyond the rates and volumes needed to kill the target species is a waste. Perhaps the most important factor is getting a good foreman and crew who will pay attention to the detail required to be successful.

Pre-Harvest Site Preparation

The goals of most pre-harvest herbicide applications include: 1) achieving long-term control without re-sprouting, 2) avoiding conifer damage, 3) minimizing application costs and 4) eliminating future entries.

So how do pre-harvest applications achieve these goals? Logging affects how the vegetation reacts to chemicals. Often after disturbance the plants go into shock and will not absorb chemical well. During logging the root to shoot ratio is increased by mechanical damage to the shoot; therefore, the remaining crown may not be able to absorb enough chemical to kill the root system. By treating before harvest, the root to shoot ratio is sufficiently low to ensure effective chemical control. In addition, plants are usually less able to resist chemical control when light, moisture and nutrients in the understory limit their sprouting vigor. Pre-harvest herbicide treatment is especially effective for schlerophyllous hardwood species that are hard to kill. Chopper (imazapyr) and high concentrations (5 to 10 percent) of glyphosate (especially with deciduous hardwoods) work well. Pre-harvest application also leads to greater success of fall planting due to increased soil moisture (*fig. 6*).

Application costs are lower before harvest, assuming that good access is available. The added benefit is reduced release costs after planting[4]. Lower rates of

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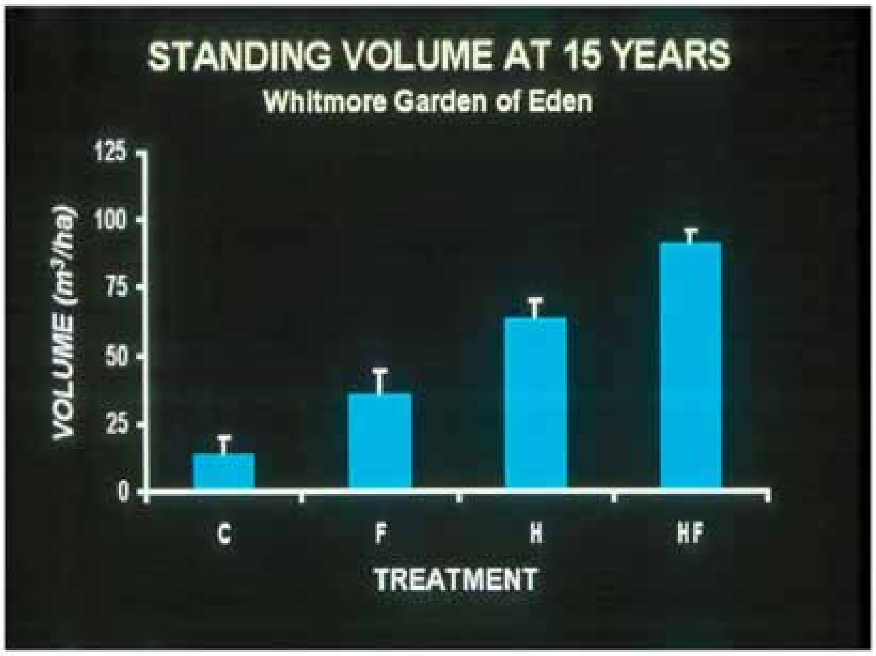
[4] E. Fredrickson, unpublished data from Roseburg Resources field trials.

residual chemicals are required for herbaceous species than for both herbaceous and woody species. Also, options for chemicals to apply are much greater before harvest than after planting because conifer tolerance of the available chemicals varies.



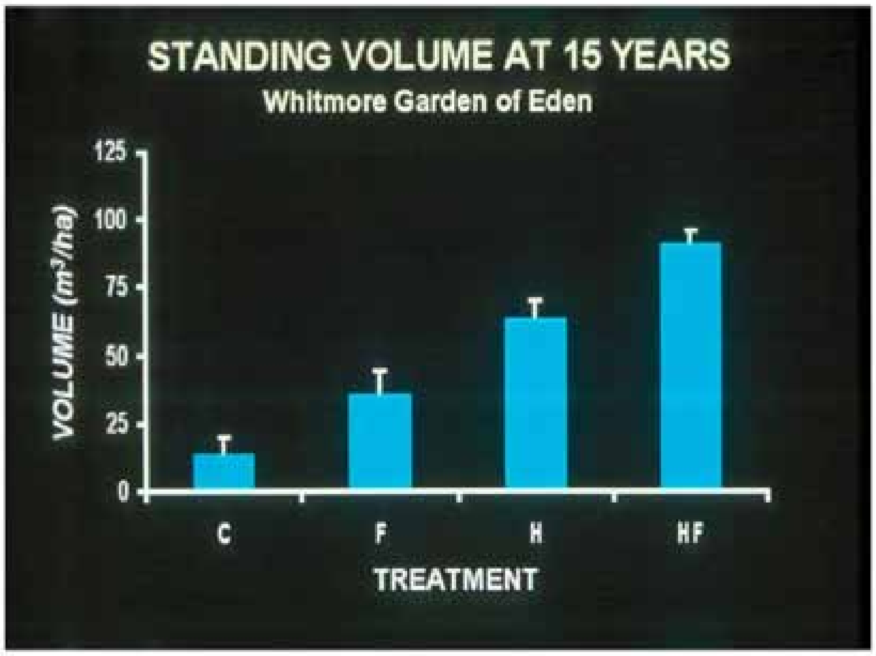


Figure 5—demonstrating the value of vegetation control, F=fertilization, H=herbicide, C=control. (Robert Powers, USDA Forest Service, Pacific Southwest Research Station, Albany, CA, unpublished results).

Keys pieces of advice are: 1) treat at least one year in advance of logging, 2) use esterified seed oils or seed oil silicone surfactant combinations to get good penetration (usually 5 to 10 percent depending on time of year, with more late in year), and 3) providing access is critical - *the most common challenge to pre-harvest site preparation is adequate planning with the logging department to ensure access in advance.*

Figure 6—Difference in available moisture three months after planting in units receiving competing vegetation control (left) vs. units receiving no control (right).

Fall Planting

Fall planting is difficult to manage historically gives unpredictable results. So why do we continue to pursue it? We found greatest success with fall planting above 5000 ft (1520 m), where snow limits access until after summer drought begins. Planting at high elevations can usually be done only at such a late date that little or no precipitation typically occurs after planting, and extreme temperatures can still occur before the trees develop root systems sufficient for survival.

So what does it take to be successful? Planting only after receiving early fall moisture (preferably 1 to 2 in (2.5 to 5 cm) in September) is critical, although we have seen that, in areas with pre-harvest treatment, a half-inch (1.2 cm) has been enough for successful reforestation. Choosing the proper sites and planting stock are important. Container stock works best due to its flexibility with respect to lifting and shipping on short notice. If weather prevents planting, then container stock stores well. Having stock that is hardened off enough to deal with the elements is important to deal with any hot weather in the fall. However, the trees should not be so hardened off that they are in dormancy and still capable of growing roots. The ideal is to plant at least three weeks before soil temperatures drop below 40˚F (4˚C), at which time root growth largely stops (above 5,000 ft (1520m) this usually occurs by November). When fall planting; a “hot plant” is essential; that is, the trees should be planted within 10 to 14 days of lifting so that they are ready to grow right out of the cooler. When storing trees, do not store below 40 degrees, otherwise the trees will be pushed into dormancy and will not grow roots after planting. If seedlings are kept longer than 10 days at 40˚F (4˚C) or more, there is an increased risk of fungal attack (*Botrytis* spp.).

Biodiversity

The use of herbicides often raises concerns about adverse impacts on biodiversity. DiTomaso et al. (1995) conducted a biodiversity study in three different fires, the Pondosa burn of 1977 (treated and planted between 1981 and 1988), the Tamarack fire of 1985, and the Fountain fire (FF) of 1992. The primary site preparation chemical was Velpar (active ingredient is hexanzinone). In the Pondosa burn and Tamarack fire, stands treated with herbicide showed greater species diversity than stands with no treatment; however, the trend over this chronosequence of three sites shows that with time the diversity will approach pre-burn species diversity in both treated and untreated stands (*fig. 7*). Untreated stands are rapidly dominated by a few species of brush, but treated units have greater diversity of grasses and forbs. The treatment effect is relatively short lived, so is consistent with the goal of setting back the competing vegetation only long enough for successful establishment and growth of seedlings.

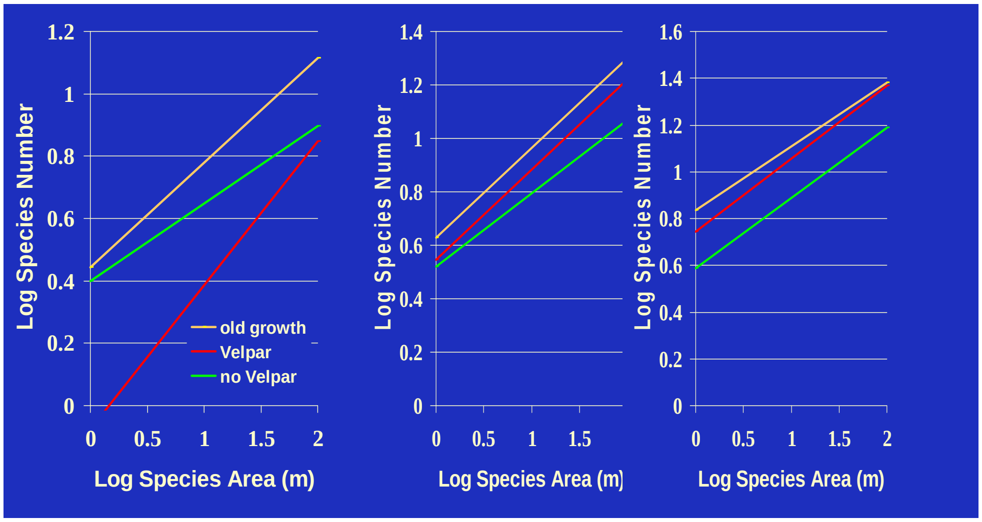


Figure 7—Plant species richness for three stand conditions (old growth, young plantation with Velpar treatment, young plantation with no Velpar treatment) and three stand ages (Fountain Fire (1992), Tamarack Fire (1985), and the Pondosa Burn (1977).

Pre-Commercial Thinning

Most foresters generally leave too many trees after thinning (Webster, 1997), often necessitating a second precommercial thinning to avoid stand stagnation and excessive mortality. Evaluating stand density using Reineke's (1933) Stand Density Index (SDI=equivalent number of 10-in (25-cm) trees per ac), Oliver and Uzoh (1997) have shown that ponderosa pine stands rarely exceed an SDI of 365 (902 25-cm trees per ha). However, heightened risk of bark beetle mortality begins at an SDI of about 230 (568 per ha). Disturbance from snow breakage is confined almost exclusively to stand densities of more than 183 SDI (452 per ha). Ensuring that trees in the stand get to a commercial size (minimum mean diameter of 14 in (36 cm)) as they reach these SDIs requires pre-commercial thinning to approximately 140 trees per ac (346 tree per ha) or less (Table 1). The stand density bounds for a managed depend on the level of risk one is willing to accept under stressful conditions such as drought.

Table 1*. Stand density index (SDI) given by four different combinations of tree density (trees/ac) and quadratic mean diameter (QMD, in). SDIs are based on a slope of –1.77 for the maximum size density limit (Oliver and Powers 1978).*

140 **(346)**	14 (35.6)	254 (627)
140 **(346)**	15 (38.1)	287 (709
140 **(346)**	16 (40.6)	322 (795)
140 **(346)**	17 (43.2)	358 (884)

Pre-commercial thinning presents the problem of slash disposal to prevent excessive fuel loading and fire risk. Within the public agencies, reducing fuel risk is a high priority, so lopping or some type of fuel treatment is required after thinning. In forest industry, minimizing cost is a higher priority, so lopping is done only along roads to meet regulatory requirements. This type of fuel creates an elevated fuel hazard and after ignition supports longer flame lengths, making fire risk and potential severity higher for 5 to 10 years after the thinning.

Where co-generation plants exist, cutting and hauling the material to the plant can reduce other types of fuel. Marketing the biomass of small-diameter material in this way can offset the costs of thinning and reducing fuel loads.

Western Pine Shoot Borer

The western pine shoot borer (*Eucosma sonomana* Kearfott) is endemic to the interior West and attacks young ponderosa pine. The female attraction pheromone was identified in the 1970s. Many studies have explored the use of pheromone for tree protection. One application technology referred to as microencapsulated, or MEC, was tested aerially in 2002 and 2004 and worked very well (Gillette and others 2004). This product was developed by 3M and involves microscopic capsules mixed in water. Unfortunately, the company has decided to discontinue manufacturing the product. Applied Pheromone Technologies in Vancouver, Washington, has just registered a hand application product in California (Last Call AK). This product is a mating disruptant and an insecticide that together "attract and kill" (AK) the target species. Another aerial product consisting of tiny flakes and produced by Hercon is being tested in 2005.

Williams (1989) showed about a five percent increase in volume growth where the male pheromone was introduced as a disruptant. In that study a single application to a young stand yielded better results than multiple treatments to older stands. Recent observations (Webster and Gray, unpublished data) show a doubling of damage to the terminal shoot after pre-commercial thinning. Endemic populations of the western pine shoot borer within the Pondosa Burn of eastern Siskiyou County damaged 20 to 40 percent of the trees in 1996. After thinning, damage increased to 80 to 90 percent of the trees, and it took about five years for damage to return to endemic levels. If only a single pheromone treatment was feasible, these observations suggest that a treatment immediately after pre-commercial thinning may be the best strategy.

Conclusions

Three main conclusions are worth re-emphasizing:

- Planning is critical!!
- Attention to detail is essential!!
- Research is invaluable!!
    - Increases success
    - Reduces cost in the long term
    - Increases the potential for growth
    - Develops treatments friendlier to the environment

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