CHANGES IN FOREST SOILS AS THE RESULT OF EXOTIC DISEASES, TIMBER HARVEST, AND FIRE EXCLUSION AND THEIR IMPLICATIONS ON FOREST RESTORATION

Russell T. Graham and Theresa B. Jain

Introduction

In the western United States and throughout the world, three general classes of coniferous forests can be identified with each having similar vegetative complexes, native disturbances, and climate (Daubenmire and Daubenmire 1968, Hann et al. 1997). Dry forests, often dominated by pines (*Pinus*), cold forests often dominated by spruces (*Picea*), and moist forests dominated by hemlocks (*Tsuga*) represent the majority of the world's forests. The soils within each of these forest classifications are a product of their inherent climate and vegetation dynamics. Because soils are the foundation of the forest ecosystem, they are integral to all forest restoration efforts. However, because people value and cherish trees, they are often the focus of most restoration activities and the changes occurring in forests are most noticeable in the tree component. Presently in the western United States the most evident changes in many forests are the large number of trees occupying them compared to the amounts of trees that occurred in historical (pre-1900) forests. In several forests not only has the number of trees increased but their composition is far different than what occurred historically. Normally, in the absence of disturbance, early-seral (shade intolerant) vegetation is succeeded by mid- and late-seral (shade tolerant) vegetation. For example, in the western United States, in the absence of disturbance, quaking aspen (*Populus tremuloides*), an early-seral species, can be succeeded by late-seral ponderosa pine (*Pinus ponderosa*) in some locales in less than 100 years and it may take western redcedar (*Thuja plicata*) hundreds of years to succeed mid-seral western white pine (*Pinus monticola*) (Daubenmire and Daubenmire 1968). Five key disturbances, or the lack thereof, influencing forest successional patterns and rates are human uses and resource extractions, fire, insects, diseases, and weather (Graham and Jain 2006).

In the western United States, harvesting of early-seral species started in earnest early in the 20th Century and by the 1960s, harvesting peaked and by the 1980s it began to diminish (Lewis 2005). This harvesting greatly decreased the amount of old forest structures occurring in the landscape and, along with fire exclusion, facilitated many forests to be dominated by late-seral vegetation. Fire suppression in the United States and in other parts of the world started during the 1800s but only became effective in the 1930s with the development of airplanes, road systems, fire danger ratings, and effective fire fighting techniques (Steen 1976). Insects and diseases, along with weather (e.g., snow, wind, ice), historically influenced forest development, however in concert with harvesting and fire exclusion, their impacts along with climate change appear to be greater than what they were historically. As a result, restoring forests to a desired state or developmental pathway requires the understanding of how all of these disturbances interact with the vegetation within a given biophysical setting. Restoration objectives are relevant to forests in all three classifications but, because the amount of departure or change exhibited in the dry and moist forests is the greatest, we will limit our discussion to these two forest classifications while examining the implications soil changes have on restoration efforts (Covington and Moore 1994, Hann et al. 1997). In addition, we will use the dry and moist forests of the western United States to illustrate how soil, as well as vegetation, should be included in restoration planning and activities. Nevertheless, the concepts we point out will most likely have application in similar forests throughout the world where restoration activities are planned or are being considered.

Moist Forest Changes

In the western United States fire exclusion, insects, diseases, and weather interacted with climate change (climate cycles) to modify the moist forests from those dominated by early- and mid-seral species (e.g., pines) to forests dominated by mid- to late-seral species (e.g., cedar) (Hann et al. 1997). Historically, wildfires burning the moist forests were highly variable. It has been estimated that non-lethal (surface) fires occurred at relatively frequent intervals (15 to 25 yrs) within 25% of the moist forests. Lethal crown fires burned about 25% of the moist forests every 20 to 150 years but occasionally extended to 300 year intervals. The mixed fire regime (combination of lethal and non-lethal) burned about 50% of the moist forests at 20 to 150 year intervals with some lethal events occurring at 300 year intervals. Fires typically started burning in July and were usually out by early September with the change in weather. Most fires were small but occasionally large fires did occur with 74% of fires killing a portion of the canopy. Fire exclusion has impacted the moist forests and

USDA Forest Service, Rocky Mountain Research Station, Moscow, ID, USA; rtgraham@fs.fed.us

it is estimated that surface fires now burn 10% of the moist forests, mixed fires burn 30% and 60% of the moist forests are currently burned by crown fires (Hann et al. 1997).

Although fire exclusion played a role in altering the moist forests of the northern Rocky Mountains, introduction in 1909 of the European white pine blister rust (*Cronartium ribicola*), caused the greatest change (Fins et al. 2001). The disease attacked and killed western white pine, the primary commercial species of the moist forests. Upon introduction of blister rust, a massive effort was initiated to control the disease by removing currant (*Ribes* spp.) bushes (the alternate host). Beginning early in the 1900s and, in earnest during the 1930s, *Ribes* bushes were pulled, sprayed, and grubbed throughout the northern Rocky Mountains. The effort has been described "like bailing the ocean with a teacup" and efforts to control the disease proved futile. By 1968 the western white pine blister rust program and management of the species was "realigned," resulting in the accelerated removal of naturally occurring western white pine before they succumbed to the rust (Hutchison 1942, Ketcham et al. 1968).

The forest changes resulting from the introduction of blister rust were exacerbated by harvesting of ponderosa pine and western larch (*Larix occidentalis*) two additionally high value timber species of the moist forests (Hutchison 1942). The partial and intermittent canopy removal and minimal soil surface disturbance caused by both tree harvest and western white pine mortality were ideal situations for grand fir (*Abies grandis*) and western hemlock (*Tsuga heterophylla*) establishment, which aggressively encroached along with Douglas-fir (*Pseudotsuga menziesii*) and readily filled the niches western white pine, ponderosa pine, and western larch once held (Hann et al. 1997, Jain et al. 2004).

Recent (1991) native insect and pathogen activity in the moist forests far exceed those of the past (Hessburg et al. 1994). The Douglas-fir beetle (*Dendroctonus pseudotsugae*), mountain pine beetle (*Dendroctonus ponderosae*), spruce budworm (*Choristoneura occidentalis*), and tussock moth (*Orgyia pseudotsugata*) historically endemic, are now often epidemic. Similarly, the rot diseases *Armillaria spp*. and *Phellinus weirii* that were historically endemic are now common in the current fir-dominated forests. Weather, another formidable disturbance, in the form of snow or wind, often creates a variety of canopy openings ranging from gaps to large openings (≈16 ha). In early-seral species (ponderosa pine, western larch, western white pine) dominated forests, snow will often slip from the trees, minimizing breakage, while other species in the intermediate crown classes (grand fir, Douglas-fir) will break, creating gaps and openings, decreasing forest densities, and altering species composition. Today, species such as grand fir and Douglas-fir that dominate sites have dense crowns that hold snow often leading to stem breakage, facilitating insect (i.e. Douglas-fir beetle) epidemics and regeneration opportunities for the true firs and hemlocks (Jain et al. 2004, Graham and Jain 2006).

Western white pine, ponderosa pine, and western larch dominated forests are generally tall and self-pruning even in moderately dense stands. They have large branches high in the crowns and the base of the crowns is well above surface fuels. In general, this crown architecture protects the nutrients stored in the canopy from surface fires. In contrast, young- to mid-aged (<150 years) western hemlock and grand fir generally do not self-prune. Forests dominated by grand fir tend to concentrate both nitrogen and potassium in their foliage which often extends to the soil surface. Generally, this 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 mechanical and fire destruction. This canopy architecture favors lower crown base heights, higher crown densities, and canopies with higher nutrient (especially potassium) content than occur in western larch, ponderosa pine and/or western white pine dominated forests (Harvey et al. 1999, Mika and Moore 1990).

Dry Forest Changes

In the western United States, domestic livestock grazing and harvesting of ponderosa pine forests was occurring by the mid 1800s. In mesic forests, grand fir and/or white fir (*Abies concolor*) and Douglas-fir rapidly colonized these sites when ponderosa pine was harvested. Especially where ponderosa pine was the late-seral species, grass cover tended to decrease ponderosa pine seedling establishment and survival. However, when heavy livestock grazing ceased (and the damage livestock did to seedlings) 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, Graham and Jain 2006).

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 (Barrett and Arno 1982, Stewart 1951). Cultural burning by Native Americans augmented and even dominated burning in several locations. In the northern Rocky Mountains of Idaho and western Montana, dry settings (late-seral ponderosa pine and/or Douglas fir forests) were historically burned by non-lethal wildfires at 15 to 23 year mean return intervals and these fires could be quite large often burning for weeks to months. Mesic forests containing ponderosa pine (where grand fir and/or Douglas-fir were the late-seral species) were burned frequently by mixed fires at mean return intervals extending to over 60 years. Non-lethal fires dominated the central and southern Rockies (where ponderosa pine and/ or Douglas-fir were late-seral species), although mixed severity fires also occurred, especially along the Front Range of the Rocky Mountains in Colorado. On the driest settings (late-seral ponderosa pine and/or woodlands), because of discontinuous surface fuels, fires tended to be few. In contrast to other locales dominated by late-seral ponderosa pine, the forests of the Black Hills of Wyoming and South Dakota possibly experienced greater extents of lethal fires because of the abundant ponderosa pine regeneration that normally occurred. Nevertheless, historical wildfires likely 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 (Shepperd and Battaglia 2002, Shinnen and Baker 1997, Bradley et al. 1992).

Within the dry forests, dense conditions often developed, exacerbated by fire exclusion that increased the abundance of insect and disease epidemics, significantly altering the composition and structure of these forests. Historically, western pine beetle (*Dendroctonus brevicomis*), pine engraver (*Ips spp*.), fir engraver (*Scolytus ventralis*), and tussock moth (*Orgyia pseudotsugata*) were insects associated with regularly burned areas. In most years bark beetles occurred at endemic levels in ponderosa pine, Douglas-fir, and grand fir killing large weakened trees that were struck by lightning, infected by root disease (*Armellaria spp*.), or too old to resist attack. Pine engraver and fir engraver beetles attacked young, densely stocked ponderosa pine, killed trees scorched by low-intensity surface fires, and severely infected trees containing root rot or dwarf mistletoe. Because of forest changes, these same insects have reached epidemic levels in many forests and today ponderosa pine continues to be susceptible to the western pine beetle. Mountain pine beetle is affecting current dry forests where Douglas-fir and grand fir are the late-seral species. The pine engraver beetle is more abundant and destructive now with some of the severest outbreaks occurring on low-elevation sites where ponderosa pine is the late-seral species. Pandora moth (*Coloradia pondora*) defoliates ponderosa pine and scattered outbreaks have occurred in Arizona, California, Colorado, and Oregon during the 20th century. Dense stands of Douglas-fir and grand fir that developed where these species are late-seral are very susceptible to both defoliators and root diseases (Harvey et al. 2000, Hessburg et al. 1994).

Harvesting western larch, ponderosa pine, and the ingrowth of grand fir/white fir and Douglas-fir in the dry forests facilitated the accumulation of both above- and below-ground biomass and associated nutrients close to the soil surface. Even low-intensity surface fires often now consume the surface organic layers, killing fine roots, volatilizing nutrients, killing trees, and increasing soil erosion potential. In addition, fir ingrowth creates nutrient-rich ladder fuels that facilitate crown-fire initiation, increasing the likelihood of nutrient loss. The risk of nutrient loss is greater on infertile sites because dense stands of late-seral species are more demanding of nutrients and water than the historical stands dominated by widely-spaced early-seral species (Harvey et al. 1987, Minore 1979, Harvey et al. 1999).

With the advent of fire exclusion, animal grazing, timber harvest, and climate cycles in areas where grand and/or white fir are late-seral species, ponderosa pine is being succeeded by Douglas-fir, grand fir and/or white fir. Fire intolerant vegetation, dense forest canopies, and homogenous and continuous horizontal and vertical structures are developing thus creating forests favoring crown fires rather than low intensity surface fires that historically occurred in these forests. Within the Northwestern United States, the extent of mid-seral (e.g., Douglas-fir, grand fir) vegetation has increased by nearly 8 million acres and the extent of single storied mature vegetation (e.g., ponderosa pine) has decreased by over 4 million acres since the early 1900s (Hann et al. 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 grand fir in 300 to 400 years within forests historically burned by frequent fires but in many locations grand fir has succeeded ponderosa pine in less than 50 years. Grand fir in particular has the propensity to concentrate nutrients in their foliage and when these trees are burned by wildfires or prescribed fires, nutrients in these abnormal forest structures can be lost from the forest. Nitrogen can be replaced through fixation (i.e., symbiotic or non-symbiotic), deposition, and lightning relatively quickly (10s to 100s of years), however, nutrients such as potassium are replaced through the weathering of rocks, a very long-term (100s to 1000s of years) process (Gruell et al. 1982, Harvey et al. 1999, Smith and Arno 1999).

Forest Floor Changes

The shift in species composition from western white pine, western larch and/or ponderosa pine to Douglas-fir, grand fir/white fir, and/or western hemlock dominated forests (including the shrub and forb components) has changed litter (soil surface) type from that which occurred historically. In addition, the accumulation of both above- and below-ground biomass from roots, needles, and boles in fir forests is accelerating activities of decomposers by increasing and changing the basic substrate they utilize. Associated with these changes in litter type and quantity is a likely change in ectomycorrhizal relationships, soil surface chemistry, including allelopathic substances with the potential to alter a variety of microbial activities (Harvey et al. 1999, Harvey et al. 2000). Also, these forest floor alterations have been exacerbated in many locales singly, or in combination, by soil compaction and displacement, fire exclusion, timber harvesting, animal grazing, and climate shifts. For example, decomposed true firs create white rotten wood which rapidly disperses into the soil and is quickly consumed by decomposers. In contrast, decomposed ponderosa pine, western white pine, and western larch create brown rotten wood which can persist in soil for centuries and has been shown to retain nutrients and hold water. Western larch and ponderosa pine tend to be deep-rooted in contrast to the relatively shallow-rooted western hemlock and grand fir, which have abundant feeder roots and ectomycorrhizae in the shallow soil organic layers. The soil microbial activities in fir-dominated forests compared to pine-dominated forests may diminish the post-fire acquisition and cycling of nutrients. Moreover, these changes in soil microbial activities may increase the likelihood of uncoupling any continuity between current and preceding vegetative communities (Harvey et al. 1987, Harvey et al. 1999, Harvey et al. 2000, Graham and Jain 2006).

In the dry forests, biological decomposition is more limited than biological production. When fire return intervals reflected historical fire frequencies, the accumulation of thick organic layers was minimized and nutrient storage and nutrient turnover was dispersed into the mineral soils. In the absence of fire, bark slough, needles, twigs, and small branches can accumulate on the forest floor allowing ectomycorrhizae and fine roots of all species to concentrate in the surface mineral soil and thick organic layers. In addition, 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. 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 variations found within the dry forests. As a result, they are often subject to high insect and pathogen mortality and can not be considered either long-lived or stable (Harvey et al. 1999).

Restoration Strategies

Even though trees are the dominant forest structures associated with restoration strategies soils and changes in them that may have occurred because of the disruption or alteration of disturbances are integral to all restoration activities. In general, restoration strategies that affect soils can be direct or indirect. Very often the reintroduction of fire directly into forests in which the natural fire regime has been interrupted for decades is suggested as a restoration tool. As mentioned before, deep layers of organic materials often accumulate on the forest floor in the absence of fire, and if they are not managed appropriately, both soil and tree structures can be damaged when the forest floor is burned. We have applied prescribed fire around the base of large trees and mixed the surface of these layers when the lower forest floor (duff) layers are cool ($< 4.4^{o}$ C) and their moisture contents exceed 100%. We have successfully reduced the amount of organic materials around these trees without stressing or killing them. However, depending on the amount of material accumulated, it may take numerous treatments to reduce these uncharacteristic depths to levels reminiscent of those that occurred historically. When the forest floor conditions warrant it, appropriately planned fires can be broadcast throughout the forest reducing the risk of killing these often large and valuable trees. Raking away the deep organic layers from the base of trees may prevent heat from killing the cambial tissue during a fire but if fine roots occur in these layers, this physical destruction may stress or even kill the tree similar to destroying them with fire. These conditions do not develop in all dry forests in which fire has been excluded but it can easily be checked by grabbing a hand sample of organic material and, if roots are present, judicious application of fire is warranted (Graham and Jain 2006).

Particularly in the moist forests, a large proportion (≈40%) of the surface organic layers and incorporated into the mineral soil can be decayed wood. This material is often rich in nutrients and water, is the site of numerous fine roots and the center of ectomycorrhizal activity (Harvey et al. 1987). The source of these materials is coarse woody debris (CWD) and the manipulation of CWD indirectly affects these soil materials. CWD originates from tree boles and large limbs usually defined as materials exceeding 8.0 cm in diameter. Historically CWD did not readily accumulate in the dry forests burned by frequent fires but in forests having mixed and lethal fire regimes where these materials often accumulated between fires. When groups and patches of trees were killed by a fire, the fallen dead boles could be readily buried by eroding soil. These materials, depending on the forest setting, contributed to the organic component of the forest soil. After a major disturbance (e.g., wildfire, timber harvest), to ensure that this soil component is readily available to future forests, 6.5 to 37.0 Mg ha^{-1} of CWD are recommended depending on the forest locale (Graham et al. 1994). Over the years these materials can be buried and/or decompose and slowly become part of the soil matrix. If there is a choice between species of CWD to leave or manage for, those that form red-rotted materials (e.g., pines) are preferred to those forming white-rotted materials (e.g., true firs) because red-rotted materials are much longer lasting (100s vs. 10s of years) (Reinhardt et al. 1991, Harvey et al. 1999, Harvey et al. 2000, Graham and Jain 2006) .

The uncharacteristic plurality of mid- and late-seral species occupying many forests not only alters the rotten wood dynamics in forests but these species tend to concentrate nutrients in their foliage. The removal or burning of this mid- and/or late-seral tree encroachment can remove nutrients which are often in short supply. Forest treatments can be tailored to restore the desired forest structures yet conserve the nutrients. Mechanical mastication of these often fine fuels can be used to keep the nutrients on site but these treatments need to be applied to not exacerbate the fire hazard. Depending on the forest locale this can be accomplished by creating a variety of materials and creating a variety of forest floor conditions that encourage the decomposition of the materials.

Restoration activities can conserve organic materials and, where they are depauperate, activities encouraging their creation are warranted. Mechanical piling and mastication of forest residues can be used to manage CWD and fine materials but, as in all forest management activities, the conditions of the soils and materials after mechanical treatments depend on the machine used, the biophysical conditions of the forests, and more importantly the operator of the machine. The operator determines the conditions of the organic material after treatment and the amount of allied soil disturbance (e.g., soil compaction and displacement). There is a trend, especially near urban areas to chip small trees, limbs, and CWD as a method of decreasing the fire hazard. Thin layers of wood chips spread on the forest floor tend to dry and rewet readily and deep layers of chips and chip piles may have insufficient air circulation making both conditions poor for decomposition. When layers of small woody material are spread on the forest floor they can insulate the soils and when decomposition does occur, the decomposing organisms utilize large amounts of nitrogen reducing its availability to plants. Large chunks and a variety of piece sizes and their uneven dispersal create conditions favorable for rotting organisms no matter the fungal species, its favored substrate, or its moisture and heat requirements.

Fire is very appropriate for use in restoring forests and forest soils. However, unless fuel and weather conditions are appropriate, fire can create conditions adverse to vegetative development and impair soil productivity. The amount of forest floor consumed by a fire is dependent on its moisture content particularly in the lower humus and fermentation layers. Fire applied when the moisture content of these lower duff layers exceeds 100% most often conserves the organic materials. Burning under these conditions, the nutrients (P, N, K) in the litter and fine fuels (≤ 8.0 cm) have the opportunity to condense in these lower layers and, therefore, are not lost from the site. Fires that leave a variety of burn severities also create a multitude conditions for seed germination and vegetation development (Ryan 1982, Hungerford et al. 1991).

Integrated Soil and Organic Material Restoration Strategies

Even though the concept of integrating soil and organic material into restoration strategies might appear to be new, they have their genesis in the practice of silviculture. Silviculture is the art and science managing forests to meet management objectives and restoration is a very appropriate objective. In addition, soils and organic materials are integral forest components and are as appropriate as trees for focusing silvicultural methods and systems. A silvicultural system is a planned series of treatments through the life of a forest and using such an approach for planning soil restoration treatments is appropriate. In addition, such activities could be incorporated into holistic silvicultural systems that integrate all forest components relevant to contemporary forest management objectives including, but not limited to restoration, timber, wildlife habitat, or water production to name a few (Graham and Jain 2004).

Literature Cited

Barrett, S. W.; Arno, S. F. 1982. Indian fires as an ecological influence in the Northern Rockies. *Journal Forestry* 80(10): 647-651.

Bradley, A. F.; Fischer, W. C.; Noste, N. V. 1992. Fire ecology of the forest habitat types of eastern Idaho and western Wyoming. Forest Service, General Technical Report INT-290. 92 p.

Covington, W. W.; Moore, M. M. 1994. Post-settlement changes in natural fire regimes and forest structure: Ecological restoration of old-growth ponderosa pine forests. In: *Assessing forest ecosystem health in the Inland West*. New York: The Haworth Press Inc. 153-181.

Daubenmire, R.; Daubenmire, J. B. 1968. Forest vegetation of eastern Washington and northern Idaho. Technical Bulletin 60. Pullman, WA: Washington Agricultural Experiment Station. 104 p.

Fins, L. Byler J.; Ferguson, D.; et al. 2001. Return of the giants: Restoring white pine ecosystems by breeding and aggressive planting of blister rust-resistant white pines. Moscow, ID: University of Idaho. 20 p.

Graham, R. T.; Jain, T. B. 2004. Past, Present and Future Role of Silviculture in Forest Management. In: Forest Service, RMRS-P-34. 1-14.

Graham, R. T.; Harvey, A. E.; Jurgensen, M. F. et al. 1994. Managing coarse woody debris in forests of the Rocky Mountains. Forest Service, Research Paper INT-RP-477. 13 p.

Graham, R. T.; Jain, T. B. 2006. Ponderosa pine ecosystems. In: Forest Service, General Technical Report PSW-GTR-198. 1-32.

Gruell, G. E.; Schmidt, W. C.; Arno, S. F.; et al. 1982. Seventy years of vegetative change in managed ponderosa pine forest in western Montana: implications for resource management. Forest Service, General Technical Report GTR-INT-130. 42 p.

Hann, W. J.; Jones, J. L.; Karl, M. G., et al. 1997. Chapter 3: Landscape dynamics of the Basin. In: Forest Service, General Technical Report PNW-GTR-405. 338-1055.

Harvey, A. E.; Jurgensen, M. F.; Larsen, M. J.; et al. 1987. Decaying organic materials and soil quality in the Inland Northwest: a management opportunity. Forest Service, General Technical Report INT-225. 15 p.

Harvey, A. E.; Graham, R. T.; McDonald, G. I. 1999. Tree species composition change- Forest soil organism interaction: potential effects on nutrient cycling and conservation processes in interior forests. In: Forest Service, General Technical Report PNW-GTR-461. 137-145.

Harvey, A. E.; Graham, R. T.; McDonald, G. I. 2000. Fire/decay: managing codependent processes across the landscape. In: Joint fire science conference and workshop. International Association Wildland Fire. 179-189.

Hessburg, P. F.; Mitchell, R. G.; Filip, G. M. 1994. Historical and current roles of insects and pathogens in eastern Oregon and Washington forested landscapes. In: Eastside forest ecosystem health assessment, Volume III, Forest Service, Pacific Northwest Research Station. 72 p.

Hungerford, R. D.; Harrington, M. G.; Frandsen, W. H.; et al. 1991. Influence of fire on factors that affect site productivity. In: Forest Service, General Technical Report INT-280. 32-50.

Hutchison, S. B.; Winters, R. K. 1942. Northern Idaho forest resources and industries. Miscellaneous Publication 508. U.S. Department of Agriculture. 75 p.

Jain, T. B.; Graham, R. T.; Morgan, P. 2004. Western white pine growth relative to forest openings. *Canadian Journal Forest Research* 34: 2187-2197.

Ketcham, D. A.; Wellner, C. A.; Evans, Jr. S. S. 1968. Western white pine management programs realigned on Northern Rocky Mountain National Forests. *Journal Forestry* 66(4): 329-332.

Lewis, J. G. 2005. *The Forest Service and the Greatest Good: A Centennial History*. Durham, NC: Forest History Society. 286 p.

Mika, P. G.; Moore, J. A. 1990. Foliar potassium explains Douglas-fir response to fertilization in the Inland Northwest. *Water, Air, and Soil Pollution* 54: 477-491.

Minore, D. 1979. Comparative autecological characteristics of northwestern tree species---A literature review. Forest Service, General Technical Report PNW-87. 72 p.

Pearson, G. A. 1950. Management of ponderosa pine in the southwest as developed by research and experimental practice. Monograph 6. Forest Service. 218 p.

Reinhardt, E. D.; Brown, J. K; Fischer, W. C.; et al. 1991. Woody fuel and duff consumption by prescribed fire in northern Idaho mixed conifer logging slash. Forest Service, Research Paper INT-443. 22 p.

Ryan, Kevin C. 1982. Burning for site preparation on steep-slope helicopter-logged clearcuts in northwestern Montana. In: Symposium proceedings: site preparation and fuel management on steep terrain. Pullman, WA: Washington State University. 25-33.

Shepperd, W. D.; Battaglia, M. A. 2002. Ecology, silviculture, and management of Black Hills ponderosa pine. Forest Service, General Technical Report RMRS-GTR-97. 112 p.

Shinnen, D. J.; Baker, W. L. 1997. Non equilibrium dynamics between catastrophic disturbances and old-growth forests in ponderosa pine landscapes of the Black Hills. *Conservation Biology* 11(6): 1276-1288.

Smith, H. Y.; Arno, S. F. 1999. Eighty-eight years of change in a managed ponderosa pine forest. Forest Service, General Technical Report RMRS-GTR-23. 55 p.

Steen, H. K. 1976. *The U.S. Forest Service*. Seattle and London: University of Washington Press. 356 p.

Stein, S. J. 1988. Explanations of imbalanced age structure and scattered distribution of ponderosa pine within a high-elevation mixed coniferous forest. *Forest Ecology and Management* 25: 139-153.

Stewart, O. C. 1951. Burning and natural vegetation in the United States. *Geographical Review* 41: 317-320.