23

Restoring Dry and Moist Forests of the Inland Northwestern United States

Theresa B. Jain and Russell T. Graham

CONTENTS

23.1 Introduction

The complex topography of the Inland Northwestern United States (58.4 million ha) interacts with soils and a highly variable climate to provide a mosaic of dry and moist mixed conifer forest settings. Approximately 20% of the area is covered by dry forests dominated by *Pinus ponderosa, Pseudotsuga menziesii* and contains a diversity of lower vegetation ranging from a grass savannah on the less-productive sites to shrub and forb dominated vegetation on the more-productive sites. An estimated 18% of the area is covered by moist mixed conifer forests with some places growing up to 10 different conifer species, in addition to a diversity of surface vegetation often dominated by shrub and forb vegetation (e.g., with *Pinus monticola* and *Tsuga heterophylla*).

In the dry forests, historically frequent surface and mixed fires burned over 75% of the area of dry forests; however, successful fire exclusion, harvesting and a cool and moist period allowed dense stands of *Abies grandis, P. menziesii*, and small *P. ponderosa* to develop. Historically, forest canopies and their nutrients were located well above the soil surface; fine roots and microbial activity were located deep in mineral soils, thus protecting them from wildfire. In contrast, the *Abies* and *Pseudotsuga* dominated forests of today contain nutrient-rich crowns that extend to the forest floor. Nutrients and microbial activity are located near the soil surface, increasing their susceptibility to loss from wildfire.

In the moist forests, fire exclusion, harvesting, and the introduction of *Cronartium ribicola* (a stem disease) from Europe are the primary change agents. In the Northern Rocky Mountain moist forests, early-seral *P. monticola* has nearly been extirpated and mid- to late-seral conifers now dominate. In the moist forest of the eastern Cascades Mountains in Washington and Oregon, an increase in homogeneity of mid-seral forests containing *A. grandis, T. heterophylla,* and *P. menziesii* has occurred, encouraged by the harvesting of *Larix occidentalis* and *P. ponderosa*. These changes have elevated the risk to large-scale insect and disease epidemics and uncharacteristic wildfires.

Successful restoration strategies in dry and moist forests should apply concepts learned from the past but we must also be cognizant of the changes that have occurred not only in the tree component but also those occurring in the lower vegetation and soil and across landscapes. The reintroduction of fire alone is not the answer to restore these forests, because today we have ever-changing social desires, changes in soil microbial and chemical properties, potential changes in long-term climate, and both native and exotic diseases and insects that prevent reverting to the past. Rather a multiscale approach applied that integrate these conditions over short- and long-term temporal (decades to centuries) and spatial (site to landscape) scales may provide a template for restoring the moist and dry forests of the Inland Northwestern United States.

23.2 Forests of the Inland Northwest

The Inland Northwestern United States (58.4 million ha) is defined by the Bitterroot, Selkirk, Cabinet, Salmon River, Lemhi, Steens, Purcell, Cascade, and Blue mountain ranges with elevations over 1500 m (Figure 23.1). Within these ranges, the valley bottoms can be low (225 m) and the topography steep. This rough and complex topography results in a variety of forest settings ranging from steep slopes, in narrow V-cut canyons, to gentle rolling slopes, in wide river valleys. During the Pleistocene, alpine glaciers shaped the canyons and valleys; today, a mantle of glacial till covers these glaciated landscapes. Much of the fine silt washed out by the glaciers was redeposited by winds, leaving deep layers of loess over many landscapes. Some 12,000 to 15,000 years ago, Glacial Lake Missoula repeatedly filled and emptied, flooding most of northern Idaho and eastern Washington. The eruption of prehistoric Mt. Mazama 7500 years ago formed Crater Lake in Oregon and deposited a layer of ash up to 62 cm thick across the area. Disturbance events continually modify the granitic and metasedimentary rocks, ash, and loess deposits, giving rise to diverse soils (Quigley et al. 1996).

Moist marine air originating from the Pacific Ocean moderates temperatures within the Inland Northwestern United States, while continental dry and cold air from the north and east brings cold weather in winter and hot weather in summer. During the summer, these air masses interact and bring convective precipitation, lightning, and cool periods. Dry Arctic air in the winter brings damaging frosts and cold temperatures (≤5°C) that alternate with wet warm periods. This highly variable climate interacts with the heterogeneous and rugged topography to create mosaics of dry and moist mixed conifer forests (Franklin and Dryness 1973; Foiles et al. 1990; Graham 1990; Hann et al. 1997).

Until 1900, forests covered over 47% of the Inland Northwest (Figure 23.1). Dry forests occupied an estimated 11 million ha, dominated by *P. ponderosa,* and moist forests covered

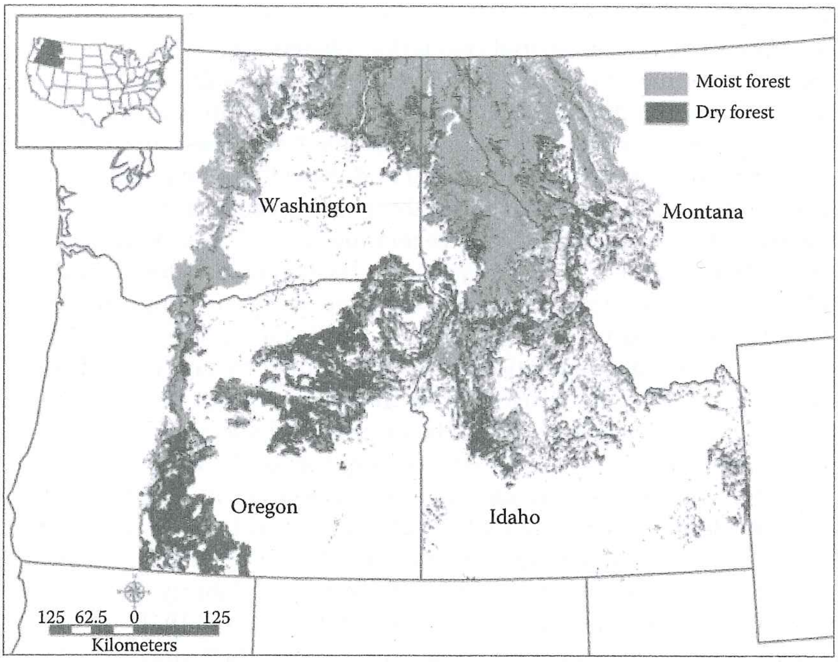

FIGURE 23.1
There are 58,361,400 ha in the Inland Northwestern United States framed by the Columbia River Basin. The topography is rugged, ranging from the Cascade Mountains in the West to the Bitterroot and Salmon River Mountains in Idaho. Elevations in the region range from 225 m to over 3000 m. Dry and moist forests make up 90% of the forests occurring in the Inland Northwestern United States. The moist forests occur primarily in northern Idaho, northwestern Montana, and northeastern Washington and along the eastern slopes of the Cascade Mountains in Washington and Oregon. The dry forests are dispersed throughout the region. (From Hann, W.J., J.L. Jones, M.G. Karl et al. In *An Assessment of Ecosystem Components in the Interior Columbia Basin and Portions of the Klamath and Great Basins: Volume II*, edited by T.M. Quigley and S.J. Arbelbide, 338–1055. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 1997.)

an estimated 10.5 million ha (18%). The United States Forest Service and the Bureau of Land Management administer more than 50% of both the dry and moist forests (Quigley et al. 1996). Other federal and state agencies administer approximately 5% of these forests and several industrial and nonindustrial owners manage smaller tracts. Both the moist and dry forests have lost many native structures (large early-seral tree component) and processes (native fire regimes) that were integral in maintaining these systems and the myriad plants, animals, and uses they supported (Quigley et al. 1996). Similar series of events occurred in other locales with similar forest types located in British Columbia, Canada, and throughout the western United States (e.g., Hessburg et al. 1994; Burton and MacDonald 2011; Franklin and Johnson 2012).

23.3 Dry Forests

Dry forests occur across a wide range of elevations in northeastern Washington, northeastern Oregon, central and southern Idaho, and south-central Oregon (Figure 23.1) (Hann et al. 1997). These forests are complex, depending on the weather, physical setting, disturbances, forest succession, and potential vegetation. Depending on the combination of these components, multiple tree, shrub, and forb species can vary within a given site and across landscapes. Soil parent materials include granites, metasedimentaries, glacial tills, and basalts. Tree and plant communities that are considered dry forests occur in places that are water-limited and often are subject to drought and these forests can also occur on shallow soils which also influence the nutrient and water holding capacity. *P. menziesii, P. ponderosa,* and dry *A. grandis/Abies concolor* potential vegetation types (PVTs) dominate these settings (Hann et al. 1997). Potential vegetation type is a classification system based on the physical and biological environment characterized by the abundance and presence of vegetation in the absence of disturbance (Daubenmire and Daubenmire 1968; Pfister et al. 1977; Cooper et al. 1991). Potential vegetation types are defined by and named using indicator species (tree and surface vegetation) that grow in similar environmental conditions. When *L. occidentalis* is present it is always an early successional species (dominant after disturbance). *A. grandis/A. concolor* are late-successional species and are more shade-tolerant than *P. ponderosa* and *L. occidentalis. P. ponderosa* and *P. menziesii* can play both late- and early-successional roles, depending on the PVT (Daubenmire and Daubenmire 1968). Places that support forests but have limited water availability, *P. ponderosa* plays a late-successional role. In contrast on productive sites with more water availability, *P. ponderosa* plays an early-successional role in *A. grandis/A. concolor* and *P. menziesii* PVTs. *P. menziesii* plays a mid-successional role in *A. grandis/A. concolor* and also in moist forests. Surface vegetation in the dry forests includes shrubs (*Arctostaphylos uva-ursi, Ceanothus* spp., *Purshia tridentata, Symphoricarpos albus, Physocarpus malvaceus*), grasses (*Calamagrostis rubescens, Bromus vulgaris*), and sedges (*Carex* spp.) (Foiles et al. 1990; Hermann and Lavender 1990; Oliver and Ryker 1990).

Disturbances and physical setting historically maintained a variety of structural and successional stages (Table 23.1). Fire, insects, diseases, snow, ice, and competition thinned these forests, and surface fires provided opportunities for regeneration (Foiles et al. 1990; Hermann and Lavender 1990; Oliver and Ryker 1990; Jain et al. 2012; Stine et al. 2014). Approximately 18% of the area was in a grass, forb, and shrub stage for long (100s years) periods and 15% contained early-seral *P. ponderosa* with diameters ranging from 5 to 80 cm (Meyer 1938). As these forests aged, mid-seral multistoried forest structures developed. Three percent of the area contained late-seral *P. menziesii* and *A. grandis/A. concolor* with multiple canopies. Large, widely-spaced (~250 trees per ha) *P. ponderosa* often dominated 21% of the dry forests, with the plurality of diameters ranging from 30 to 60 cm (Figure 23.2a) (Daubenmire and Daubenmire 1968; Hann et al. 1997). Late-seral single-storied forests containing *P. menziesii* and *A. grandis/A. concolor* complexes dominated some settings (2%).

23.3.1 Dry Forest Change

The dry forests were adapted to a wide range of site conditions and short-term climate variation. These characteristics created an ecosystem that appeared to be long-lived and relatively resilient to disturbances (fire, insect, and disease) (Harvey et al. 1994). Since 1900, approximately 8% (600,000 ha) of the dry forests have been converted to agriculture, urbanization, and industry (Hann et al. 1997). Fire exclusion, harvesting, and changes in fire regime altered the composition and structure of the remaining dry forests (Everett et al. 1994; Hann et al. 1997; Lewis 2005; Jain et al. 2012). The area burned by surface fires has decreased from an estimated 80% to less than 50% of the area. The mean fire return

TABLE 23.1

Historical (1850–1900) and 1991 Distribution of Forest Structures within the Dry and Moist Forests of the Inland Northwest

Forest Structure	Historical (%)	1991 (%)	Change (%)
Dry Forests			
Grass/forb/shrub	18	1	−17
Early seral intolerant	15	14	−1
Early seral tolerant	3	3	0
Mid-seral intolerant	21	35	+14
Mid-seral tolerant	8	22	+14
Late seral—intolerant single story	21	5	−16
Late seral—tolerant single story	2	3	+1
Late seral—intolerant multistory	9	8	−1
Late seral—tolerant multistory	3	9	+6
Moist Forests			
Northern Rocky Mountain Region (NRM)			
Early seral—single story	29	20	−9
Mid-seral	41	69	+28
Late seral—single story	11	3	−8
Late seral—multistory	19	8	−11
Eastern Cascade Region (Northern Cascade/ Southern Cascade)			
Early seral—single story	23/25	32/15	+9/−10
Mid-seral	37/34	48/37	+11/+3
Late seral—single story	9/9	4/29	−5/+20
Late seral—multistory	31/32	16/19	−15/−13

Source: Adapted from Hann, W.J. et al. In *An Assessment of Ecosystem Components in the Interior Columbia Basin and Portions of the Klamath and Great Basins: Volume II*, edited by T.M. Quigley and S.J. Arbelbide, 338–1055. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 1997.

interval has also increased from less than 20 years to 40–80 years. Mixed-fires (combination of surface and crown fires) have increased from 5% to an estimated 35% of burned area and the mean fire return interval has increased from 45 to 60 years. A similar increase in crown fires has also occurred (Hann et al. 1997). Mid-seral structures have increased (from an estimated 29%–57% of the area), often containing dense areas of small *P. ponderosa, P. menziesii*, or *A. grandis/A. concolor* (Table 23.1; Figure 23.2b). The proportion of the dry forests occupied by late-seral single-storied *P. ponderosa* has declined from 21% to 5% (Figure 23.2a). In addition, small diameter trees have encroached and now occupy all but one percent of the dry forests that formerly were covered by grasses, forbs, and shrubs (Figure 23.2b). The dominant tree species has changed from *P. ponderosa* to *P. menziesii* or *A. grandis/A. concolor,* changing the character and canopy architecture of the forest.

The shift in species composition from *P. ponderosa* to *Abies* and *Pseudotsuga* dominated forests changed litter type and quantity, which changed soil chemistry, microbial processes, and ectomycorrhizal relationships (Rose et al. 1983). For example, decomposed true firs create white rotten wood, which rapidly disperses into the soil and is quickly consumed by decomposers. In contrast, decomposed *P. ponderosa* and *L. occidentalis* create brown rotten wood, which can persist in soil for centuries and has been shown to retain nutrients

(a)

(b)

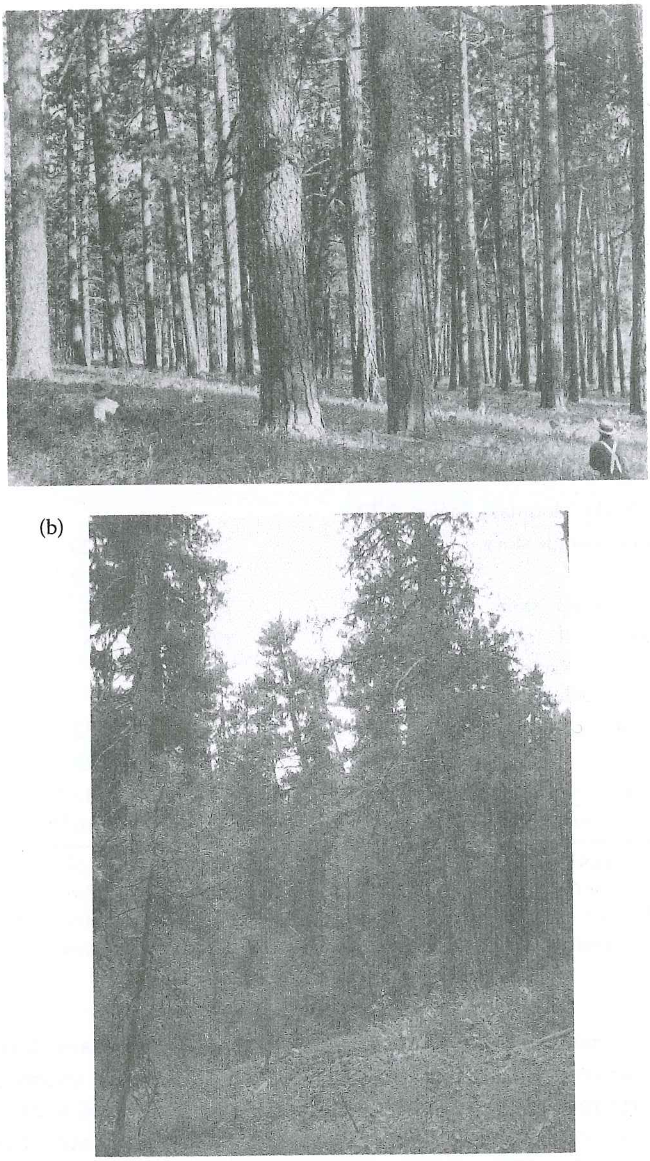

FIGURE 23.2
A historical (1850–1900) (*Pinus ponderosa*) stand exhibiting a lush understory layer of forbs and grasses (a). These conditions were maintained by frequent (<20 year) nonlethal surface fires. Within the dry forests, successful fire exclusion and harvesting have allowed dense stands of vegetation to develop (b). (USDA Forest Service photographs.)

and hold water (Larsen et al. 1980; Harvey et al. 1987). *L. occidentalis* and *P. ponderosa* tend to be deep-rooted, in contrast to the relatively shallow-rooted *Pseudotsuga* and *Abies*, which have abundant feeder roots and ectomycorrhizae in the shallow soil organic layers (Minore 1979; Harvey et al. 1987). *P. ponderosa* and *L. occidentalis* forests are generally tall and self-pruning, even in moderately dense areas. 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) *P. menziesii* and *A. grandis/A. concolor* generally do not self-prune. This canopy architecture favors lower crown base heights, higher crown densities, and canopies with higher nutrient (especially potassium) content than it occurs in *L. occidentalis* and *P. ponderosa* dominated forests (Figure 23.2b) (Minore 1979; Harvey et al. 1999).

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 in the mineral soils (Marschner and Marschner 1996; Harvey et al. 1999). In the absence of fire, bark slough, needles, twigs, and small branches accumulated on the forest floor allowing ectomycorrhizae and fine roots of all species to concentrate in the surface mineral soil and thick organic layers (Harvey et al. 1994; Hood 2010; Jain et al. 2012).

Harvesting the *L. occidentalis* and *P. ponderosa* and the ingrowth of *A. grandis/A. concolor* and *P. menziesii* in the dry forests together facilitated the accumulation of both above- and below-ground biomass and their nutrient content close to the soil surface (Harvey et al. 1986; Hood 2010). Even low-intensity surface fires now consume the surface organic layers, killing fine roots, volatilizing nutrients, killing trees, and increasing soil erosion potential (Debano 1991; Hungerford et. al. 1991; Ryan and Amman 1996; Robichaud et al. 2000; Hood 2010). In addition, fir ingrowth creates nutrient-rich ladder fuels that facilitate crown-fire initiation, increasing the likelihood of nutrient loss (Van Wagner 1977; Minore 1979; Harvey et al. 1999). The risk of nutrient loss is great on infertile sites, because dense areas of late-seral species are more demanding of nutrients and water than the historical areas dominated by widely-spaced early-seral species (Minore 1979; Harvey et al. 1999).

23.4 Moist Forests

Moist forests of the Inland Northwestern United States occur in two locations, the eastern Cascade Mountains (east of the Cascade Crest in Washington and Oregon) and the Northern Rocky Mountains (northeastern Washington and Oregon, northern Idaho, and western Montana) (Figure 23.1). They grow at elevations ranging from 460 to 1600 m and occasionally occur at elevations up to 1800 m (Foiles et al. 1990; Graham 1990; Packee 1990; Schmidt and Shearer 1990; Hann et al. 1997) (Figure 23.1). These forests are influenced by a maritime climate with wet winters and dry summers. Most precipitation occurs during November through May, with amounts ranging from 500 to 2300 mm (Foiles et al. 1990; Graham 1990; Packee 1990; Schmidt and Shearer 1990). Precipitation comes as snow and prolonged gentle rains, accompanied by cloudiness, fog, and high humidity. Rain-on-snow events are common from January to March. A distinct warm and sunny drought period occurs in July and August with rainfall in some places averaging less than 25 mm per month.

Soils that maintain these forests include, but are not limited to, Spodosols, Inceptisols, and Alfisols. A defining characteristic of the Northern Rocky Mountains is the layer of fine-textured ash (up to 62 cm thick) that caps the residual soils. The ash soils and loess deposits throughout the moist forests are continually being modified by disturbance events giving rise to soils with differing levels of productivity (Foiles et al. 1990; Graham 1990; Packee 1990; Schmidt and Shearer 1990). The combination of climate, topography, parent material, soils, weathering, and ash depth (unique to the Northern Rocky Mountains) creates the most productive of all forests occurring within the Inland Northwest.

The historical vegetation complexes in the Cascades and Northern Rocky Mountains ranged from early- to late-seral, and occurred within a landscape mosaic possessing all possible combinations of species and seral stages. The PVT's in the Northern Rocky Mountains include *Thuja plicata, T. heterophylla* and *A. grandis* with *P. monticola, L. occidentalis, Pinus contorta, P. menziesii* and *P. ponderosa* are always the early- and mid-seral species (Daubenmire and Daubenmire 1968; Hann et al. 1997). The eastern Cascades PVTs include *T. plicata, T. heterophylla, A. grandis, Abies amabilis,* and *Abies procera*. The early- and mid-seral species include *P. contorta, P. menziesii* and *P. ponderosa* while *P. monticola* and *L. occidentalis* are less abundant when compared to the Northern Rocky Mountains (Franklin and Dyrness 1973; Lillybridge et al. 1995).

Lush ground-level vegetation is the norm in the moist forests. The vegetation complexes are similar to those occurring on the west-side of the Cascade Mountains and in some Pacific coastal areas. Tall shrubs include *Acer circinatum, Achylys triphylla, Acer glabrum, Alnus sinuata, Oplopanax horridus, Rosa* spp., *Ribes* spp., *Vaccinium* spp., and *Salix* spp. Forbs include *Actaea rubra, Adenocaulon bicolor, Asarum caudatum, Clintonia uniflora, Cornus canadensis,* and *Coptis occidentalis*. Phytogeographic evidence indicates that some plant populations on the west side of the Cascade Mountains also occur as disjunct populations in the moist forests. For example, low-elevation riparian areas in northern Idaho contain disjunct populations of *Alnus rubra, Cornus nuttallii, Symphoricarpos mollis, Selaginella douglasii,* and *Physocarpus capitatus* (Foiles et al. 1990; Graham 1990; Packee 1990; Schmidt and Shearer 1990).

Snow, ice, insects, disease, and fire, when combined, created heterogeneity in patch sizes, forest structures, and compositions. Ice and snow created small gaps and openings, thinning forest densities and altering species composition (Figure 23.3a). Native insects (e.g., *Dendroctonus* spp.) and diseases (e.g., *Armillaria* spp., *Arceuthobium* spp.) infected and killed the very old or stressed individuals, which tended to diversify vegetation communities (Figure 23.3b) (Hessburg et al. 1994; Rippy et al. 2005). A mixed-fire regime best defines the role fire played in creating a mosaic of forest compositions and structures. Nonlethal surface-fires occurred at relatively frequent intervals (15–25 years) in a quarter of the area (Figure 23.4a). Lethal crown-fires burned about a quarter of the area at intervals of 20–150 years but occasionally extended to 300 years (Figure 23.4a and c). The mixed-fire regime occurred across the rest of the moist forests at 20 to150 year intervals. Fires typically started burning in July and were usually out by early September (Hann et al. 1997).

23.4.1 Moist Forest Change

The current distribution of successional-stage, forest structure, species composition, and disturbance regimes differs from the historic (1850–1900) patterns of the moist forest (Hann et al. 1997). In some settings, the mixed-fire regime maintained closed canopy conditions, which allowed for the mid-seral stage to develop into late-seral multistory stages (Hann et al. 1997). The late-seral multistory structure, which typically developed in cool, moist bottoms and basins, has decreased by about half in the last century (Table 23.1). The early-seral single-story stands that once occupied an estimated 25%–30% of the area now occupy only 9%–10% of the area, except in the northern Cascades (Washington) where they increased in abundance. The mid-seral stages have generally increased in abundance in the Northern Rocky Mountains and to a lesser degree in the eastern Cascades.

Species composition has shifted in the Northern Rocky Mountains (Hann et al. 1997; Neuenschwander et al. 1999; Fins et al. 2001); before 1900, *P. monticola* (early- to mid-seral species) dominated, often representing 15%–80% of the trees within stands (Figure 23.5)

(a)

(b)

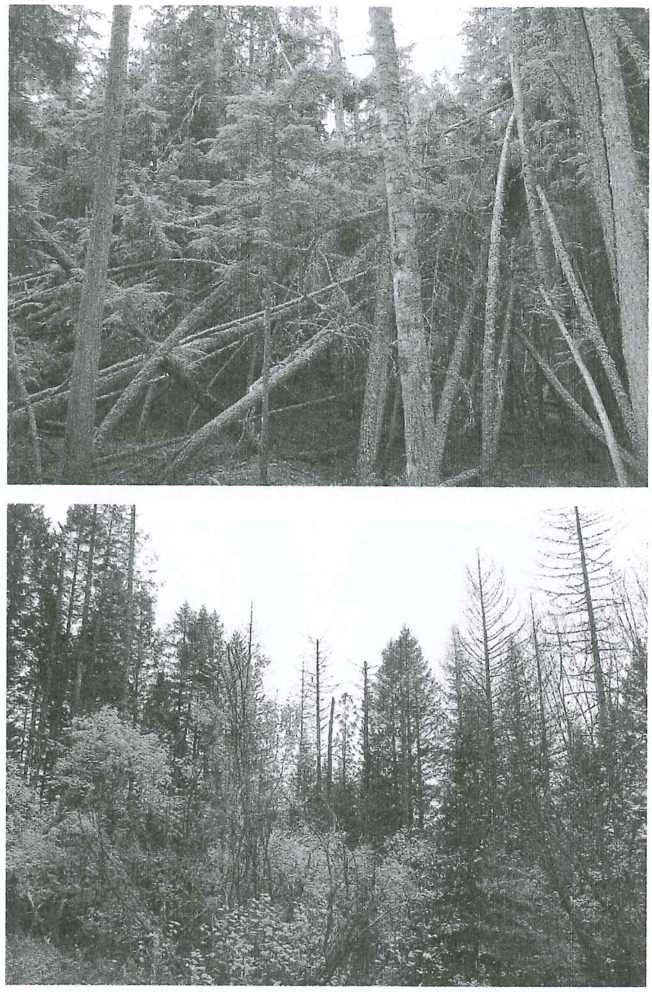

FIGURE 23.3
Ice and snow damage create small gaps and openings, decrease forest densities, and alter species composition (a). Historically (1850–1900), in the moist forests, diseases (e.g., *Armillaria* spp., *Arceuthobium* spp.) attacked the very old, unthrifty, or stressed individuals (b). These disturbances stabilized and diversified vegetation communities. Currently, in the changed systems, epidemics of these disturbances often occur. (USDA Forest Service photographs.)

(Fins et al. 2001). This species is resistant to many endemic insects and diseases; it is long-lived (300 years) and can grow across 90% of the moist forest environments. It is a prolific seed producer after age 70 and is the only moist-forest conifer with seed remaining viable for up to three years, which allows it to regenerate abundantly after disturbance (Haig et al. 1941; Graham 1990). It is broadly adapted genetically (an ecological generalist) to the environment (Rehfeldt et al. 1984) and it is moderately shade-tolerant, allowing it to establish and develop within a wide range of canopy openings (Haig et al. 1941; Graham 1990; Jain et al. 2002, 2004). *P. monticola* often reaches 30 m in height or greater within 50 years of establishment (Graham 1990). *L. occidentalis* and *P. ponderosa* also occurred in the early- and mid-seral structures, but declined along with *P. monticola* and were succeeded by *A. grandis*, *P. menziesii*, and *T. heterophylla* (Hann et al. 1997; Atkins et al. 1999).

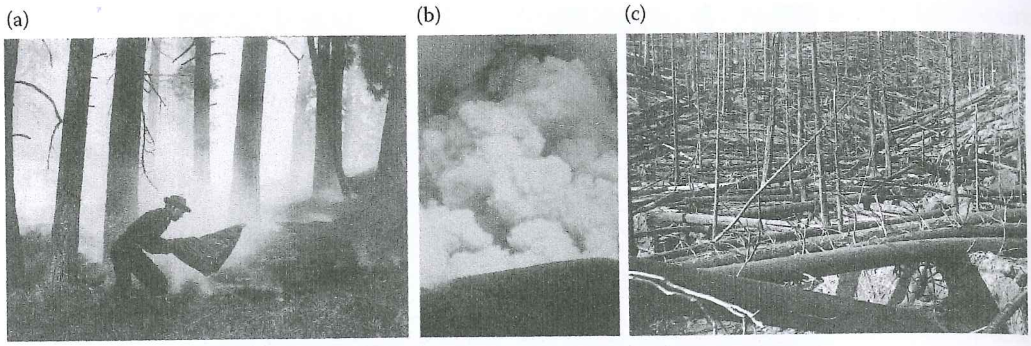

FIGURE 23.4
Historically (1850–1900), in the moist forests, nonlethal surface fires (a) occurred at relatively frequent intervals (15–25 years) in 25% of the moist forests while lethal crown fires (b and c) occurred over 25% of the forests at 300 years intervals. (USDA Forest Service photographs.)

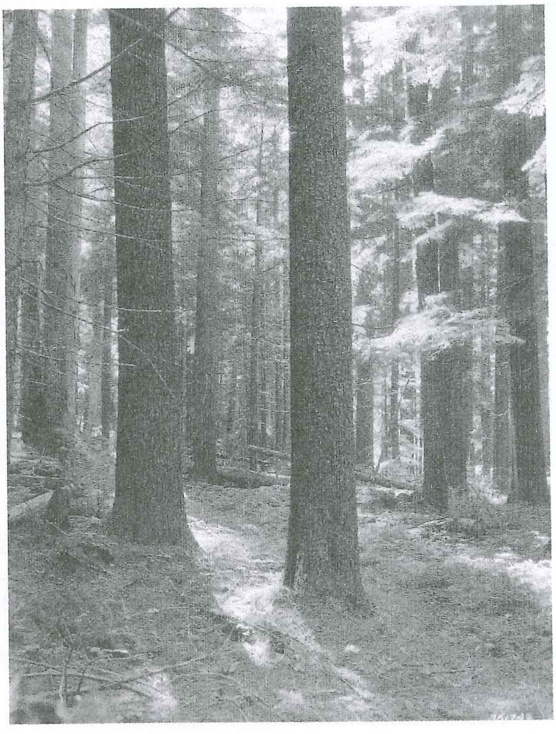

FIGURE 23.5
Historically (1850–1900), 25%–50% of the moist forests were dominated by *Pinus monticola* and 15%–80% of the trees within stands were *Pinus monticola*. This photo shows 150–180-year-old *Pinus monticola* (circa 1935) growing in northern Idaho. (USDA Forest Service photograph.)

The eastern Cascades had limited amounts of *P. monticola* and *L. occidentalis*; therefore, *P. ponderosa*, *P. contorta*, and *P. menziesii* played more of a role in occupying the early- to mid-seral successional stages.

Native insects and pathogens occurred in these forests, but recent (1991) activity levels far exceed those of the past (Hessburg et al. 1994). Within the *A. grandis*/*A. concolor*

PVT, fire maintained landscapes that contained a plurality of early-seral *P. ponderosa* and *L. occidentalis;* the insects, *Dendroctonus pseudotsugae, Choristoneura occidentalis,* and *Orgyia pseudotsugata* were generally endemic. But they are often epidemic in the current forests dominated by *A. grandis/A. concolor* and *P. menziesii* (Hessburg et al. 1994). Similarly, the diseases *Armillaria* spp. and *Phellinus weirii* were historically endemic, but the current fir-dominated forests make epidemics of these diseases more common (Hessburg et al. 1994; Hann et al. 1997). As in many forest ecosystems in the western United States, effective fire exclusion contributed to these changes. Historically, 25% of the area had surface fires, 50% mixed fires, and 25% stand-replacing crown fires. Today, crown fires burn approximately 60% of the areas in these forests and only 15% are burned by surface fires and 20% are burned by mixed fires (Hann et al. 1997).

Although fire exclusion played a role in altering forests in the Northern Rocky Mountains, introduction of a European stem rust, *C. ribicola,* caused the greatest change (Figure 23.6a). The rust infects all five-needle pines, and subsequently decimated the abundant *P. monticola* (Figure 23.6b). Because the rust killed so many trees, the majority of surviving pines were harvested under the assumption they too would succumb to the rust (Ketcham et al. 1968). *A. grandis* and *P. menziesii* readily filled the niche *P. monticola* once held. In the eastern Cascades, blister rust was less severe since *P. monticola* was not the dominant species, thus fire exclusion and harvesting were more important agents in altering these forests. Harvesting removed the early-seral, shade-intolerant species (e.g., *P. ponderosa, L. occidentalis*) that were resistant to fire and other disturbances. Partial canopy removal and minimal soil surface disturbance in these harvests were ideal for *P. menziesii* and *A. grandis* which regenerated aggressively, rather than the shade-intolerant *Pinus* and *L.* species. Fire exclusion also prevented the creation of canopy openings and receptive seedbeds for the regeneration of *Pinus* and *Larix.* Similar to the dry forests, high canopies (>50 m) of *P. monticola, L. occidentalis, P. ponderosa* and other early- and mid-seral species currently are absent. In their place, the present forest structure and composition (*A. grandis* and *P. menziesii*) favor the compression of nutrients, microbial

(a) (b)

FIGURE 23.6
White pine blister rust (*Cronartium ribicola*) canker occurring on a young *Pinus monticola* (a). A mid-aged (70–80 years) stand of *Pinus monticola* experiencing extreme mortality from blister rust (b). (Photograph A is USDA Forest Service and photograph B is from USDA. Forest Service, Ogden, Archive, Bugwood.org)

processes, and root activity toward the soil surface (Harvey et al. 1999). When wildfires occur, surface organic layers can be consumed, decreasing the nutrition and microbial processes important for sustaining these forests.

23.5 Restoration Approaches

When forests fail to recover after disturbances; human intervention may be required to alter a forest's trajectory and create opportunities for self-renewal and postdisturbance recovery (Stanturf 2004). Unlike timber management methods, treatments directed at forest restoration are not always similar, thus there is a need to develop techniques that reinstate processes that lead to restored ecosystems. Some scientists hypothesize that returning forest structures and compositions (vegetative characteristics, fire regimes, and species mixes) similar to those that existed prior to European settlement will reset a forest to a more functional condition (Moore and Covington 1999; Caprio and Graber 2000; Keane et al. 2009; Churchill et al. 2013). They use these reference conditions to help in formulating restoration targets and management strategies. However, simply returning a forest to a state that occurred at specific point-in-time and assume a forest will function as it did in the past is imperfect for several reasons. A serendipitous series of events (moisture, seed availability, and disturbance) occurred decades to centuries ago that created the environments for a forest to regenerate and develop (Haig et al. 1941; Oliver and Larson 1990). These events are not static but they fluctuate through time and space (Marshall 1928). The climate and disturbances that created a forest's structure and composition in the past differs from those of today and most likely will differ from those of the future. For example, American Indian burning contributed to historical fire regimes and such fires will most likely not occur now or in the future because of massive changes that have occurred over the last 100 years to the biophysical, social, and economic environments of western forests (Pyne 2010; Jain et al. 2012). For example, hunting, fishing, and gathering were primary historical land uses and now society values roads, recreation, forest products, and wildlife habitat and dislike many aspects associated with forest clearcutting (Reynolds et al. 1992; Bliss 2000; DellaSala 2003; Mercer 2005; De Groot et al. 2013). For these reasons, replicating forest conditions of the past may not be the best approach to use in restoration strategies but rather scientists and managers need to develop a pragmatic view of forest restoration that recognizes how forests and their social and economic context have changed over the last 100 years (Stanturf 2004).

This philosophy holds true when developing restoration strategies for the dry and moist mixed-conifer forests in the northern Rocky Mountains. In the last 100 years, these forests have changed and in some cases it may take centuries to achieve restoration goals. However, by integrating past and present knowledge, scientists have identified some ecological elements that if restored can increase opportunities for self-renewal (e.g., Harvey et al. 1994; Neuenschwander et al. 1999; Hessburg 2000; Fins et al. 2001; Hood 2010; Jain et al. 2014). Increasing landscape heterogeneity alters how disturbances move across a landscape and allows for wildlife corridors that will help maintain genetic diversity. Within stand, heterogeneity provides vegetative regeneration opportunities, snags, variable canopy cover, and down woody debris which are important for many wildlife species. Because early- and mid-seral species tend to be resistant to both endemic and introduced disturbances, their increased presence and abundance are usually central to restore both the dry and moist forests of the northern Rocky Mountains. Also, the abundance of late-seral regeneration in both forests confounds most restoration strategies as does the changes to the forest floor (deep duff) and species shifts that have altered soil properties (from red rot to white rot dominated systems). The challenge for those developing restoration strategies is to develop them within the context of changing societal values, economic uncertainty, and an indeterminate future climate.

23.5.1 Increase Landscape Heterogeneity in Dry and Moist Forests

In moist and dry mixed conifer forests, endemic disturbances (wind, ice, snow, disease, insects, and wildfire) coupled with the native vegetation, complex topography, and soils historically created and sustained various patch sizes and patch mosaics throughout the landscape (Marshall 1928; Hessburg et al. 2001; Moritz et al. 2011; Perry et al. 2011). These disturbances also created fine scale (≤1.0 ha) mosaics of canopy gaps, burned surfaces, live trees, snags, decadent large trees, and many other forest structures and vegetative compositions. Conversely, timber management that was practiced for several decades in the Inland Northwest tended to homogenize and regulate forests to achieve orderly and predictable amounts of forest products (Puettmann et al. 2008). This homogeneity created conditions where disturbances were no longer self-regulating, which has led to large and severe disturbances (wildfires and bark beetle epidemics) (Hessburg et al. 2007; Bentz et al. 2009; Littell et al. 2009; Mortz et al. 2011; Stephens et al. 2014). Moreover, in moist mixed-conifer forests, *A. grandis, T. heterophylla,* and *P. menziesii* replaced *P. monticola, L. occidentalis,* and *P. ponderosa* making these forests highly susceptible to epidemics of insects and diseases and catastrophic wildfire. It will undoubtedly take from decades to centuries to restore these early- and mid-seral species to their historic presence and abundance, but by increasing landscape heterogeneity and structural diversity within these forests the extent and severity of disturbances will likely diminish. However, the economic and social constraints prevent restoring an entire landscape creating a situation where scientists and managers need to develop approaches for prioritizing treatment areas (Hessburg et al. 2000; Hann and Bunnell 2001; Reynolds and Hessburg 2005; De Groot et al. 2013; Bollenbacher et al. 2014).

Several scientists suggest using a multiscale planning process to identify restoration opportunities and treatment priorities (e.g., Hann and Burnell 2001; Reynolds and Hessburg 2005; De Groot et al. 2013). At the broadest scale (e.g., subbasin ≈ 750,000 ha), managers can identify endangered species habitat, old growth, stream networks for anadromous fish, and social and economic infrastructure (population trends, cities, towns, and communities) (Hessburg et al. 2000). This analysis can identify areas that already have good road access; areas of concern that need protected, and identify priority areas for watershed restoration. At the mid-scale or river basins forest composition, distributions, and similar elements can be addressed. For example, in the Coeur d'Alene River Basin (≈350,000 ha) of the northern Rocky Mountains, Jain et al. (2002) related historical patterns of *P. monticola* and physical characteristics that identified places for restoring the species. At the fine scale (≈1000 ha), Camp et al. (1997) identified fire refugia as a function of physical landscape attributes in the eastern Cascades of Washington. Managers could use such techniques in both dry and moist forests to plan and prioritize restoration opportunities reflecting the key issues and values that may affect forest management.

Once landscape priorities are identified, landscape heterogeneity can be addressed by the strategic placement of forest treatments to create treatment mosaics that can be dispersed across the large landscape. For example, at Priest River Experimental Forest in northern Idaho, USA, scientists created forest mosaics of different sized vegetative patches distributed across the landscape (Figure 23.7) (Jain et al. 2008, 2012). The objective of the treatments was to increase landscape heterogeneity but also integrate other objectives such as maintaining stream buffers and when appropriate maintain and enhance old-growth forests. Simultaneously, a goal of the treatments was to create fuel patterns as to modify fire growth and behavior if one was to occur and decrease crown fuel homogeneity (Finney 2001). As a result of the treatments, a much more heterogeneous landscape was created that will most likely be more resilient to disturbances if and when they may occur compared to the landscape prior to the treatment.

Characteristics that made dry forests disturbance resilient included low overstory densities, fire-resistant tree species, and continuous intervention of disturbance (fire, insect, and disease) returning at various intervals. The complex interaction between disturbance and weather created a series of regeneration phases over time that was serendipitous and varied over the landscape making them difficult to replicate. These disturbances and regeneration process tended to develop forests containing a fine scale mosaic that included tree groups, single trees, and openings distributed throughout the landscape. Recent guides have attempted to quantify this historical pattern for developing restoration management strategies (Moore and Covington 1999; Larson and Churchill 2012; Churchill et al. 2013). Target forest conditions in these guides primarily focus on creating historical structure

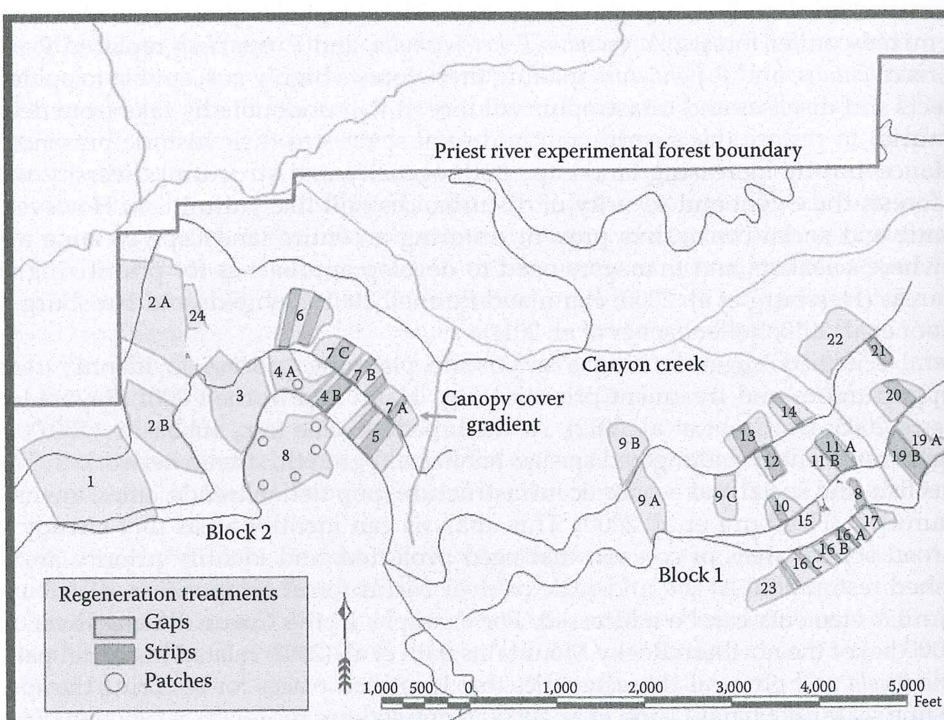

FIGURE 23.7
At Priest River Experimental Forest in northern Idaho, USA, researchers develop, implement, and evaluate silvicultural methods to introduce landscape heterogeneity by applying treatment mosaics (Jain et al. 2012). Within each treatment mosaic we leave stream corridors, canopy cover gradients alongside harvested units. Within harvested units when appropriate *Pinus monticola*, *Larix occidentalis*, or *Pinus ponderosa* pine remain to provide seed and contribute to variation in canopy cover within the strips (dark gray strips).

and composition. This specific pattern has been suggested as a plausible reference condition that made these dry forests resilient to fire and insects.

However, these characteristics alone do not provide for numerous contemporary values important to wildlife where fires were suppressed. This is exemplified by wildlife which may require late-seral vegetation, down wood, snags (of all sizes), and deep organic layers (fungi habitat) that are not integral to such rigid restoration prescriptions (e.g., Reynolds et al. 1992; Long and Smith 2000; Spies et al. 2006). Similarly, reintroducing fire may be preferred, but smoke, burn windows, risk, and liability may limit the use of prescribed or wildfire in restoration strategies. As such, most wildfires will be suppressed and particularly those which are located near homes, towns, and within municipal watersheds. Some wildfires and the environments they create will still occur but most likely not to the same extent, location, or frequency. More importantly, multiple land ownerships occur throughout these areas, probably with different objectives; limiting large landscape restoration activities (millions of hectares).

On the Boise Basin and Black Hills Experimental Forests, treatments are being evaluated that will produce wildfire and insect resilient forests while also providing timber and recreation opportunities (Jain et al. 2008, 2014). For example, on Boise Basin, an irregular silvicultural system is being assessed for its application in maintaining old forest structures. In this system, old *P. ponderosa* (>400 years old) were left and younger trees (approximately 180 years old) were retained based on their crown ratio (>40%) and diameter to height ratio (>80), both indicators of tree vigor. Dense patches of *P. menziesii* established after the last wildfire (Graham and Jain 2005). The objective was not to recreate a historical forest structure but to separate crowns and remove ladder fuels to diminish crown fire potential, and also to leave areas with interlocking crowns to provide habitat for the northern goshawk and its prey (Reynolds et al. 1992). To encourage grass, forb, and shrub regeneration, large openings were created to allow sufficient sunlight and growing space for their establishment. This increase in understory diversity would provide habitat for ground dwelling birds and mammals (Reynolds et al. 1992). Some thickets of *P. menziesii* were dispersed through the forest to provide hiding cover for wildlife, but some of them were located away from the large trees and on steep north facing aspects where they would thrive. Delayed tree mortality has increased snag abundance, which provides wildlife habitat and as they decay and fall, they will produce large woody debris. When opportunities arise, we will use prescribed fire, but we will also use mastication, mowing, cutting, or harvesting to maintain the desired forest floor conditions (Jain et al. 2012, 2014).

23.5.2 Increase within Stand Heterogeneity in Moist Forests

Site-specific restoration strategies should not be isolated to an individual stand but employed within the context of creating diverse and heterogeneous landscapes with a variety of stand compositions and structures. Silvicultural treatments that create within stand heterogeneity requires opening sizes large enough to favor shade-intolerant tree species, shrubs, forbs, and grasses and creating a variety of forest floor and mineral soil surfaces including blackened, mineral, and organic (Graham et al. 2004; Oliver and O'Hara 2004). Such treatments reflect the environments that were created by windstorms, root disease, ice, snow, and low intensity surface fires (Figures 23.3 and 23.4a). Again, these treatments did not necessarily emulate historical conditions but these treatments were initiated as to increase the abundance of early- and mid-seral species, produce wildlife habitat, and diversify the fuel matrix.

A variety of irregular uneven-aged and two-aged silvicultural systems can be developed that have application in both the dry and moist forests (Graham et al. 1983; Graham and Jain 2005; Graham et al. 2007; Churchill et al. 2013; Franklin et al. 2013). Even-aged (e.g., clearcut, seed tree, and shelterwood) silvicultural systems with varying amounts of reserve trees left in the cuttings create two-aged structures, and uneven-aged systems can introduce spatial and vertical heterogeneity. Even though uneven-aged selection systems are most often associated with the rigid reverse-j shaped diameter distributions, this is only one approach applicable for describing uneven-aged structures (Smith et al. 1997; Puettmann et al. 2008). These systems maintain multiaged (diameter distributions) forest structures by planning for and executing frequent (e.g., 10–20 years) entries (treatments). The heterogeneity introduced in these systems requires the timely establishment and tending of regeneration during each entry. Also, horizontal diversity in crowns is preferred to avoid crown fire but it is allowed for a diversity of postfire outcomes if a fire outbreaks. The majority of these systems differ from the top down "command and control" approach designed to produce timber as they allow for and encourage adaptive management (Drever 2006; Puettmann et al. 2008). A key to their success is maintaining the long-term (decades to centuries) vision while implementing short-term silvicultural methods (e.g., regenerating, weeding, cleaning, thinning, and releasing) aimed at producing and sustaining the desired forest structures and compositions.

At Priest River, Experimental Forest within stand heterogeneity was created using several silvicultural techniques (Jain et al. 2008). Because the area occurred on steep slopes, it was harvested using a cable (sky-line) system and cutting units of varying widths were created. Along the harvest edge, a canopy cover gradient was used to enhance the regeneration of both overstory and understory species (Figure 23.7). This variable density edge also increased the abundance of snags, down wood, and other structures that was favored by wildlife. We placed the largest openings (<5 ha) where *L. occidentalis*, *P. monticola*, and *P. ponderosa* were present to promote natural regeneration of these species. In all cases, early- and mid-seral species with good tree vigor were chosen as leave-trees. After harvest, a diversity of forest floor conditions (site preparation) was produced which ensured seed germination substrates that would favor a wide range of tree species. For example, in one stand, we masticated, prescribed burned, and grapple piled the postharvest slash and combined with the variation in canopy cover a plethora of vegetative establishment and development conditions were created. As such, highly heterogeneous forest floor vegetative and fuel complexes will be created.

In the dry forests, on the Black Hills Experimental Forest, we applied two different cleaning and weeding prescriptions to create highly variable ground-level vegetation complexes (Jain et al. 2014). The first approach used a 4 × 4 m tree spacing as a separate stratum ignoring the overstory trees. This method tends to produce ladder fuels and favors regularly spaced trees. The second approach was to clean the advanced regeneration to 4 × 4 m spacing but include overstory trees in the prescription. This technique favored maximum growing space for each crown class, eliminated ladder fuels from beneath the large trees, and produced horizontal separation between the overstory and understory.

23.5.3 Change Species Composition

In the moist forests, disturbances provided opportunities for early- and mid-seral species to regenerate, and reestablishing such species is central to restoring the moist forests (Everett et al. 1994; Hann et al. 1997; Harvey et al. 1999; Neuenschwander et al. 1999; Fins et al. 2001; Franklin et al. 2013). *P. monticola*, *L. occidentals*, and *P. ponderosa* are resilient to many endemic insects and diseases. *P. monticola* is an opportunistic species and can regenerate in a variety of openings. It, along with *L. occidentalis*, which is deciduous, compete well with shade-tolerant species and their crown architectures shed snow readily and diminish wind damage (Graham 1990; Schmidt and Shearer 1990; Jain et al. 2004, 2012). These species along with late-seral *T. plicata* are well-suited to feature in forest restoration strategies especially in face of climate change. In dry mixed-conifer forests, *P. ponderosa* and *L. occidentalis* have thick bark, especially as they age, which allows them to survive both prescribed and wild surface fires. These species, in both moist and dry forests, when combined with minor amounts of late-seral species (*A. grandis/A. concolor, P. menziesii*, and *T. heterophylla*) create greater species diversity than what is present today.

Early- and mid-seral conifer species were historically managed with even-aged silvicultural systems and in the moist forests, clearcutting was the dominant method used. For various reasons including cumulative watershed effects, wildlife habitat destruction, and just being unsightly clearcutting is being used minimally and systems that maintain high forest cover are being favored. To facilitate, the maintenance of high forest cover yet increase the presence and plurality of early and mid-seral species; Jain et al. (2004) identified four opening size thresholds that favor *P. monticola* establishment, competitive advantage over other species, in which it is free-to-grow, and conditions when optimum growth can be achieved. At Priest River Experimental, we used these thresholds to design the variety of opening sizes as we developed the treatment mosaic (Jain et al. 2002). In this study, the range of opening sizes were specifically designed to favor free-to-grow (0.5 hectare) to optimum growth (4.5 hectares) for *P. monticola*. However, in many places, seed sources did not exist and *P. monticola, L. occidentalis*, and *T. plicata* were regenerated artificially.

In the dry mixed-conifer forests, *P. ponderosa* is the preferred species on the driest sites (*P. ponderosa* and *P. menziesii* PVTs) with *Populus tremuloides* and *Picea* spp. occurring in wet areas or in cold pockets. On more productive mesic sites, *L. occidentalis* is also preferred. These forests can be quite topographically diverse and depending on the location and topographic diversity, inclusions of *T. plicata* PVT can occur in the draws that contained *P. monticola, P. contorta*, and *T. plicata* such as those located in northern Idaho and southern British Columbia (Daubenmire 1980) (Figure 23.8). This mosaic of pines, other conifers, and hardwood, all contributed to create disturbance resilient forests. Moreover, the

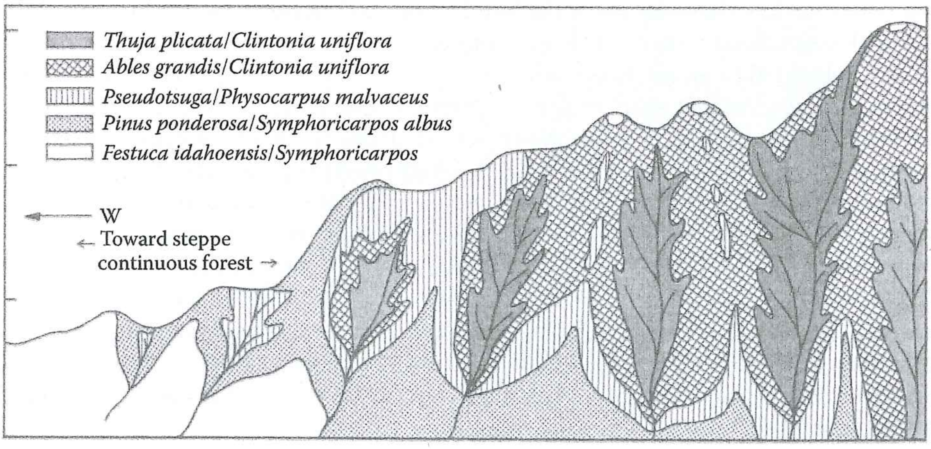

FIGURE 23.8
Topographic diverse south facing aspects have a diversity of vegetation. In draws, *Thuja plicata* dominates while on ridges *Pinus ponderosa* and *Pseudotsuga menziesii* dominate. Ecotones most often contain *Abies grandis*. *Armellaria* species and other root diseases often occur in these areas that create pockets that contain shrubs. This diversity in vegetation contributed to the mixed and variable fire regime common in moist mixed-conifer forests. (Reprinted with permission from Daubenmire, R. *Northwest Science* 54, 1980: 146–52.)

hardwood species are critical habitat for wildlife species (Reynolds et al. 2012). Fortunately, on dry forests, seed sources are plentiful but continuous disturbance is required to keep pine regeneration controlled and to provide opportunities for hardwoods to sprout and dominate. Thus restoration strategies in these forests need to control pine regeneration. On the Black Hills, both mechanical treatments and prescribed fire are being used to remove or kill *P. ponderosa* regeneration (Battaglia et al. 2008; Jain et al. 2012, 2014).

23.5.4 Integrate Disturbance into Restoration Strategies

Different from timber producing treatments, treatments designed to restore forests need to integrate disturbance into their management strategies, including introduced disturbances such as *C. ribicola* (a stem disease, blister rust) and its influence on *P. monticola* (Figure 23.6). Integrating introduced disease offer additional challenges, but a breeding program currently produces rust-resistant *P. monticola* seedlings for reforestation (68% of which exhibit some resistance to *Cronartium*; Fins et al. 2001) and continued breeding will ensure that rust mutations will not compromise the resistant material. Fortunately, *P. monticola* native populations do contain some natural resistance to the rust, thus in addition to breeding programs, mass selection presents an opportunity to utilize the rust-resistance occurring in natural stands (Hoff and McDonald 1980; Graham et al. 1994). In stands where blister rust has killed the majority (over 70%) of the *P. monticola*, approximately 7%–10% of the progeny, often thousands per ha, of the survivors exhibit rust-resistance (Hoff and McDonald 1980). Regenerating these wild populations can supplement the genetic diversity contained in breeding programs and provide prospects for restoring *P. monticola* when planting opportunities do not exist.

For example, on the Deception Creek Experimental Forest, we evaluated the amount of *C. ribicola* resistance occurring in naturally regenerated *P. monticola*. Using *P. monticola* planted in 1936 in the Forest, we collected cones from trees randomly, collected from trees exhibiting no rust, and from trees exhibiting no rust after all other *P. monticolas* were removed from the plantation. The last collection occurred after a full cone cycle to ensure that the cones which are collected reflected the reduced pollen cloud that would have fertilized the flowers. Seeds were extracted from each of these collections and seedlings were grown and subjected to an artificial inoculation by *C. ribicola* using *C. ribicola* infected *Ribes* spp. leaves. The *P. monticola* seedlings were planted in nursery beds and inspected biannually for rust infections during a four-year period. Trees from all collections were successfully exposed to blister rust as all trees exhibited needle spots indicating that *C. ribicola* spores had germinated on the tree needles. After four years, approximately 2% of the trees originating from the last collection were totally free from rust and approximately 1% was free from rust from the other collections. Natural regeneration of *P. monticola* can exceed over 10,000 seedlings ha^{-1}; if only 1% of seedlings survive, 100 trees ha^{-1} would remain and because this species is resilient to many endemic diseases, several species could survive to produce cones (Haig et al. 1941). These preliminary results show promise that through silvicultural treatments, *P. monticola* rust-resistance can be increased and utilized in restoration strategies.

23.5.5 Surface Vegetation and Forest Floor

Although trees are most frequently removed to modify fire behavior, the entire fuel matrix influences fires and, in particular, surface fuels (Rothermel 1983). The most effective strategy for reducing crown fire occurrence and severity is to (1) reduce surface and ladder fuels, (2) increase height to live crown, (3) reduce canopy bulk density, and (4) reduce continuity of the forest canopy, in that order of importance (Graham et al. 2004) (Figure 23.9a). More importantly, in dry mixed-conifer forests where soil surface organic layers were historically reduced by surface fires, fire suppression has allowed thick and deep layers of needle and bark slough to accumulate at the base of live trees. Burning these surface layers can result in smoldering combustion that can last for days. As such, these types of fires can girdle the tree by killing the tree cambium or kill fine roots that could cause delayed tree mortality (Hood 2010; Graham et al. 2012; Jain et al. 2012).

In dry forests, restoring conditions in which surface fires (whether prescribed or wild) can burn, requires approaches that conserve nutrients, microbial activity, and fine roots

(a)

(b)

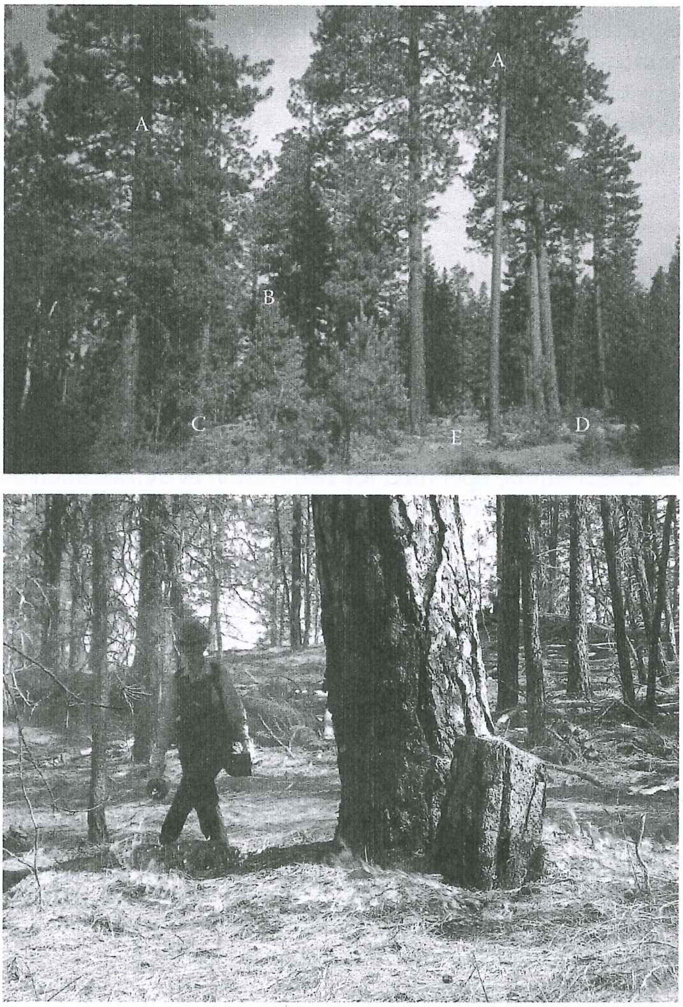

FIGURE 23.9

There are six layers of canopy fuels (a). They are canopy (A), shrubs and small trees (B), low, nonwoody vegetation (C), woody fuels (D), and ground fuels (E). Fire exclusion has caused deep organic layers to develop around old *Pinus ponderosa* boles. Applying prescribed fire (A) or raking when the lower organic layers are high in moisture (>100%), and soil temperatures are low (–2°C), the depth of these layers can be decreased with minimal damage to the fine roots they contain. (USDA Forest Service photographs.)

that often develop in the uncharacteristically deep surface organic layers (Jain et al. 2012). Gradually, decreasing the depth and volume of these organic layers by repeating treatments over a series of years will force the fine roots to migrate down into the mineral soil. Depending on the dry forest setting, it may take one to multiple combinations of mechanical and carefully executed prescribed burns before fuel loads, species composition, and forest structures allow prescribed fires to burn freely in the dry forests (Hood 2010; Jain et al. 2012). Burning, when moisture is high (>100%) in the lower layers, will preserve them, but it allows the drier upper layers to be consumed (Brown et al. 1985). Moreover, burning when soil temperatures are low (less than 2°C) and when fine root growth is minimal results in minimal root damage (Kramer and Kozlowski 1979). These conditions occur most often in early spring (Figure 23.9b). Nutrient volatilization is minimized by burning surface fuels when the lower layers have high amounts of moisture (>100% moisture by weight) (Hungerford et al. 1991). If fire is not an option for decreasing these layers, raking or removing them using a leaf blower is suggested; however, if fine roots are present in the organic layers, treatments need to be applied when soil temperatures are less than 2°C (Hood 2010; Jain et al. 2012).

23.6 Economic and Social Aspects

Ecological information is available to restore the moist and dry forests of the US Pacific Northwest (Reynolds et al. 1992; Covington and Moore 1994; Hann et al. 1997; Harvey et al. 1999; Nuenschwander et al. 1999; Long and Smith 2000; Finns et al. 2001). Interweaving the social, economic, and political needs of the society present the greater challenge in restoring these forests. Costs can be specified for restoration activities such as harvesting, thinning, planting, weeding/cleaning, prescribed burning, exotic plant control, riparian area treatments, and for planning, monitoring and analysis. Specific costs will vary depending on site characteristics and stand structure. For example, the cost per unit area ranges from $75 (U.S. dollars) per ha for vegetation management to $750 per ha depending on slope, material removed and intensity of surface treatments (Jain et al. 2012). However, current extent and severity of disturbances also have an economic trade-off when considering factors such as fire suppression and resource damage from extreme and large wildfires (Holms et al. 2008).

These cost estimates do not reflect some of the benefits of restoring moist and dry forests (De Groot et al. 2013). For example, an estimated 83% of the recreational benefits within the Interior Columbia Basin come from federally administered Forest Service and Bureau of Land Management lands (Phillips and Williams 1998; Reyna 1998). These recreation benefits include trail use, hunting, fishing, camping, boating, wildlife viewing, winter sports, day use, and motor viewing. When restoration positively influences these activities, the benefit may exceed the costs. Converting forests dominated by late-seral structures to forests dominated by early- and mid-seral structures most likely will benefit these recreational activities. Moreover, restored forest conditions may improve the habitat of legally threatened or endangered wildlife species, but all restoration treatments have to be implemented to avoid negative impacts on other protected species (such as, the bull trout, grizzly bear, and Canadian lynx). Nevertheless, a vocal segment of society prefers the status quo and resists actively nmanaging forests.

Active management including timber harvest and restoring forests near timber-dependent and isolated communities most likely would be a positive benefit. Reyna (1998) and Phillips and Williams (1998) reported that 137 communities specialized in logging and wood products manufacturing in the Interior Columbia River Basin, with 64 being isolated. Timber harvesting and wood products manufacturing generally has been important in these communities since the 1800s when many towns were established. However, timber harvesting is sometimes a controversial issue, but collaborative groups are working with public land managers to identify socially acceptable management techniques that will promote restoration opportunities (Cheng and Sturtevant 2012). Moreover, the infrastructure for manufacturing wood products has decreased in recent years, increasing transportation costs for the harvested material (Haynes 2002; Jain et al. 2012). Perhaps the greatest economic challenge is that the small trees (especially *P. ponderosa*) available for harvesting in most restoration activities have low value (Lippke 2002; McKetta 2002). However, small-diameter (15–25 cm in diameter) *A. grandis* and *P. menziesii* seem to be an exception, and are of value in producing construction materials (McKetta 2002). Although there is value in this material, out of the 5.7 million forested hectares in the Inland Northwest not in wilderness, national parks, or in other reserved status, it is estimated that there are only 800,000 hectares that have material of value that needs to be removed (Jain et al. 2012).

In general, private landowners have more flexibility for conducting restoration activities where timber production is often the primary objective (Blatner et al. 1994). Unlike their federal counterparts, managers of private lands have fewer requirements for analysis and planning prior to conducting activities, which allows them to respond quickly to insects, diseases, wildfires, and storms, as well as changing markets. Moreover, forests containing *P. ponderosa, P. menziesii, P. monticola,* and *L. occidentalis* tend to have high commercial value and are resilient to native disturbances (Nuenschwander et al. 1999; McKetta 2002). However, just species presence would not necessarily indicate that a forest is restored; the entire suite of forest structures (biological and physical properties of vegetation, soils, microbes, and water) and compositions distributed in a mosaic over the landscape would most likely constitute a restored forest (Nuenschwander et al. 1999).

23.7 Conclusions

A combination of harvesting and the introduction of *C. ribicola* greatly impacted the inland moist forests of the US Pacific Northwest. Fire exclusion played a role in changing these forests, but not to the same extent that it did in the dry forests. Because of current (recreation, scenic, and wildlife habitat) and past (harvesting and road construction) human uses and values, restoring both the dry and moist forests will be challenging, but it is not impossible. By viewing and developing management strategies using a multiscale approach, landscape plans and silvicultural systems can be designed that move these forests on a trajectory toward their desired restored compositions and structures.

Majestic stands of *P. monticola, P. ponderosa, L. occidentalis, T. plicata,* and all possible combinations of these species and their associates once populated the forests of the Inland Northwest. Once these systems reflect the historical composition and structure, endemic levels of other disturbances can aid in sustaining these forests into the future. However, because of human presence, the extent and intensity of endemic disturbances plus exotic introductions into these forests will make restoration activities challenging. If society determines that the dry and moist forests should be restored, it will take time, patience, perseverance, and commitment by both public and private individuals and organizations to accomplish the task.

References

Atkins, D., J. Byler, L. Livingston, P. Rogers, and B. Bennett. *Health of Idaho's Forests: A Summary of Conditions, Issues and Implications*. Missoula, MT: U.S. Department of Agriculture Forest Service, Northern Region, 1999.

Battaglia, M.A., F.W. Frederick, and W.D. Sheppard. Can prescribed fire be used to maintain fuel treatment effectiveness over time in Black Hills ponderosa pine forests? *Forest Ecology and Management* 256, 2008: 2029–2038.

Bentz, B., J. Logan, J. MacMahon et al., *Bark Beetle Outbreaks in Western North America: Causes and Consequences*. Salt Lake, UT: University of Utah Press, 2009.

Blatner, K.A., C.E. Keegan III, J. O'Laughlin, and D.L. Adams. Forest health management policy. *Journal of Sustainable Forestry* 2, 1994: 317–37.

Bliss, J.C. Public perceptions of clearcutting, *Journal of Forestry* 98, 2000: 4–9.

Bollenbacher, B.L., R.T. Graham, and K.M. Reynolds. Regional forest landscape restoration priorities: Integrating historical conditions and an uncertain future in the northern Rocky Mountains. *Journal of Forestry* 112, 2014: 474–83.

Brown, J.K., E.D. Reinhardt, and W.C. Fischer. *Predicting Duff and Woody Fuel Consumed by Prescribed Fire in the Northern Rocky Mountains*. INT-RP-337. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station, 1985.

Burton, P.J. and S.E. Macdonald. The restorative imperative: Assessing objectives, approaches, and challenges to restoring naturalness in forests. *Silva Fennica* , 45 (5), 2011: 843–63.

Camp, A., C. Oliver, P. Hessburg, and R. Everett. Predicting late-successional fire refugia pre-dating European settlement in the Wenatchee Mountains. *Forest Ecology and Management* 95, 1997: 63–77.

Caprio, A.C. and D.M. Graber. Returning fire to the mountains: Can we successfully restore the ecological role of pre-euro American fire regimes to the Sierra Nevada? In *Wilderness Science in a Time of Change Conference: Volume 5- Wilderness Ecosystems, Threats, and Management*, compiled by N.C. David, S.F. McCool, W.T. Borrie, and J. O'Loughlin, 233–41. RMRS-P-15-VOL-5. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, 2000.

Cheng, A.S. and V.E. Sturtevant. A framework for assessing collaborative capacity in community-based public forest management. *Environmental Management* 49, 2012: 675–89.

Churchill, D.J., A.J. Larson, M.C. Dahlgreen, J.F. Franklin, P.F. Hessburg, and J.A. Lutz. Restoring forest resilience: From reference spatial patterns to silvicultural prescriptions and monitoring. *Forest Ecology and Management* 291, 2013: 442–57.

Cooper, S.V., K.E. Neiman, and D.W. Roberts. *Forest Habitat Types of Northern Idaho: A Second Approximation*. INT-GTR-236. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station, 1991.

Covington, W.W. and M.M. Moore. Post settlement changes in natural fire regimes and forest structure. In *Assessing Forest Ecosystem Health in the Inland West*, edited by R.N. Sampson and D.L. Adams, 153–81. New York: Haworth Press, 1994.

Daubenmire, R. Mountain topography and vegetation patterns. *Northwest Science* 54, 1980: 146–52.

Daubenmire, R. and J. Daubenmire. *Forest Vegetation of Eastern Washington and Northern Idaho*. Bull. 60. Pullman, WA: Washington State Agricultural Experiment Station, 1968.

Debano L.F. Effects of fire on soil properties. In *Proceedings: Management and Productivity of Western-Montane Forest Soils*, compiled by A.E. Harvey and F.L. Neuenschwander, 151–6. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station, 1991.

De Groot, R.S., J. Blignaut, S. Van Der Ploeg, J. Aronson, T. Elmqvist, and J. Farley. Benefits of investing in ecosystem restoration. *Conservation Biology* 27, 2013: 1286–1293.

DellaSala, D.A., A. Martin, and R. Spivak et al. A citizen's call for ecological forest restoration: Forest restoration principles and criteria. *Ecological Restoration* 21, 2003: 14–23.

Drever, C.R., G. Peterson, C. Messier, Y. Bergeron, and M. Flannigan, 2006. Can forest management based on natural disturbances maintain ecological resilience? *Canadian Journal of Forest Research* 36, 2006: 2285–99.

Everett, R.L., P. Hessburg, M. Jensen, and B. Bormann. *Eastside Forest Ecosystem Health Assessment: Volume I, Executive Summary*. PNW-GTR-317. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 1994.

Finney, M.A. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. *Forest Science* 47, 2001: 219–28.

Fins, L., J. Byler, D. Ferguson et al. *Return of the Giants: Restoring White Pine Ecosystems by Breeding and Aggressive Planting of Blister Rust-Resistant White Pines*. Moscow, ID: University of Idaho, 2001.

Foiles, M.W., R.T. Graham, and D.F. Olson Jr. *Abies grandis* (Dougl. ex D. Don.) Lindl. In *Silvics of North America, Volume 1, Conifers*, technical coordinators R.M. Burns and B.H. Honkala, 52–9. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service, 1990.

Franklin, J.F. and C.T. Dryness. *Natural Vegetation of Oregon and Washington*. Corvallis, OR: Oregon State University Press, 1973.

Franklin, J.F. and K.N. Johnson. 2012. A restoration framework for federal forests in the Pacific Northwest. *Journal of Forestry* 110, 2012: 429–39.

Franklin, J.F., K.N. Johnson, D.J. Churchill, K. Hagmann, D. Johnson, and J. Johnston. 2013. *Restoration of Dry Forests in Eastern Oregon: A Field Guide*. Portland, OR: The Nature Conservancy.

Graham, R., M. Finney, C. McHugh et al., *Fourmile Canyon Fire Findings*. RMRS-GTR-289. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, 2012.

Graham, R.T., *Pinus monticola* (Dougl. Ex. D. Don.). In *Silvics of North America, Volume 1, Conifers*, technical coordinators R.M. Burns and B.H. Honkala, 385–94. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service, 1990.

Graham, R.T., C.A. Wellner, and R. Ward. Mixed conifers, western white pine, and western redcedar. In *Silvicultural Systems for the Major Forest Types of the United States*, compiled by Burns, R.M., 67–9. Agric. Handb. 445. Washington, DC: U.S. Department of Agriculture Forest Service, 1983.

Graham, R.T., J.R. Tonn, and T.B. Jain. Managing western white pine plantations for multiple resource objectives. In *Proceedings—Interior Cedar-Hemlock-White Pine Forests: Ecology and Management*, edited by D.M. Baumgartner, J.E. Lotan, and J.R. Tonn, 269–75. Pullman WA: Washington State University, 1994.

Graham, R.T., S. McCaffrey, and T.B. Jain, tech. Eds. *Science Basis for Changing Forest Structure to Modify Wildfire Behavior and Severity*. RMRS-GTR-120. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, 2004.

Graham, R.T. and T.B. Jain. Application of free selection in mixed forests of the inland northwestern United States. *Forest Ecology and Management* 209, 2005: 131–45.

Graham, R.T., T.B. Jain, and J. Sandquist. Free selection: A silvicultural option. In *Restoring Fire-Adapted Forested Ecosystems. 2005 National Silviculture Workshop*, edited by R. Powers. PSW-GTR-203. Albany, CA: U.S. Department of Agriculture Forest Service, Pacific Southwest Research Station, 2007.

Haig, I.T., K.P. Davis, and R.H. Weidman. *Natural Regeneration in the Western White Pine Type*, Tech. Bull. No. 767. Washington, DC: U.S. Department of Agriculture Forest Service, 1941.

Hann W.J. and D.L. Bunnell. Fire and land management planning and implementation across multiple scales. *International Journal of Wildland Fire*. 10, 2001: 389–403.

Hann, W.J., J.L. Jones, M.G. Karl et al. Landscape dynamics of the basin. In *An Assessment of Ecosystem Components in the Interior Columbia Basin and Portions of the Klamath and Great Basins: Volume II*, edited by T.M. Quigley and S.J. Arbelbide, 338–1055. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 1997.

Harvey, A.E., M.F. Jurgensen, M.J. Larsen, and J.A. Schlieter. *Distribution of Active Ectomycorrhizal Short Roots in Forest Soils of the Inland Northwest: Effects of Site and Disturbance*. INT-RP-374. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station, 1986.

Harvey, A.E., M.F. Jurgensen, M.J. Larsen, and R.T. Graham. *Decaying Organic Materials and Soil Quality in the Inland Northwest: A Management Opportunity*. INT-GTR-225. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station, 1987.

Harvey, A.E., P.F. Hessburg, J.W. Byler, G.I. McDonald, J.C. Weatherby, and B.E. Wickman. "Health declines in interior forests: Symptoms and solutions. In *Ecosystem Management in Western Interior Forests*, edited by D.M. Baumgartner and R.L. Everett, 163–70. Pullman, WA: Washington State University, 1994.

Harvey, A.E., R.T. Graham, and G.M. McDonald. Tree species composition change: Forest soil organism interaction potential effects on nutrient cycling and conservation processes in interior forests. In *Pacific Northwest Forest and Rangeland Soil Organism Symposium*, edited by R. Meurisse, W.G. Ypsllantis and C. Seybold, 137–45. PNW-GTR-46.1 Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 1999.

Haynes, R.W. U.S. timber supply and demand in the United States, 1996 to 2050. In *Small-Diameter Timber: Resource Management, Manufacturing, and Markets, Symposium Proceedings*, compiled by D.M. Baumgartner, L.R. Johnson and E.J. Depuit, 33–6. Pullman, WA: Washington State University, 2002.

Hermann, R.K. and D.P. Lavender. *Pseudotsuga menziesii* (Mirb.) Franco. In *Silvics of North America, Volume 1, Conifers*, technical coordinators R.M. Burns and B.H. Honkala, 527–40. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service, 1990.

Hessburg, P.F., J.K. Agee, and J.F. Franklin. Dry forests and wildland fires of the Inland Northwest USA: Contrasting the landscape ecology of the pre-settlement and modern eras. *Forest Ecology and Management* 211, 2001: 117–39.

Hessburg, P.F., R.G. Mitchell, and G.M. Filip. *Historical and Current Roles of Insects and Pathogens in Eastern Oregon and Washington Forested Landscapes*. PNW-GTR-327. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 1994.

Hessburg, P.F., B.R. Salter, and K.M. James. Re-examining fire severity relations in pre-management era mixed conifer forests: Inferences from landscape patterns of forest structure. *Landscape Ecology* 22, 2007: 5–24.

Hessburg, P.F., R.B. Salter, R.B. Richman, and B.G. Smith. Ecological subregions of the Interior Columbia Basin, US. *Applied Vegetation Science* 3, 2000: 163–80.

Hoff, R.J. and G.I. McDonald. *Improving Rust-Resistant Strains of Inland Western White Pine*. INT-RP-245. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Forest and Range Experiment Station, 1980.

Holmes, T.P., Prestemon J.P., and K.L. Abt. (eds.). Chapter 1: An introduction to the economics of forest disturbance. In *The Economics of Forest Disturbances: Wildfires, Storms, and Invasive Species*. 3–14, Dordrecht: Springer, 2008.

Hood, S.M. *Mitigating Old Tree Mortality in Long-Unburned, Fire-Dependent Forests: A Synthesis*. RMRS-GTR-238. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, 2010.

Hungerford, R.D., M.G. Harrington, W.H. Frandsen, K.C. Ryan, and G.J. Niehoff. Influence of fire on factors that affect site productivity. In *Proceedings—Management and productivity of western-montane forest soils*, compiled by A.E. Harvey and F.L. Neuenschwander, 32–50. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station, 1991.

Jain, T.B., M.A. Battaglia, and R.T. Graham. Northern Rocky Mountain experimental forests: settings for science, management, and education alliances. *Journal of Forestry* 112, 2014: 534–41.

Jain, T.B., M.A. Battaglia, H-S. Han et al. *A Comprehensive Guide to Fuel Management Practices for Dry Mixed Conifer Forests in the Northwestern United States*. RMRS-GTR-292. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, 2012.

Jain, T.B., R.T. Graham, and P. Morgan. Western white pine development in relation to biophysical characteristics across different spatial scales in the Coeur D'Alene River Basin in northern Idaho, U.S.A. *Canadian Journal of Forest Research* 32, 2002: 1109–25.

Jain, T.B., R.T. Graham, and P. Morgan. Western white pine growth relative to forest openings. *Canadian Journal of Forest Research* 34, 2004: 2187–98.

Jain, T.B., R.T. Graham, J. Sandquist et al. Restoration of Northern Rocky Mountain moist forests: Integrating fuel treatments from the site to the landscape. In *Integrated Restoration Efforts for Harvested Forest Ecosystems*, edited by R. Deal, 147–172. PNW-GTR-733. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 2008.

Keane, R.E., P.F. Hessburg, P.B. Landres, and F.J. Swanson. The use of historical range of variability (HRV) in the landscape management. *Forest Ecology and Management* 258, 2009: 1025–37.

Ketcham D.A., C.A. Wellner, and S.S. Evans Jr. Western white pine management programs realigned on Northern Rocky Mountain National Forests. *Journal of Forestry* 66, 1968: 329–32.

Kramer, P.J. and T.T. Kozlowski. *Physiology of Woody Plants*. New York: Academic Press, 1979.

Larson, A.J. and D. Churchill. Three spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. *Forest Ecology and Management* 267, 2012: 74–92.

Larsen, M.J., A.E. Harvey, and M.F. Jurgensen. Residue decay processes and associated environmental functions in northern Rocky Mountain forests. In *Environmental Consequences of Timber Harvesting in Rocky Mountain Coniferous Forests: Symposium Proceedings*, 157–74. INT-GTR-90. Ogden, UT: U.S. Department of Agriculture Forest Service, 1980.

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

Lillybridge, T.R., B.L. Kovalchik, C.K. Williams, and B.G. Smith. *Field Guide for Forested Plant Associations of the Wenatchee National Forest*, PNW-GTR-359. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 1995.

Lippke, B. Technology transfer opportunities for small diameter timber problems. In *Small-Diameter Timber: Resource Management, Manufacturing, and Markets, Symposium Proceedings*, compiled by D.M. Baumgartner, L.R. Johnson, and E.J. Depuit, 53–56. Pullman, WA: Washington State University, 2002.

Littell, J.S., D. McKenzie, D.L. Peterson, and A.L. Westerling. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. *Ecological Applications* 19, 2009: 1003–21.

Long, J.N. and F.W. Smith. Restructuring the forest: Goshawks and restoration of southwestern ponderosa pine. *Journal of Forestry* 98, 2000: 25–30.

Marschner, G.E. and H. Marschner. Nutrient and water uptake by roots of forest trees. *Pflanzenernahr Bodenk* 159, 1996: 11–21.

Marshall, R. The life history of some western white pine stands on the Kaniksu National Forest. *Northwest Science* 2, 1928: 48–53.

McKetta, C. Why grow large trees anyway? A timber grower's perspective. In *Small-Diameter Timber: Resource Management, Manufacturing, and Markets, Symposium Proceedings*, compiled by D.M. Baumgartner, L.R. Johnson, and E.J. Depuit, 133–8. Pullman, WA: Washington State University, 2002.

Mercer, D.E. Chapter 6: Policies for encouraging forest restoration. In *Restoration of Boreal and Temperate Forests*, edited by J.A. Stanturf and P. Madsen, 97–110. Boca Raton, FL: CRC Press, 2005.

Meyer, W.H. *Yield of Even-Aged Stands of Ponderosa Pine*. Tech. Bull. 630. Washington, DC: U.S. Department of Agriculture Forest Service, 1938.

Minore, D. *Comparative Autecological Characteristics of Northwestern Tree Species-A Literature Review*. PNW-GTR-87. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Forest and Range Experiment Station, 1979.

Moore, M.M., W.W. Covington, and P.Z. Fule. Reference conditions and ecological restoration: A southwestern ponderosa pine perspective. *Ecological Applications* 9, 1999: 1266–77.

Moritz, M.A., P.F. Hessburg, and N.A. Povak. Native fire regimes and landscape resilience. In *The Landscape Ecology of Fire*, edited by D. McKenzie, C. Miller and D.A. Falk, 51–86. New York: Springer, 2011.

Neuenschwander, L.F., J.W. Byler, A.E. Harvey et al. *White Pine in the American West: A Vanishing Species—Can We Save It?* RMRS-GTR-35. Ogden, UT: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, 1999.

Oliver C.D. and K.L. O'Hara. Effects of restoration at the stand level. In *Restoration of Boreal and Temperate Forests*, 31–60. Boca Raton, FL: CRC Press, 2004.

Oliver C.D. and B.C. Larson. *Forest Stand Dynamics*. New York: John Wiley and Sons, 1990.

Oliver, W.W. and R.A. Ryker, *Pinus ponderosa* (Dougl. ex Laws). In *Silvics of North America, Volume 1, Conifers*, technical coordinators R.M. Burns and B.H. Honkala, 413–24. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service, 1990.

Packee, E.C. *Tsuga heterophylla* (Raf.) Sarg. In *Silvics of North America, Volume 1, Conifers*, technical coordinators R.M. Burns and B.H. Honkala, 613–22. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service, 1990.

Perry, D.A., P.F. Hessburg, C.N. Skinner et al. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. *Forest Ecology and Management* 262, 2011: 703–17.

Pfister, R.D., B.L. Kovalchik, S.F. Arno, and R.C. Presby. *Forest Habitat Types of Montana*. INT-GTR-34. Ogden, UT: U.S. Department of Agriculture Forest Service, Intermountain Research Station, 1977.

Phillips, R.H. and G.W. Williams. An estimation of effects of the draft EIS alternatives on communities. In *Economic and Social Conditions of Communities: Economic and Social Characteristics of Interior Columbia Basin Communities and an Estimation of Effects on Communities from the Alternatives of the Eastside and Upper Columbia River Basin Draft Environmental Impact Statements*, 83–116. Portland, OR: U.S. Department of Agriculture Forest Service, U.S. Department of Interior Bureau of Land Management, 1998.

Puettmann, K.J., K.D. Coates, and C. Messier. *A Critique of Silviculture: Managing for Complexity*. Washington, DC: Island Press, 2008.

Pyne, S.J. *America's Fires: A Historical Context for Policy and Practice*, Revised edition. Durham, NC: Forest History Society, 2010.

Quigley, T., R.W. Haynes, and R.T. Graham. *Integrated Scientific Assessment for Ecosystem Management in the Interior Columbia Basin*, PNW-GTR-382. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 1996.

Rehfeldt, G.E., R.J. Hoff, and R.J. Steinhoff. Geographic patterns of genetic variation in *Pinus monticola*. *Botanical Gaz*ette 145, 1984: 229–39.

Reyna, N.E. Economic and social characteristics of communities in the Interior Columbia Basin. In *Economic and Social Conditions of Communities: Economic and Social Characteristics of Interior Columbia Basin Communities and an Estimation of Effects on Communities from the Alternatives of the Eastside and Upper Columbia River Basin Draft Environmental Impact Statements*, 3–82. Portland, OR: U.S. Department of Agriculture Forest Service, U.S. Department of Interior Bureau of Land Management, 1998.

Reynolds, R.T., D.A. Boyce Jr., and R.T. Graham. Ponderosa pine forest structure and Northern Goshawk reproduction: Response to Beier et al. (2008). *Wildlife Society Bulletin* 36, 2012: 147–52.

Reynolds, K.M. and P.F. Hessburg, 2005. Decision support for integrated landscape evaluation and restoration planning. *Forest Ecology and Management* 207, 2005: 263–78.

Reynolds, R.T., R.T. Graham, and R.M. Hildegard. *Management Recommendations for the Northern Goshawk in the Southwestern United States*. RM-217. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Forest and Range Experiment Station and Southwestern Region, 1992.

Rippy, R.C., J.E. Stewart, P.J. Zambino et al. *Root Diseases in Coniferous Forests of the Inland West: Potential Implications of Fuels Treatments*. RMRS-GTR-141. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, 2005.

Robichaud, P.R., J.L. Byers, and D.G. Neary. *Evaluating the Effectiveness of Postfire Rehabilitation Treatments*. RMRS-GTR-63. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, 2000.

Rose, S.L., D.A. Perry, D. Pilz, and M.M. Schoeneberger. Allelopathic effects of litter on the growth and colonization of mycorrhizal fungi. *Journal of Chemical Ecology* 9, 1983: 1153–62.

Rothermel, R.C. *How to Predict the Spread and Intensity of Forest and Range Fires*. INT-GTR-143. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, 1983.

Ryan, K.C. and G.D. Amman. Bark beetle activity and delayed tree mortality in the Greater Yellowstone area following the 1988 fires. In *Ecological Implications of Fire in Greater Yellowstone USA*, edited by J.M. Greenlee, 151–8. Fairfield, WA: International Association of Wildland Fire, 1996.

Schmidt, W.C. and R.C. Shearer. *Larix occidentalis* (Nutt.). In *Silvics of North America*, Volume 1, Conifers, technical coordinators R.M. Burns and B.H. Honkala, 160–72. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture Forest Service, 1990.

Smith, D.M., B.C. Larson, M.J. Kelty, and P.M.S. Ashton. *The Practice of Silviculture: Applied Forest Ecology*. New York: John Wiley & Sons, 1997.

Spies, T.A., M.A. Hemstrom, A. Youngblood, and S. Hummel. Conserving old-growth forest diversity in disturbance-prone landscapes. *Conservation Biology* 20, 2006: 351–62.

Stanturf, J.A. What is forest restoration? In *Restoration of Boreal and Temperate Forests*, 3–11. Boca Raton, FL: CRC Press, 2004.

Stephens, S.L., N. Burrows, A. Buyantuyev et al. Temperate and boreal forest mega-fires: Characteristics and challenges. *Frontiers in Ecology and the Environment* 12, 2014: 115–22.

Stine, P.A., P. Hessburg, T. Spies et al. *The Ecology and Management of Moist Mixed-Conifer Forests in Eastern Oregon and Washington: A Synthesis of the Relevant Biophysical Science and Implications for Future Land Management*. PNW-GTR-897. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, 2014.

Van Wagner, C.E. Conditions for the start and spread of crown fire. *Canadian Journal of Forest Research* 7, 1977: 23–34.