chapter thirty

Restoring dry and moist forests of the inland northwestern U.S.

Theresa B. Jain and Russell T. Graham
US Department of Agriculture Forest Service

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30.1 Introduction

The complex topography of the inland northwestern U.S. (58.4 million ha) interacts with continental and maritime air masses to create a highly variable climate, which results in a variety of forest settings. Historically (1850 to 1900), approximately 20% of the area was covered by dry forests (*Pinus ponderosa, Pseudotsuga menziesii*), and an estimated 18% was covered by moist forests (*Pinus monticola, Tsuga heterophylla*). Frequent surface fires burned over 75% of the area of dry forests; today, these fires burn approximately 45% of the area. In the dry forests, successful fire exclusion and harvesting allowed dense stands of *Abies grandis, Pseudotsuga menziesii*, and small *Pinus ponderosa* to develop. Historically, forest canopies (*Pinus ponderosa*, *Larix occidentalis*) 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 *Pinus 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 *Abies grandis*, *Tsuga heterophylla*, and *Pseudotsuga menziesii* has occurred, encouraged by the harvesting of *L. occidentalis* and *Pinus ponderosa*. These changes have elevated the risk to large-scale insect and disease epidemics and uncharacteristic wildfires. Successful restoration strategies in both the dry and moist forests should be cognizant of the changes that have occurred not only in the tree component but also those occurring in the soil and across landscapes. The reintroduction of fire alone is not the answer to restoring these forests. Fire should only be used when the trees and soil are in harmony with its reintroduction. Given 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, a multiscale approach applied 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 Northwest.

30.2 Forests of the inland northwest

The inland northwestern U.S. (58.4 million ha) is defined by the Bitterroot, Selkirk, Cabinet, Salmon River, Lemhi, Steens, Purcell, Cascade, and Blue Mountain ranges with elevations over 1,500 m (Figure 30.1). 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

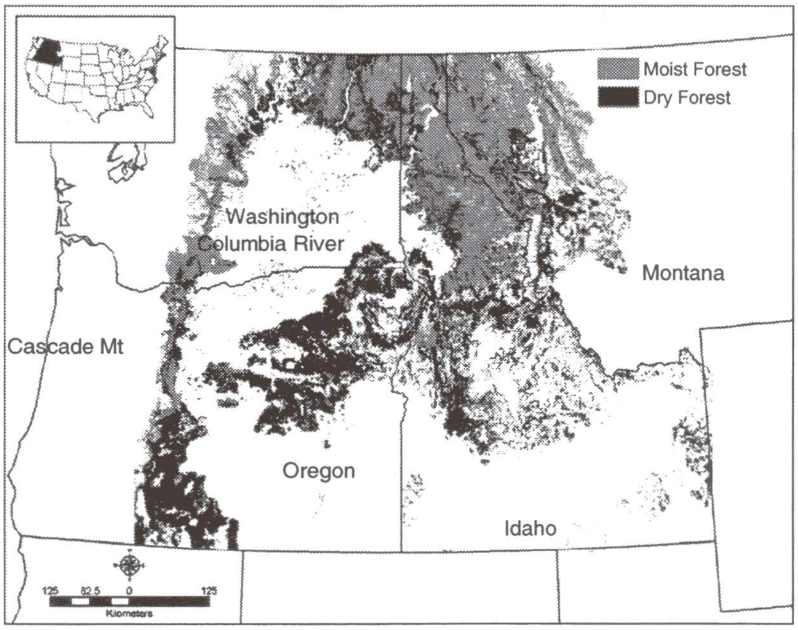


Figure 30.1 There are 58,361,400 ha in the inland northwestern U.S. framed by the Columbia River Basin. The topography is rugged ranging from the Cascade Mountains in the West to the Bitterroot and Salmon River Mountains in Idaho. Elevations in the region range from 225 m to over 3000 m. Dry and moist forests make up 90% of the forests occurring in the inland northwestern U.S. 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 (Hann et al. 1997).

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 7,500 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 Northwest, 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 compositionally and structurally diverse forests (Franklin and Dyrness 1973; Foiles et al. 1990; Graham 1990; Hann et al. 1997).

Until 1900, forests covered over 47% of the Inland Northwest (Figure 30.1). Dry forests occupied an estimated 11 million ha, dominated by *Pinus ponderosa*, and moist forests covered 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).

30.3 *Dry forests*

Dry forests occurred across a wide range of elevations in northeastern Washington, northeastern Oregon, central and southern Idaho, and south-central Oregon (Figure 30.1) (Hann et al. 1997). Soil parent materials include granites, metasedimentaries, glacial tills, and basalts. Vegetation in these forests is usually limited by water availability and is often subject to drought. Nutrient deficiencies develop in eroded areas that can limit forest development. *Pseudotsuga menziesii*, *Pinus ponderosa*, and dry *Abies 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. They are defined by and named for indicator species that grow in similar environmental conditions (Hann et al. 1997). When *L. occidentalis* is present in dry forests, it is always an early-successional species (dominant after disturbance). *Abies grandis*/*Abies concolor* or *Pseudotsuga menziesii* are late-successional species that are usually more shade-tolerant than the early-seral species they succeed, while *Pinus ponderosa* can play both roles, depending on the PVT (Daubenmire and Daubenmire 1968; Hann et al. 1997). 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).

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). In concert, these disturbances historically maintained a variety of structural and successional stages (Table 30.1). Approximately 18% of the area was in a grass, forb, and shrub stage for long (hundreds of years) periods and 15% contained

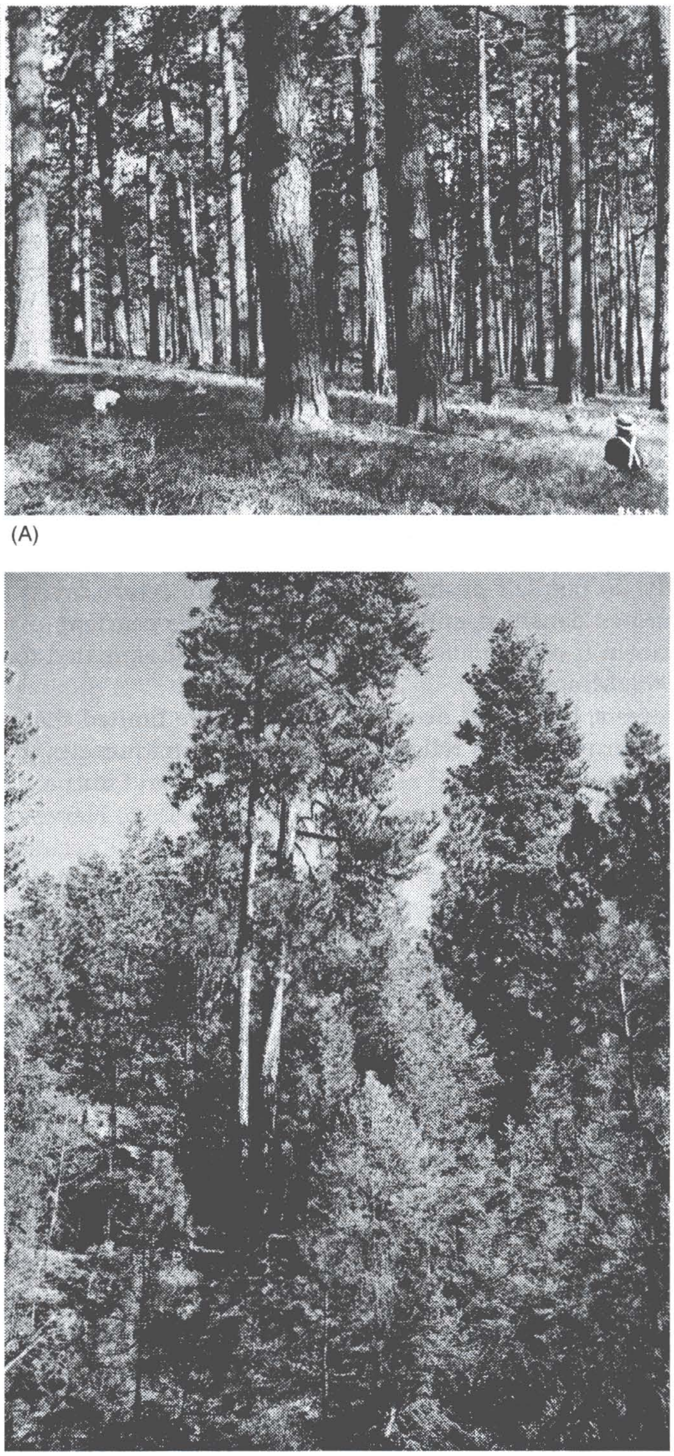
Table 30.1 Historical (1850 to 1900) and Current (1991) Distribution of Forest Structures Within the Dry and Moist Forests of the Inland Northwest (Hann et al. 1997)

Forest Structure	Historical (%)	Current (%)	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

early-seral *Pinus 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 stands dominated by late-seral *Pseudotsuga menziesii* and *Abies grandis/Abies concolor* with multiple canopies. Large, widely spaced (~250 trees per ha) *Pinus ponderosa* often dominated 21% of the dry forests, with the plurality of diameters ranging from 30 to 60 cm (Figure 30.2A) (Daubenmire and Daubenmire 1968; Hann et al. 1997). Late-seral single-storied forests containing *Pseudotsuga menziesii* and *Abies grandis/Abies concolor* complexes dominated some settings.

30.3.1 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 (Hann et al. 1997). The area burned by surface fires has decreased from an estimated 80% to less than 50% of the area. The mean fire return interval has also increased from less than 20 years to 40 to 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 to 57% of the area), often containing dense stands of small *Pinus ponderosa*, *Pseudotsuga menziesii*, or *Abies grandis/Abies concolor* (Table 30.1; Figure 30.2B). The proportion of the dry forests occupied by late-seral single-storied *Pinus ponderosa* has declined from 21 to 5% (Figure 30.2A). In addition,



(A)

(B)

Figure 30.2 A historical (1850 to 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) (U.S. Forest Service photos).

small-diameter trees have encroached and now occupy all but 1% of the dry forests that were formerly covered by grasses, forbs, and shrubs (Figure 30.2B). The dominant tree species has changed from *Pinus ponderosa* to *Pseudotsuga menziesii* or *Abies grandis / Abies concolor*, changing the character and canopy architecture of the forest.

The shift in species composition from *Pinus 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 *Pinus ponderosa* and *L. occidentalis* create brown rotten wood, which can persist in soil for centuries and has been shown to retain nutrients and hold water (Larsen et al. 1980; Harvey et al. 1987). *L. occidentalis* and *Pinus 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). *Pinus ponderosa* and *L. occidentalis* 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) *Pseudotsuga menziesii* and *Abies grandis / Abies 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 occur in *L. occidentalis*- and *Pinus ponderosa*-dominated forests (Figure 30.2B) (Harvey et al. 1999; Minore 1979).

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 were 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).

Harvesting the *L. occidentalis* and *Pinus ponderosa* and the ingrowth of *Abies grandis/Abies concolor* and *Pseudotsuga 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). 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). 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 stands of late-seral species are more demanding of nutrients and water than the historical stands dominated by widely- spaced early-seral species (Minore 1979; Harvey et al. 1999).

30.4 *Moist forests*

Moist forests of the inland northwestern U.S. 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 30.1). They grow at elevations ranging from 460 to 1,600 m and occasionally occur at elevations up to 1,800 m (Foiles et al. 1990; Graham 1990; Packee 1990; Schmidt and Shearer 1990; Hann et al. 1997) (Figure 30.1). 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 2,300 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 January through 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.

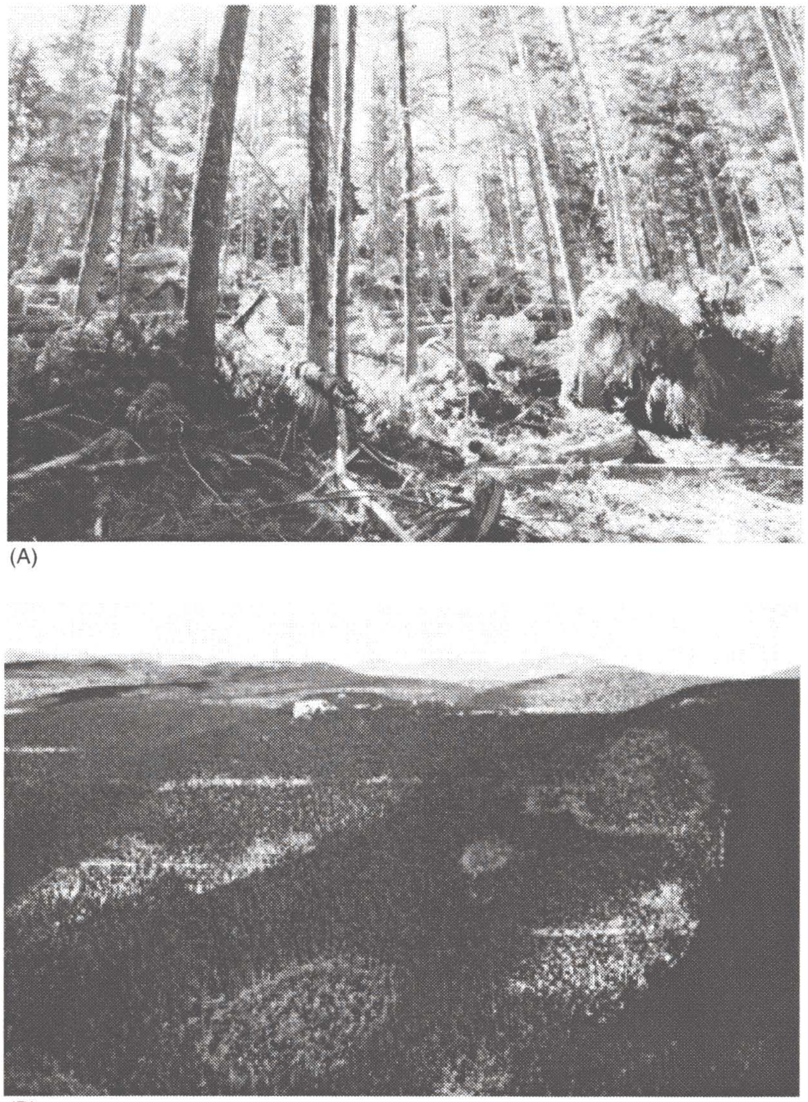
For both locations, the historical vegetation complexes ranged from early- to late-seral, and occurred within a landscape mosaic possessing all possible combinations of species and seral stages. The PVTs in the northern Rocky Mountains include *Thuja plicata, Tsuga heterophylla*, and *Abies grandis* with *Pinus monticola, L. occidentalis, Pinus contorta, Pseudotsuga menziesii*, and *Pinus ponderosa* as the early- and mid-seral species (Daubenmire and Daubenmire 1968; Hann et al. 1997). The eastern Cascades PVTs include *Thuja plicata, Tsuga heterophylla, Abies grandis, Abies amabilis*, and *Abies procera*. The early- and mid-seral species include *Pinus contorta, Pseudotsuga menziesii*, and *Pinus ponderosa* while *Pinus 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 horridum, 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).

Native disturbances (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 30.3A). Native insects (*Dendroctonus* spp.) and diseases (*Armillaria* spp., *Arceuthobium* spp.) infected and killed the very old or stressed individuals, which tended to diversify vegetation communities (Figure 30.3B) (Hessburg et al. 1994). 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 to 25 years) in a quarter of the area (Figure 30.4A). Lethal crown fires burned about a quarter of the area at intervals of 20 to 150 years but occasionally extended to 300 years (Figure 30.4B). The mixed-fire regime occurred across the rest of the moist forests at 20- to 150-year intervals. Fires typically started burning in July and were usually out by early September (Hann et al. 1997).

30.4.1 Moist forest change

The current distribution of successional-stage forest structure, species composition, and disturbance regimes differs from the historic (1850 to 1900) patterns of the moist forest (Hann et al. 1997). In some settings, the mixed-fire regime maintained closed canopy conditions,



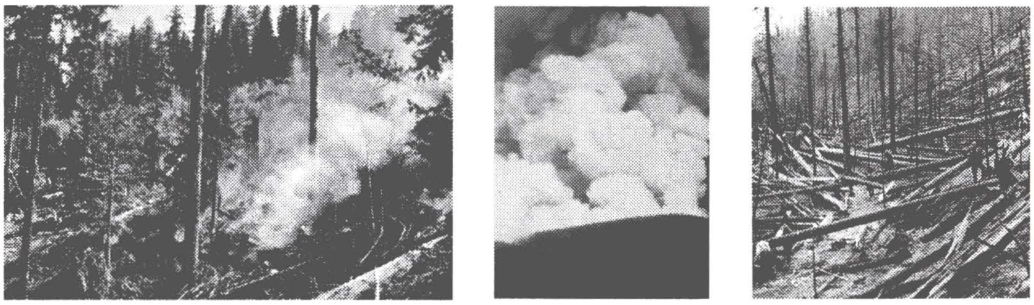
(A)

(B)

Figure 30.3 Ice and snow damage create small gaps and openings, decrease forest densities, and alter species composition (A). Historically (1850 to 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 (U.S. Forest Service photos).

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 30.1). The early-seral single-story stands that once occupied an estimated 25 to 30% of the area now occupy only 9 to 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; Nuenschwander et al. 1999; Fins et al. 2001); before 1900, *Pinus monticola* (early- to



(A) (B) (C)

Figure 30.4 Historically (1850 to 1900), in the moist forests, nonlethal surface fires (A) occurred at relatively frequent intervals (15 to 25 years) in 25% of the moist forests while lethal crown fires (B and C) occurred over 25% of the forests at 300-year intervals (U.S. Forest Service photos).

mid-seral species) dominated, often representing 15 to 80% of the trees within stands (Figure 30.5) (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 3 years, which allows it to regenerate abundantly after disturbance (Haig et al. 1941; Graham 1990). It is broadly adapted genetically (generalists) 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 2001). *Pinus monticola* often reaches 30 m in height or greater within 50 years of establishment (Graham 1990). *L. occidentalis* and *Pinus ponderosa* also occurred in the early- and mid-seral structures, but declined along with *Pinus monticola* and were succeeded by *Abies grandis, Pseudotsuga menziesii,* and *Tsuga heterophylla* (Hann et al. 1997; Atkins et al. 1999). The eastern Cascades had limited amounts of *Pinus monticola* and *L. occidentalis;* therefore, *Pinus ponderosa, Pinus contorta,* and *Pseudotsuga 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 *Abies grandis* PVT, fire maintained landscapes that contained a plurality of early-seral *Pinus 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 *Abies grandis* and *Pseudotsuga 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 U.S., 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, *Cronartium ribicola,* caused the greatest change (Figure 30.6). The rust infects all five-needle pines, and subsequently decimated the abundant *Pinus monticola.* Because the rust killed so many trees, the majority of surviving pines were harvested under the assumption that they too would succumb to the rust (Ketcham et al. 1968). *Abies grandis* and *Pseudotsuga menziesii* readily filled the niche that *Pinus monticola* once held. In the eastern Cascades, blister rust was less severe since

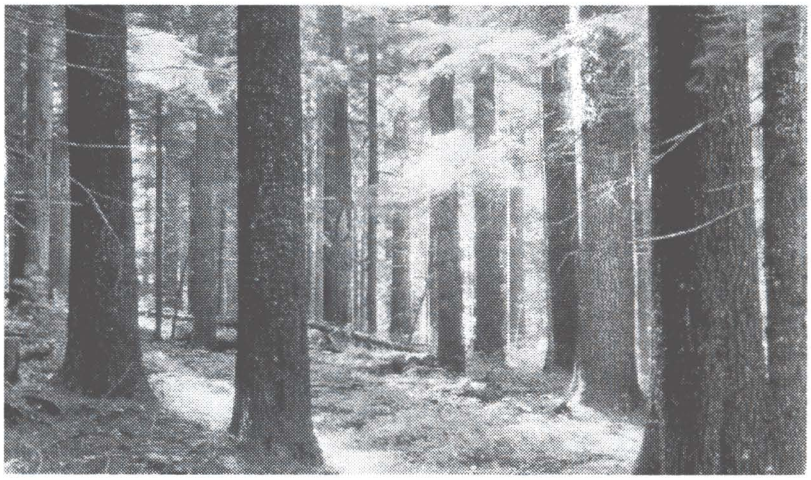
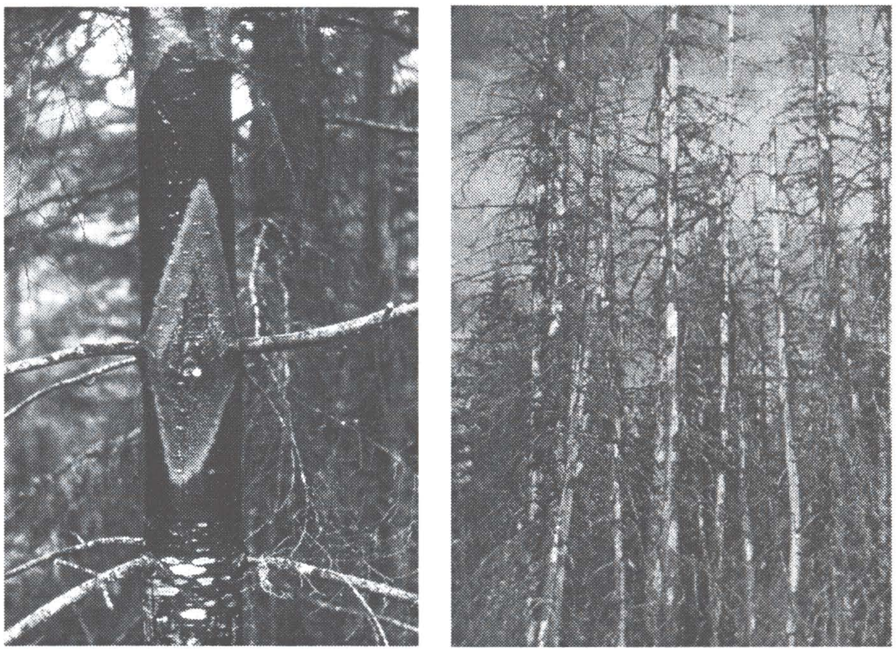


Figure 30.5 Historically (1850 to 1900), 25 to 50% of the moist forests were dominated by *Pinus monticola* and 15 to 80% of the trees within stands were *Pinus monticola*. This photo shows 150- to 180-year-old *Pinus monticola* (c. 1935) growing in northern Idaho (U.S. Forest Service photo).



(A) (B)

Figure 30.6 Photograph A shows a blister rust canker occurring on a young *Pinus monticola*. Photograph B is a mid-aged (70 to 80 years) stand of *Pinus monticola* experiencing extreme mortality from blister rust (U.S. Forest Service photos).

Pinus monticola was not the dominant species; thus, fire exclusion and harvesting were more important agents in altering these forests (Figure 30.7). Harvesting removed the early-seral, shade-intolerant species (*Pinus ponderosa*, *L. occidentalis*) that were resistant to fire and other disturbances. Partial canopy removal and minimal soil surface disturbance in these harvests

Figure 30.7 Harvesting of *Pinus monticola* (c. 1935) with an Idaho jammer. In the moist forests, these ground-lead cable-harvesting systems required closely spaced roads (150 m), which led to the creation of a dense road network within many drainages (U.S. Forest Service photo).

were ideal for *Pseudotsuga menziesii* and *Abies grandis* which regenerated aggressively, rather than the shade-intolerant *Pinus* and *Larix* 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 *Pinus monticola*, *L. occidentalis*, *Pinus ponderosa*, and other early- and mid-seral species is currently absent. In their place, the present forest structure and composition (*Abies grandis* and *Pseudotsuga menziesii*) favor the compression of nutrients, microbial 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.

30.5 Restoration strategies

Changing species composition from late-seral to early- and mid-seral species has been suggested as the key to restoring both moist and dry forests (Everett et al. 1994; Hann et al. 1997; Harvey et al. 1999; Neuenschwander et al. 1999; Finns et al. 2001); by doing this, resilience to epidemics from insects and diseases will increase. Moreover, current altered nutrient and microbial processes would be redirected toward historical conditions when early- to mid-seral species dominated these forests. Returning these forests to a more natural fire regime has also been advocated. Some see forests as they exist today, and because they contain live and green trees, assume that the present forests are typical, resilient, and functioning and should be left alone. In contrast, others conclude that because of the changes that have occurred, the forests of today are less healthy and no longer sustainable. Some see forests as valuable sources of commodities and recreational opportunities and want these benefits maintained. Opinions differ, however, whether these are problems with the Inland Northwest forests and what should be done to restore them. These conflicts have led to widely divergent opinions on the effects that management, fire, or no management have on these forests, and whether intervention can restore them to something resembling past conditions. Most of the general solutions that have been suggested apply a simple fix to a variety of complex problems, often causing mixed beneficial and/or detrimental results that can help or damage a forest, depending on details of history, current conditions, and future trends.

For example, reintroducing ground fire is not a simple matter. Current fuel (live and dead) accumulations and distributions place human, vegetative, and soil resources at high risk (Figure 30.2B). Harvesting to reduce fuels to normal levels has been advocated, but results in more roads with potential soil impacts. Even thinning dense stands, with or without prescribed burning, is problematic, as thinning species susceptible to root disease is likely to set off an epidemic, upset natural selection processes, or worsen fire effects if slash is not appropriately treated (Harvey et al. 1999).

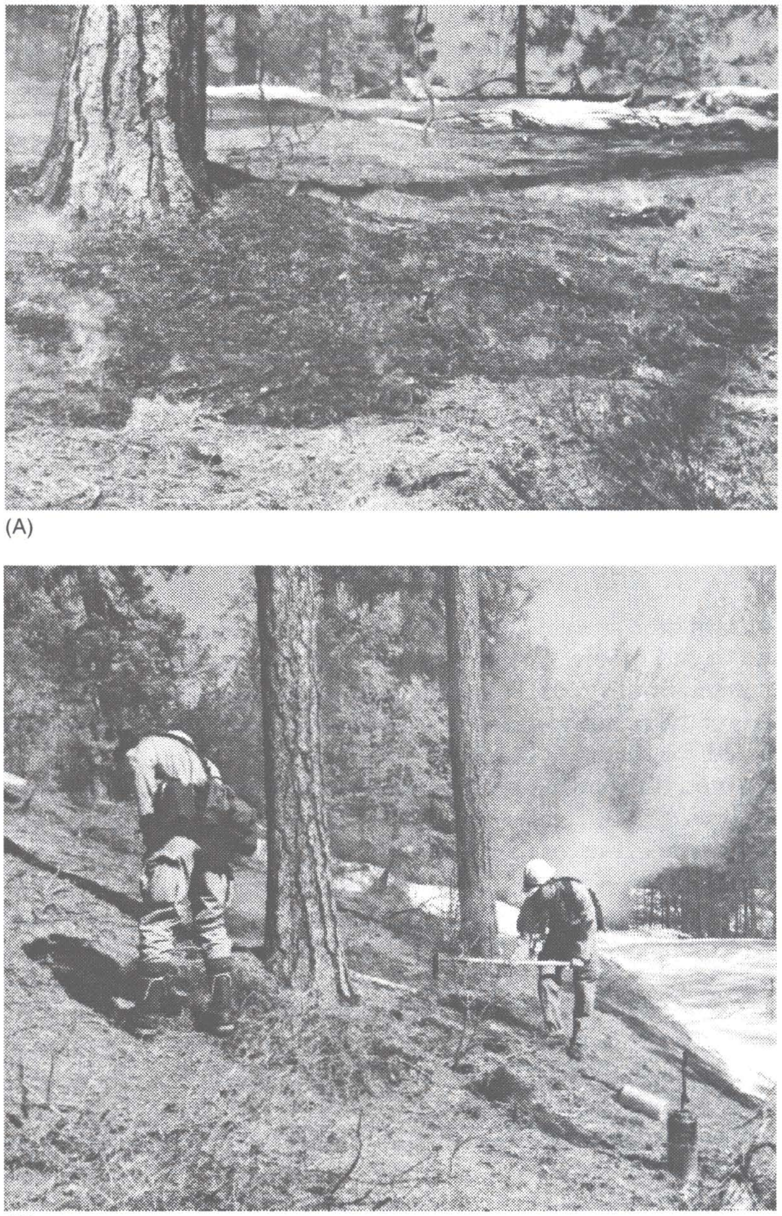
Because these forests are complex and profound changes have occurred, restoration strategies should be applied at the appropriate spatial and temporal scale, both broad (landscape, watershed, centuries) and fine (stand, gap, decades). To date, restored forest conditions (sometimes referred to as reference conditions) are defined by, but not limited to, using estimates of historical structure and composition (Covington and Moore 1994; Kauffman et al. 1994) or characteristics required by wildlife (Reynolds et al. 1992). Large landscape units (2,000 to 4,000 ha) are most applicable when developing restoration strategies. Areas this large probably contain a variety of structures (vegetation mosaics, vegetation patches) and processes (fire, decomposition, management opportunities) not observed at fine spatial scales. Landscape units will likely contain the habitats and food webs of one or more wildlife species (*Accipiter* spp., hawks) that depend on the forests (Reynolds et al. 1992). Multiple land ownerships occur throughout these units, probably with different objectives. Wildfires and the environments they create will still occur because humans are present, but not to the same extent, location, or frequency.

Landscape attributes provide a biophysical template for beginning to set restoration priorities. A step-down process from broad to fine scales for planning restoration activities has been suggested for areas as large as the Interior Columbia River Basin (USDA 2000). Factors such as wide-ranging predators (grizzly bears), anadromous fish (salmon), and social and economic infrastructure (population trends, distribution of cities, towns, and communities) are addressed at the broadest scale. At the mid-scale or river basins, forest composition and distributions can be addressed. For example, in the Coeur d'Alene River Basin (>350,000 ha) of the northern Rocky Mountains, historical patterns of *Pinus monticola* abundance were used to define possible restoration priorities (Jain 2001). At the fine scale, Camp et al. (1997) identified fire refugia as a function of physical landscape attributes in the eastern Cascades. Managers could use these techniques in both dry and moist forests to identify key wildlife habitat, or unique vegetative communities.

Within the priorities established using this step-down process, managers can set restoration targets by defining reference conditions using wildlife indicator species or historical forest structure and composition. Particularly in the dry forests, native surface fires, insects, diseases, and other local disturbances kept vegetation patches relatively small (<1.0 ha). These patches and their characteristic structural stages include live trees, snags, decadent large trees (primarily *Pinus ponderosa* and *L. occidentalis*), coarse woody debris, and organic rich soil. Moreover, these small patches are the preferred habitat for many birds and mammals (Reynolds et al. 1992). An uneven-aged silvicultural system can create and maintain this landscape-scale mosaic in the approximate composition, structure, and landscape patterns existing in dry forests before fundamental changes in natural disturbances regime and forest structure occurred (Reynolds et al. 1992; Long and Smith 2000).

Restoring native fire regimes of frequent surface fires requires approaches that conserve nutrients, microbial activity, and fine roots that developed in the uncharacteristically deep surface organic layers. 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.

Burning when moisture is high (>100%) in the lower layers will preserve them but allow the drier upper layers to be burned (Brown et al. 1985). In addition, burning at low temperatures (less than −2°C) when fine root growth is minimal results in little root damage (Kramer and Kozlowski 1979). These conditions occur in early spring in dry forests when snow is present but the forest floor around the base of trees is clear (Figure 30.8). Under



(A)

(B)

Figure 30.8 Because of fire exclusion, deep organic layers surround many old *Pinus ponderosa*. By applying prescribed fire (A) 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. In addition, mixing (B) the organic layers (increasing decomposition) also decreases organic layer depth (U.S. Forest Service photos).

high moisture conditions (>100% moisture by weight), nutrient volatilization is minimized (Brown et al. 1985). Allowing fine fuels created by mechanical treatments to overwinter before they are burned provides time for nutrients to leach and minimizes volatilization (Harvey et al. 1999).

Silvicultural systems can be developed that maintain high forest cover in moist forests, yet provide sufficient growing space to establish and develop *Larix* or *Pinus* (Jain 2001) and avoid using large openings (clearcuts >16 ha) and their associated roads. Restoration treatments using this fine-scale approach mimic many native disturbances (windstorms, root disease, ice breakage) (Figure 30.3) and their spatial patterns rather than only mimicking large crown fires (Figure 30.4). However, even-aged silvicultural systems (clearcut, shelterwood, seed-tree) are still appropriate for increasing the abundance of resilient species complexes on many lands (Haig et al. 1941; Graham et al. 1983).

The major issue facing restoring the moist forests of the northern Rocky Mountains is the susceptibility of *Pinus monticola* to *Cronartium ribicola* (Figure 30.7). Fortunately, native populations do contain some natural resistance to the rust, thus providing management opportunities. A breeding program currently produces rust-resistant (estimated at 68%) *Pinus monticola* seedlings for reforestation (Fins et al. 2001) and continued breeding will ensure that rust mutations will not compromise the resistant material. 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 *Pinus monticola*, approximately 7 to 10% of the progeny, often thousands per ha, of the survivors exhibit rust resistance. Regenerating these wild populations can supplement the genetic diversity contained in breeding programs.

30.6 Economic and social aspects

Ecological information is available to restore the moist and dry forests (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 presents 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 planning 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 for riparian treatments, with cleaning and weeding costing $200 per ha and prescribed burning costing $80 per ha (USDA 2000).

These cost estimates do not reflect some of the benefits of restoring moist and dry forests. 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. In addition, 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 species such as the bull trout, grizzly bear, and Canadian lynx. Nevertheless, a vocal segment of society prefers the status quo and resists actively managing forests.

Restoration activities will probably affect timber-dependent and isolated communities the most. 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. Active management and restoring forests near these communities most likely would be a positive benefit. However, timber harvesting is usually a controversial issue and likely will continue to be so into the future. In addition, the infrastructure for manufacturing wood products has decreased in recent years, increasing transportation costs for the harvested material (Haynes 2002). Perhaps the greatest economic challenge is that the small trees (especially *Pinus ponderosa*) available for harvesting in most restoration activities have low value (Lippke 2002; McKetta 2002). However, small-diameter (15 cm to 25 cm in diameter) *Abies grandis* and *Pseudotsuga menziesii* seem to be an exception, and are of value in producing construction materials (McKetta 2002).

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 *Pinus ponderosa*, *Pseudotsuga menziesii*, *Pinus 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 resemble a restored forest (Nuenschwander et al. 1999).

30.7 Conclusion

A combination of harvesting and the introduction of *Cronartium ribicola* greatly impacted the moist forests. 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, wildlife habitat) and past (harvesting, road construction) human uses and values, restoring both the dry and moist forests will be challenging, but not impossible. By viewing and developing management strategies using a multiscale approach, areas and treatments can be designed that move these forests on a trajectory toward their historical compositions and structures. Majestic stands of *Pinus monticola*, *Pinus ponderosa*, *L. occidentalis*, *Pseudotsuga menziesii*, 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.

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