Chapter 3.

Fuel Management in Forests of the Inland West

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Introduction

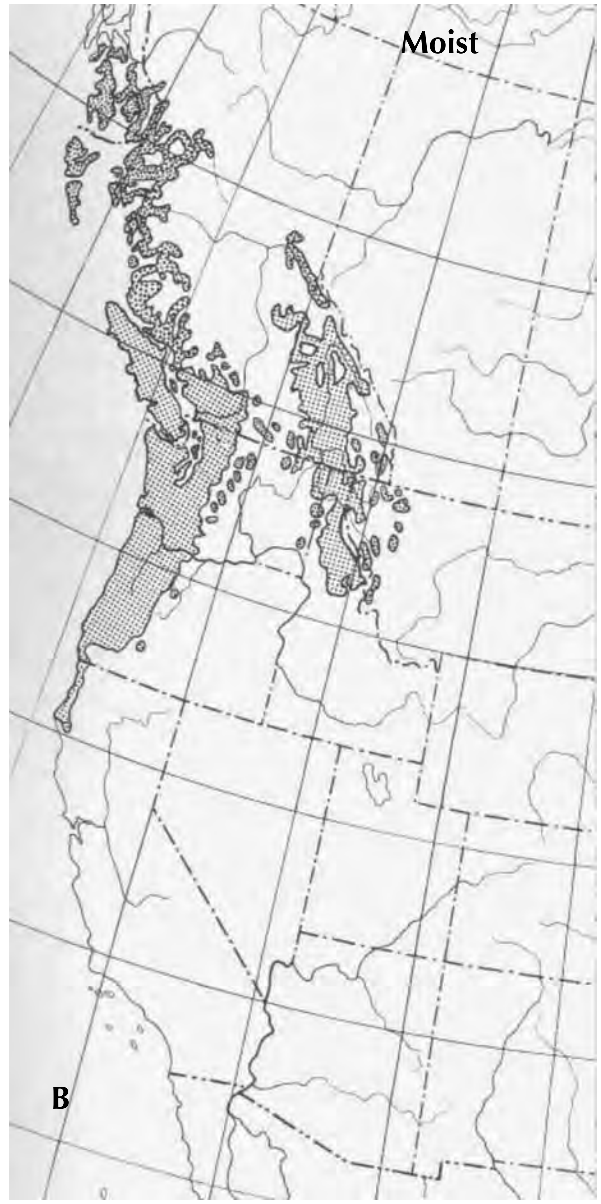
Recent estimates indicate that nearly 40.5 million ha (100 million ac) of forest lands that were historically burned by frequent surface fires in the western United States may benefit from the restoration of surface fire. An additional 4.5 million ha (11 million ac) of forests need to be treated to protect communities from wildfire (Aplet and Wilmer 2003). Rummer and others (2003) estimate that over 26.7 million ha (66 million ac) of forestlands could benefit from fuel reduction. Even with uncertainties in these estimates and arguments as to their precision and accuracy, they clearly illustrate the staggering number of hectares (acres) that need fuel treatments in order to modify fire behavior and burn severity. Access and operability issues further limit the options available on a large portion of western forests. Costs and lack of industrial infrastructure to use small diameter material are other critical factors influencing treatment possibilities. We, the authors, recognize that theoretically, all forests of the western United States could be treated in one way or another to modify wildfire behavior and burn severity. Many of the principles and concepts we discuss are relevant for fuel treatments within other forests and locales; however, we will emphasize forest treatments applicable for use in the cold, dry, and moist forests of the inland western United States. We will discuss forest treatments that influence watershed processes, defined as those that occur when water transports sediment, woody debris, chemicals, heat, flora, or fauna away from a site and deposits it on another site. We define a cumulative effect as one that results from the incremental effects of an event when added to other past, present, and reasonable projected future effects regardless of the triggering action or event (Reid 1988).

Forests of the Inland Western United States

Major river drainages dissect the Rocky, Bitterroot, Salmon, and other mountain ranges along with the Colorado Plateau within the inland western United States. Maritime, continental, and Gulf of Mexico air masses converge and intermingle across this rugged topography with its wide diversity of geologies, soils, and climates that give rise to a plethora of biophysical settings. As the names of the forest classifications infer, what differentiates these forests is their respective differences in climate and biophysical settings and the resulting suite of forest vegetation. Moreover, the biophysical setting, combined with the vegetation composition and structure, result in different disturbances with varying intensities and severities that lead to a variety of interactions and subsequent effects. Fire, grazing, insects, diseases, weather, and timber harvesting, along with vegetation establishment, growth, and succession, interact to create fine (less than 0.1 ha, 0.25 ac) to large (greater than 500 ha, 1,235 ac) mosaics distributed across landscapes. These dynamics occur on some of the most majestic and rugged topography in the world, ranging from less than 300 m (984 ft) above sea level on settings along the Clearwater and Snake Rivers in Idaho to over 3,600 m (11,811 ft) in the mountains of Colorado. There are approximately 147 million ha (363 million ac) of forests within the western United States. About 101 million ha (258 million ac) of these lands are administrated by federal, state, or other public agencies (USDA Forest Service 2001). The forests occupying these lands have a variety of dominant vegetation, ranging from the woodland communities such as chaparral (for example chamise, *Adenostoma fasciculatum*; manzanita, *Arctostaphylos* spp.; *Ceanothus* spp.) in California and pinyon-juniper (*Pinus edulis/Juniperus* spp.) in Arizona, New Mexico, and Utah to forests that can be classified as dry, cold, or moist (Hann and others 1997). Dry forests are typically dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*), moist forests are dominated by grand fir (*Abies grandis*), western redcedar (*Thuja plicata*), or western hemlock (*Tsuga heterophylla*), and cold forests are dominated by Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), or lodgepole pine (*Pinus contorta*). To refine this broad characterization, potential vegetation type (PVT) is often used. Potential vegetation type is a classification system based on the abundance and presence of the potential vegetation (or sometimes called indicator species) that may grow in a particular area in the absence of disturbance. Since not all vegetation can grow in all places because of limitations in water, soils, etc., this vegetation presence and abundance reflects the physical and biological environment (Daubenmire and Daubenmire 1968; Hann and others 1997; Smith and Arno 1999). Classifications usually offer insight into the type of seral or climax vegetation (ground level and canopy) and the different seral stages that can develop within a particular PVT (Cooper and others 1991; Pfister and others1977). In addition, these classifications provide insight into their expected response to disturbance, how they interact with fire, and the distribution of different species.

Moist Forests

Moist forests of the inland western United States occur in 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 the western portion of Montana) (fig. 1). They grow at elevations ranging from sea level to 2,300 m (7,550 ft) (Foiles and others 1990; Graham 1990; Hann and others 1997; Packee 1990; Schmidt and Shearer 1990). The topography is usually steep and broken with V-shaped and round-bottomed valleys. In the northern Rocky Mountains and eastern Cascades, total precipitation averages from 500 to 1,520 mm (28 to 60 in) and is influenced by a maritime climate that tends to favor wet winters and dry summers. Most precipitation occurs during November through May with amounts ranging from 508 mm to 2,280 mm (20 to 90 in) (Foiles and others 1990; Graham 1990; Packee 1990; Schmidt and Shearer 1990). Precipitation comes as snow in the Inland West, often accompanied by cloudiness, fog, and high humidity. Rain-on-snow events are common January through March in the northern Rocky Mountains along with a distinct warm and sunny drought period occurring in July and August with rainfall in some places averaging less than 25 mm (1.0 in) per month. Throughout these forests, the soils are quite diverse and can include sedimentary, metamorphic, and igneous parent materials. Soil orders include, but are not limited to, Spodosols, Ultisols, Entisols, Histosols, Inceptisols, and Alfisols. A defining characteristic of the northern Rocky Mountains is the layer of fine-textured decomposed ash (up to 64 cm, 25 in thick) that caps the residual soils. The combination of climate,



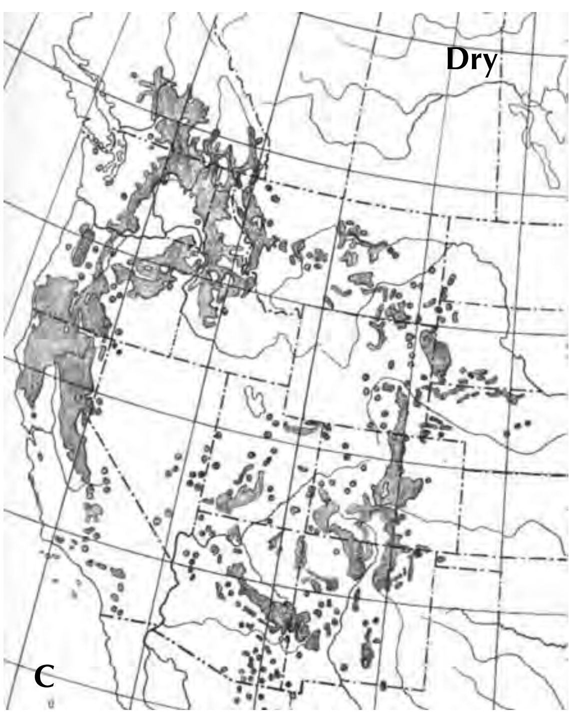


Figure 1. (A) The extent of the cold forests is represented by the range of subalpine fir, (B) the extent of the moist forests is represented by the range of western redcedar, and (C) the extent of the dry forests is represented by the range of ponderosa pine.

topography, parent material, soils, weathering, and ash depth creates the most productive of all forests occurring within the inland western United States.

Vegetation

The vegetation complexes of the moist forests range from early- to late-seral and occur within landscape mosaics possessing all possible combinations of species, seral stages, and structural stages (Cooper and others 1991; Oliver and Larsen 1990). The PVTs in the moist forests of the northern Rocky Mountains include western redcedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), and grand fir (*Abies grandis*). Western white pine (*Pinus monticola*), western larch (*Larix occidentalis*), lodgepole pine (*Pinus contorta*), Douglas-fir, and ponderosa pine occur as the early- and mid-seral species (Daubenmire and Daubenmire 1968; Hann and others 1997). The eastern Cascades and Pacific coast PVTs include western redcedar, western hemlock, grand fir, white fir (*Abies concolor*), Pacific silver fir (*Abies amabilis*), Port-Oxford-cedar (*Chamaecyparis lawsoniana*), incense cedar (*Libocedrus decurrens*), and noble fir (*Abies procera*). On these PVTs, the early- and mid-seral species include lodgepole pine, Douglas-fir, Sitka spruce (*Picea sitchensis*), and ponderosa pine. While less abundant than in the northern Rocky Mountain moist forests, western white pine and western larch do occur as early- and mid-seral species (Franklin and Dyrness 1973; Lillybridge and others 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 vine maple, (*Acer circinatum*), Rocky Mountain maple (*Acer glabrum*), Sitka alder (*Alnus sinuata*), devils club (*Oplopanax horridum*), rose (*Rosa* spp.), gooseberry (*Ribes* spp.), huckleberry (*Vaccinium* spp.), and willow (*Salix* spp.). Forbs include baneberry (*Actaea rubra*), pathfinder (*Adenocaulon bicolor*), wild ginger (*Asarum caudatum*), queencub beadlilly (*Clintonia uniflora*), bunchberry dogwood (*Cornus Canadensis*), and golden thread (*Coptis occidentalis*).

Soil Surface Characteristics and Coarse Woody Debris

Moist forests tend to accumulate large amounts of coarse woody debris (CWD) (fig. 2). Depending on forest age, surface organic layers can contain deep pockets of rotten wood, sometimes representing up to 60 percent of the surface organic horizons (Graham and others 1994; Reinhardt and others 1991). Old forests (greater than 200 years) can have deep (30 cm, 12 in) layers of surface organics and large amounts of CWD ranging between 35 to 72 Mg/ha (15 to 32 tons/ac) (Brown and See 1981; Graham and others 1994).

Cold Forests

Within inland western North America, cold forests generally occur at high elevations (relative to surrounding landscapes) and extend throughout the western United States and into Alberta and British Columbia, Canada (fig. 1). At their northern extent, they occasionally occur at or near sea level and, in southern forests, their range extends to elevations exceeding 3,658 m (12,000 ft) (Lotan and Critchfield 1990). Within the Inland Northwest, they occur primarily in northern Idaho, central Idaho, and the northern Cascades Mountains of Washington (fig. 1). Growing seasons in cold forests are short, ranging from approximately 90 days at low elevations to just a few weeks at the high elevations with frosts occurring any time during the year. These forests are limited by poorly developed soils and, in some areas, by moisture. Nearly all (99 percent) of the cold forests within the inland northwestern United States occur over 1,220 m (4,000 ft), but cold air drainage allows some of them to extend below this elevation (Hann and others 1997; Steele and others 1981). On subalpine fir PVTs, mean annual temperatures range from 3.8 to 4.4 ^{o}C (25 to 40 ^{o}F) with the majority of the precipitation falling as snow and sleet, coming early in the season and staying late. Annual snow fall easily exceeds 1,300 cm (512 in) in the cold forests of the Cascade Mountains with lesser amounts falling where lodgepole pine persists (central Oregon and central Idaho) (Alexander and others 1990). Cold forests have a wide precipitation range with generally less precipitation in southern

Figure 2. A typical mid-aged and mid-seral moist forest containing abundant layers of vegetation and a robust covering on the forest floor consisting of vegetation and coarse woody debris.

cold forests, such as those occurring in New Mexico compared to those in Washington and Idaho. For example, precipitation in the cold forests of the Cascades of Washington ranges from 610 to 2,540 mm (24 to 100 in). Precipitation within the northern Rocky Mountains ranges from 610 to 1,520 mm (24 to 60 in). The central Rocky Mountains receive an average of 610 to 1,400 mm (24 to 55 in) of precipitation and the southern Rocky Mountains receive from 61 to 1,020 mm (24 to 40 in) of precipitation (Alexander and others 1990; Steele and others 1981). Cold forests occurring in southwestern Oregon and California are cool and moist or cold and moist with summer temperatures rarely exceeding 29 ºC (85 ºF) and winter temperatures rarely dipping below -29 ºC (20 ºF). Unique to this area is a 4- to 5-month dry spell between April or May through October where precipitation from thunderstorms is rare. Most precipitation occurs during other months, primarily in the form of snow. The snow pack can exceed 4 m (13 ft) in the cold forests of the Sierra Nevada Mountains of California and in the cold forests of southwestern Oregon and northwestern California, up to 2 m (7 ft) of snow often accumulates. In general, the total precipitation of these California and Oregon cold forests ranges from 750 to 1,500 mm (30 to 60 in). The soils supporting the cold forest are relatively young, often shallow, and poorly developed. Mountain glaciers extensively covered these forests during the Pleistocene and have generally been free of ice less than 12,000 years. Most soil parent material is alluvium or glacial tills, but soil surfaces range from very weakly weathered (rocky with no organic layers) to thick soils composed primarily of organic materials. The soil orders common to the cold forests include the Entisols, Inceptisols, Alfisols, and Spodosols (Laacke 1990).

Vegetation

The potential vegetation types dominating the cold forests include subalpine fir (with and without Engelmann spruce) and mountain hemlock (*Tsuga mertensiana*). These

PVTs occur at the highest elevations within the Rocky and Cascade Mountains. Western larch and lodgepole pine are early-seral species in the subalpine fir/Engelmann spruce PVT, Douglas-fir and western white pine are mid-seral species, and western redcedar, grand fir, Engelmann spruce, and subalpine fir are late-seral. The species composition that occurs in the subalpine fir/Engelmann PVT is highly dependent on elevation, associated climate, and disturbance frequency and type. In California, the PVT's dominating the cold forests include red fir, white fir (*Abies concolor*), and noble fir (*Abies procera*), which are also the dominant late-seral species. Lodgepole pine, western white pine, incense-cedar, Brewer spruce (*Picea breweriana*), and Jeffery pine (*Pinus jefferei*) are common early-seral species (Laacke 1990).

Like the moist forests, lush ground-level vegetation is the norm for most settings in the cold forests. Tall shrubs include false huckleberry (*Menziesia ferruginea*) and Sitka alder, and the dominant medium and low shrubs are often huckleberries. Pinegrass (*Calamagrostis rubescens*), bluejoint reedgrass (*Calamagrostis canadensis*), and elk sedge (*Carex geyeri*) typify the graminoids occurring in the cold forests. Some of the most commonly occurring forbs include beargrass (*Xerophyllum tenax*), round-leaved violet (*Viola orbiculata*), and queencup beadlilly (*Clintonia uniflora*) (Cooper and others 1991).

Medium sized shrubs include species such as gooseberry (*Ribes* spp.), pinemat manzanita (*Arctostaphylos nevadensis*), mahala mat (*Ceonothus prstratus*), mountain pride (*Penstemon newberryi*), and mountain whitethorn (*Ceonothus cordulatus*). Areas with deep soils often contain bush chinquapin (*Chrysolepis sempervirens*) or greenleaf manzanita (*Arctosstaphylos partula*). Small upland meadows are common in these forests and provide habitats for a wide variety of sedges, grasses, and forbs (Laacke 1990).

Soil Surface Characteristics and Coarse Woody Debris

In cold forests, such as along the Continental Divide in central Montana, slow decomposition causes needles and other surface litter to accumulate and create dense organic mats from 2 to 8 cm (0.75 to 3.0 in) thick intermixed with large boulders in some areas (fig. 3). Coarse woody debris amounts vary depending on forest age. In mature subalpine forests in Idaho and Montana, CWD can range from 22 to 55 Mg/ha (13 to 25 tons/ac) (Brown and See 1981; Graham and others 1994). In young forests where lodgepole pine dominates the site as an early-seral species, up to 40 percent of the soil surface can be composed of CWD and rotten wood (Brown and See 1981; Harmon and others 1986) (fig. 3).

Dry Forests

Dry forests are typically dominated by ponderosa pine and/or Douglas-fir and can occur throughout the western United States, southern Canada, and northern Mexico (fig. 1) (Little 1971). Their greatest extent is in the inland northwestern United States and in northern California. These forests also occur in the Black Hills of South Dakota and Wyoming, along the Front Range of the Rocky Mountains in Colorado, and along the Mogollon Rim in Arizona, the rugged escarpment that forms the southern limit of the Colorado Plateau. Elevations range from sea level to 3,281 m (10,000 ft) depending on latitude (Oliver and Ryker 1990).

The topography of the dry forests is highly variable and dependent on the region where they occur. East of the Continental divide in central Montana, South Dakota, eastern Wyoming, and central Nebraska, the dry forests tend to occur within discontinuous mountainous regions, plateaus, and canyons intermixed with the plains. Large expanses of dry forests dominated by ponderosa pine occur in the Black Hills of South Dakota and Wyoming, which is largely an isolated mountain range (fig. 4). In the southwestern Rocky Mountains (Colorado, New Mexico, Arizona, Utah), dry forests can occur on flat plateaus to steep mountain slopes and often populate all slope aspects depending on location (Boldt and Van Deusen 1974; Oliver and Ryker 1990).

Figure 3. (A) Mature lodgepole pine growing on a subalpine potential vegetation type typifying much of the cold forests. (B) The inset shows a large accumulation of brown rotten wood that is often present in many cold forests.

Figure 4. The Black Hills of western South Dakota are dominated by late-seral ponderosa pine forests.

Average annual temperatures in dry forests range between 5 and 10 °C (41 and 50 °F), and July and August temperatures average between 17 and 21 °C (62 and 70 °F). The number of frost-free days ranges from 90 to 154 in eastern Montana and South Dakota and to more than 200 days in central California (Oliver and Ryker 1990). Soil moisture influences growth and development of these forests and is highly variable depending on location. In the southwestern United States, Colorado Rockies, Black Hills, and Utah forests, summer rains often occur although little precipitation occurs in May and June. In eastern Oregon and Washington, July through September is usually dry with precipitation amounts in Montana and east of the Continental divide averaging from 280 to 430 mm (11 to 17 in), with approximately half of it occurring from May to August. In the dry forests of eastern Oregon, Washington, and Idaho, precipitation in general averages from 355 to 760 mm (14 to 30 in) with much of it falling in the form of snow. In these forests, July to September are generally dry with precipitation averaging less than 25 mm (1 in). The dry forests growing on the west slope of the Sierra Nevada Mountains in northern California receive on the average 1,750 mm (69 in) of precipitation with very little moisture falling in July and August.

Parent material supporting the dry forests comes from a variety of substrates including igneous, metamorphic, and sedimentary rocks, and they often include quartzite, argillite, schist, shale, basalt, andesite, granite, cinders, pumice, limestone, and sandstone. The soils derived from these materials are most often included in the orders Entisols, Inceptisols, Mollisols, Alfisols, and Ultisols. The vegetation distribution within the dry forests is a function of available moisture, with soil texture and depth influencing the availability.

Vegetation

Ponderosa pine is the primary conifer that defines the dry forests throughout much of the western United States, southern Canada, and northern Mexico (fig.1) (Little 1971). It is the principle species on over 27 million acres and, for every 2.8 ha (7 ac) that it dominates, it is present on an additional 1.4 ha (3.5 ac) (fig. 5). Within the western United States, California alone contains the greatest concentrations of ponderosa pine (2.07 million ha, 5 million ac) closely followed by Oregon (1.9 million ha, 4.7 million ac). When combined, Arizona and New Mexico contain an additional 2.5 million ha (6 million ac) of ponderosa pine (Van Hooser and Keegan 1988). The species occupies sites with elevations ranging from sea level to 3,050 m (10,000 ft) depending on latitude (Oliver and Ryker 1990). In terms of area occupied, it is second only to Douglas-fir.

In the southern and extreme eastern portion of the range, ponderosa pine grows primarily on ponderosa pine PVTs. On these settings, quaking aspen (*Populus tremuloides*) is the most frequent early-seral tree species (Huffman and Alexander 1987; Youngblood and Mauk 1985). Ground-level vegetation includes grasses (for example *Festuca* spp. *Agropyron* spp.) and shrubs such as snowberry (*Symphorcarpus* spp.), spirea (*Spirea* spp.), and russet buffaloberry (*Shepherdia canadensis*) (fig. 5).

With increasing moisture, ponderosa pine occurs as a mid-seral species and Douglas-fir becomes the late-seral species. Quaking aspen and lodgepole pine are early-seral associates of ponderosa pine on these Douglas-fir PVTs (Mauk and Henderson 1984). These ponderosa pine forests occur in the Rocky Mountains along the Front Range of Colorado, in Utah, and in southern Idaho. They also occur along the western slopes of the Sierra Nevada Mountains in California and the eastern slopes of the Cascades in Oregon (Franklin and Fites-Kaufman 1996; Hann and others 1997). On these settings, ground-level vegetation includes ninebark, elk sedge, and pine grass. These species, in particular, exemplify aggressive survivors after disturbance (for example, fire, mechanical site preparation) and can colonize a site quickly after disturbance (Baumgartner and others 1986).

In several locales, dry grand fir and/or white fir PVTs represent the dry forests (Hann and others 1997). On such settings, ponderosa pine and Douglas-fir occur but are succeeded by late-seral grand fir and/or white fir in the absence of disturbance (Bradley and others 1992a) (fig. 6). Additional trees that can occur in such forests include juniper,

Figure 5. Ponderosa pine growing in central Oregon. (A) Note the open understories and high crown base heights of these mature trees and the forest floor covered by a layer of needles. This needle layer does not readily decompose and provides a highly flammable fuel. (B) The inset shows lush ground-level vegetation developing after a ponderosa pine timber harvest on a Douglas-fir potential vegetation type in central Idaho.

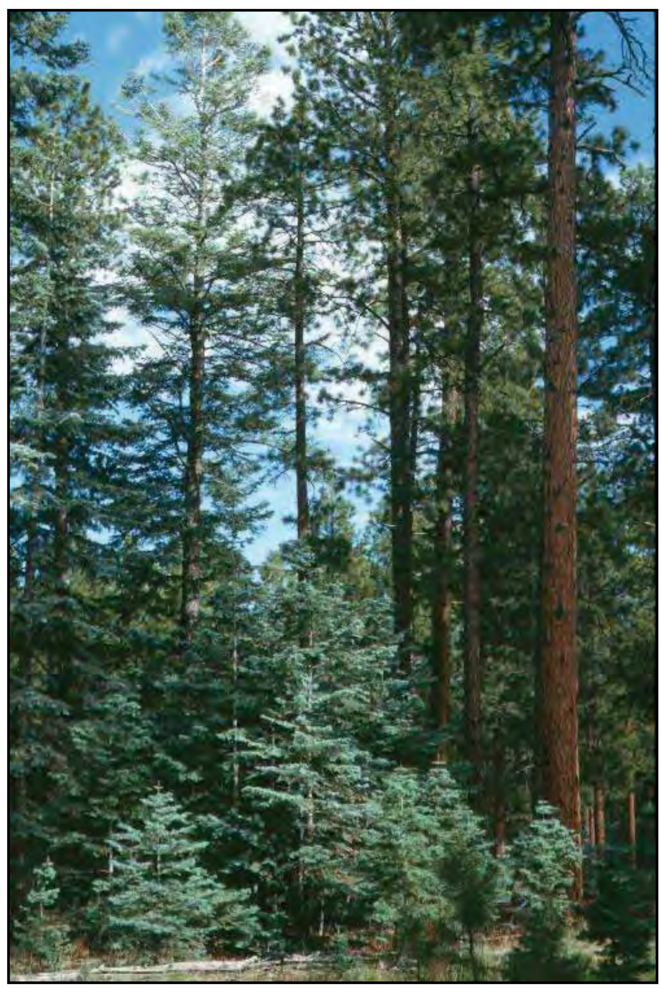


Figure 6. On the Douglas fir and grand fir potential vegetation types, ponderosa pine is readily succeeded by these species in the absence of disturbance (for example fire, harvesting). This succession results in abundant fuel layers and nutrient rich crowns that extend to the soil surface making them highly susceptible to loss during fires.

pinyon pine, sugar pine (*Pinus lambertiana*), incense-cedar (*Libocedrus decurrens*), western larch, Jeffrey pine, and lodgepole pine. Pine grass and ninebark (*Physocarpus malvaceus*) are frequent associates, but tall shrubs such as Rocky Mountain maple often occur.

Soil Surface Characteristics and Coarse Woody Debris

In the dry forests, needle shed from long needle pines is a major component of the surface litter (fig. 5). Without fire, these layers can quickly accumulate, often minimizing the amount of grasses, forbs, and shrubs that were more prevalent on these sites when fire maintained smaller amounts of these litter layers. Coarse woody debris (CWD) is relatively scarce when compared to moist and cold forests (figs. 2 and 5). However, the more productive dry forests tend to accumulate relatively high amounts of CWD, especially in the absence of fire. For example, on grand fir and Douglas-fir PVTs in Montana and Idaho, CWD ranges from 10 to 30 Mg/ha (5 to 20 tons/ac). In contrast, on ponderosa pine PVTs in Arizona, CWD can range from 10 to 20 Mg/ha (5 to 10 tons/ac) but can quickly accumulate where white fir are the late-seral species, as exemplified by the 10 to 30 Mg/ha (5 to 15 tons/acre) reported by Graham and others (1994) for these PVTs.

Forest Change

Most forest conditions of the western United States have changed from those that occurred historically (pre-1900). In particular, many of these changes relate to both the decrease in fire occurrence and the increase in the size and severity of how the wildfires burned. The changes in how fires burn, often because of successful fire exclusion, combined with timber harvesting and climate cycles resulted in changes in the abundance, composition, and distribution of tree, ground level vegetation, and forest floor components within the three-forest types (Graham and others 2004; Jain and Graham 2005). These changes include shifts in species composition, volume, and density, thus creating situations such as altering vegetative chemistry and the biological and chemical composition of the forest floor (Harvey and others 1999a) (fig. 6).

For the most part, western forests of the late 1800s and the first half of the 20th century produced timber crops for expanding local and national economies. Thus, the forests were aggressively protected from damaging animals, insects, diseases, and fire (Graham and Jain 2004). The species harvested included western white pine, western redcedar, western larch, ponderosa pine, and Douglas-fir that produced diverse products such as toothpicks, matches, ship masts, railroad ties, fuel, and building material. At this time, many tree and shrub species were considered weeds (for example, western hemlock and grand fir) and were removed (slashed and burned) to provide growing space for product producing species. In many locales throughout the western United States, this product centric forest management continued into the 1950s and early 1980s (Davis 1942; Graham and others 2005; Steen1976).

Two large fire events, the Peshtigo fires that burned in northern Wisconsin in October of 1871 and the wildfires that burned in northern Idaho, eastern Washington, and western Montana in August of 1910, burned millions of forested hectares (acres) and towns and caused loss of life (Gess and Lutz 2002; Pyne 2001) (fig. 7). These destructive events were the catalyst that cemented the mission of the newly formed Forest Service to protect the valuable forestlands of the United States from damaging agents (Steen 1976). The result was a significant decrease in the number of fires burning and the amount of area burned in western forests. The effectiveness of fire exclusion increased dramatically especially with the use of airplanes and smoke jumpers to aerially combat fires and the development of access roads (Graham and others 2004; Graham and Jain 2004). Although fire exclusion and timber harvesting influenced the moist and cold forests, the dry forests had the most noticeable changes (Covington and Moore 1994; Hann and others 1997).

Figure 7. The wildfires that burned in the northern Rocky Mountains in 1910 consumed over 1. 2 million ha (3 million acres) and burned communities, reinforcing the need to protect forests from disturbances and fire in particular.

Since the dry years of the 1930s, the climate of the western United States has been relatively moist, facilitating the regeneration and development of large amounts of forest vegetation (Haig and others 1941; Pearson 1950). Historically, native insects (for example, pine beetle [*Dendroctonus* spp.]), diseases (for example, root rots [*Armillaria* spp.]), and mistletoe (*Arceuthobium* spp.) infected and killed the very old or stressed individuals, which tended to diversify vegetation communities (Hessburg and others 1994). However, in present forests, changes in vegetation have facilitated development of unprecedented epidemic levels of these insects and diseases in many locales (fig. 8). These disturbance agents were often encouraged by weather events such as ice storms, windstorms, and periodic droughts. In addition, the movement of Europeans into the

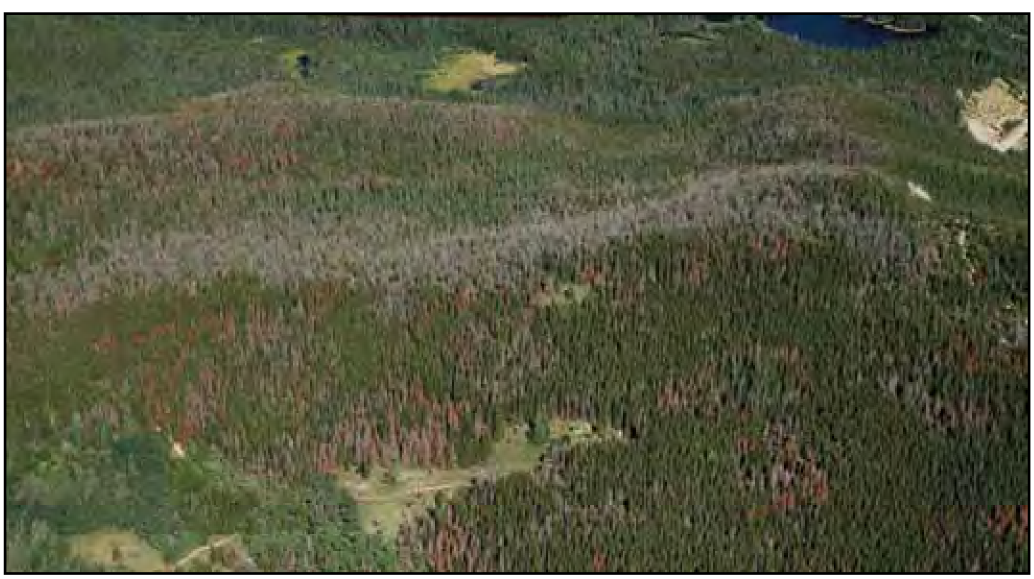


Figure 8. Lodgepole pine killed by mountain pine beetle in Colorado (photo by William M. Ciesla, Forest Health Management International, Bugwood.org).

western United States resulted in the intentional and inadvertent introduction of both exotic plant and animal species that often displaced native species. Non-native insects and diseases often found no natural checks or balances, enabling them to invade western forest ecosystems unimpeded. In concert, these and allied events have facilitated the development of abundant amounts of forest vegetation and, in some settings, a different suite of vegetation, insects, and diseases than those that occurred prior to 1900.

Fire Regimes

Fires, in concert with other disturbances, create mosaics of forest composition and structure within and among stands and across landscapes. These mosaics can occur over relatively fine spatial scales (in other words, less than 0.5 ha, 1.2 ac) to rather large mosaics exceeding hundreds of hectares (acres). Wildfires historically burned the cold, moist, and dry forests at various intensities and frequencies giving rise to a wide variety of burn severities and producing a variety of vegetative successional pathways and a diversity of vegetation and forest floor conditions. A fire regime is a generalized description of the role fire plays in a forest and is related to a fire's frequency, severity, and intensity (Agee 1993). While multiple fire regime classifications are available, we find that for describing fires occurring in the dry, cold, and moist forests, a system containing three classes—stand replacing fires (lethal), mixed fires, and low intensity surface fires (non-lethal)—is most useful (Hann and others 1997; Schmidt and others 2002).

The most extreme fire regime, the stand replacing (lethal) regime, kills all canopy layers across stands or relatively large areas (in other words, areas larger than ~2 ha, 5 ac). In general, these fires kill the standing vegetation and could be classified as intense (all black stems), but some moderately severe (brown needles present) burning may exist in numerous areas. Within forests containing burned and blackened trees, the level of soil burn severity can be highly variable, ranging from light to highly severe depending on the state of the ground-level vegetation, surface fuels, forest floor, and burning conditions during the fire.

Low intensity surface fire (non-lethal) regimes clean the forest floor of vegetation and accumulated woody debris but yet leave the majority of the high forest cover alive. These non-lethal fires rarely kill dominant trees, but soil burn severity can range from nothing to highly severe depending on the amount, condition, and extent of the surface fuels and the heat and residence time of the fire.

Mixed severity fires combine lethal and non-lethal fires, killing small to medium sized groups of dominant trees, burning surface fuels in other areas while leaving the dominant trees alive. Mixed fire spatial extents can range from small patches of vegetation to rather large landscapes that burned for days to weeks. By their nature, fires described by this regime leave compositionally and structurally fine scale mosaics of forest vegetation and simple to complex burn patterns. With the advent of successful fire exclusion, these mixed severity fires were frequently extinguished preventing them from modifying many forested landscapes. Historically, all three fire regimes occurred in all three forest types allowing fire suppression activities, to some degree, to impact all three forest types.

Moist Forest Changes

Fire exclusion, insects, diseases, and weather interacted with climate change (climate cycles) to alter the moist forests from one dominated by early-and mid-seral species to one dominated by mid-to late-seral species. Historically, wildfires burning the moist forests were highly variable. Estimates suggest that nonlethal surface fires occurred at relatively frequent intervals (15 to 25 years) within 25 percent of the moist forests. Lethal crown fires burned about 25 percent of the moist forests every 20 to 150 years but occasionally extended to 300-year intervals. The mixed fire regime burned across about 50 percent of the moist forests at 20- to 150-year intervals with some lethal events occurring at 300-year intervals. Fires typically started burning in July and were usually out by early September with the change in weather. Most fires were small, but occasionally,

Figure 9. Blister rust, an introduced Eurasian disease, significantly reduced the abundance of western white pine from much of its historical range.

large fires did occur with 74 percent of fires killing a portion of the canopy (Hann and others 1997; Pyne 2001) (fig. 7). Because of fire exclusion, surface fires now burn 10 percent of the moist forests, mixed fires burn 30 percent, and crown fires burn 60 percent of the moist forests. Although fire exclusion played a role in altering the moist forests in the western United States, introduction of a European stem rust, white pine blister rust (*Cronartium ribicola*), was a greater factor in causing change in these forests (fig. 9) (Jain and Graham 2005). The disease attacked and killed western white pine, a primary commercial species. Upon introduction of blister rust, a massive effort was initiated to control the disease by removing currant (*Ribes* spp.) bushes (the alternate host). During the Civilian Conservation Corp days of the 1930s, workers pulled, sprayed, and grubbed *Ribes* bushes throughout the northern Rocky Mountains. Hutchison and Winters (1942) described the effort "like bailing the ocean with a teacup" and efforts to control the disease proved futile. By 1968, the western white pine blister rust program and management of the species was "realigned," resulting in the accelerated removal of naturally occurring western white pine before they supposedly succumbed to the rust (Ketcham and others 1968).

Harvesting of ponderosa pine and western larch, other high value timber species, exacerbated the forest changes resulting from the introduction of blister rust. The partial and intermittent canopy removal and minimal soil surface disturbance caused by both tree harvest and western white pine mortality were ideal situations for grand fir and western hemlock to establish and aggressively encroach. In the eastern Cascades, the effect of blister rust on forest composition and structure was less severe since western white pine was not as dominant, thus fire exclusion and harvesting were more important change agents in these forests. Grand fir and Douglas-fir readily filled the niches western white pine, ponderosa pine, and western larch once held.

Since 1991 native insect and pathogen activity in the moist forests far exceed those of the past. The Douglas-fir beetle (*Dendroctonus pseudotsugae*), mountain pine beetle (*Dendroctonus ponderosae*), spruce budworm (*Choristoneura occidentalis*), and tussock moth (*Orgyia pseudotsugata*) that were historically endemic are now often epidemic. Similarly, the root diseases *Armillaria* spp. and *Phellinus weirii* that were historically endemic are now common in the current fir-dominated forests (Hann and others 1997; Hessburg and others 1994).

Weather, another formidable disturbance, in the form of snow or wind, often creates a variety of canopy openings ranging from gaps to large openings (~16 ha, 40 ac). In early-seral dominated species (ponderosa pine, western larch, western white pine) forests, snow will often slip from the trees, minimizing breakage, while other species in the intermediate crown classes (grand fir, Douglas-fir) will break, creating gaps and openings, decreasing forest densities, and altering species composition. Today, species that dominate many sites, such as grand fir and Douglas-fir, have dense crowns that hold snow and tend to break more readily compared to western larch or western white pine, which shed the snow (Graham and Jain 2005; Jain and Graham 2005).

Disturbances in the moist forests, singularly or in combination, or the lack thereof, have culminated in the current distribution of successional-stage, forest structure, species composition, and disturbance regimes that differ from historical conditions (1850 to 1900) (Hann and others 1997). In some settings, the mixed-fire regime maintained closed canopy conditions that allowed the mid-seral stage to develop into late-seral, multi-story stages (Hann and others 1997). The late-seral, multi-story structure, which typically developed in cool, moist bottoms and basins, has decreased by about half in the last century. The early-seral, single-story stands that once occupied an estimated 25 to 30 percent of the area now occupy only 9 to 10 percent, 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.

Because of fire, weather, disease, and insect interactions, species composition has shifted in the moist forests of the northern Rocky Mountains (Fins and others 2001; Hann and others 1997; Neuenschwander and others 1999). For example, before 1900, western white pine (early-to mid-seral species) dominated many settings, often representing 15 to 80 percent of the trees within stands (Fins and others 2001). Western larch and ponderosa pine also occurred in the early-and mid-seral structures but declined along with western white pine and were succeeded by grand fir, Douglas-fir, and western hemlock (Atkins and others 1999; Hann and others 1997). The eastern Cascades had limited amounts of western white pine and western larch; therefore, ponderosa pine, lodgepole pine, and Douglas-fir played more of a role in occupying the early-to mid-seral successional stages.

Western white pine, ponderosa pine, and western larch-dominated forests are generally tall and self-pruning, even in moderately dense stands. They have large branches high in the crowns and the base of the crowns is well above surface fuels. In general, this crown architecture protects the nutrients stored in the canopy from surface fires. In contrast, young- to mid-aged (less than 150 years) western hemlock and grand fir/white fir generally do not self prune. Forests dominated by grand fir tend to concentrate both nitrogen and potassium in their foliage, which often extends to the soil surface. In general, low canopy structure, combined with nutrient and microbial activities concentrated in or near the soil surface, makes these two critical ecological resources susceptible to mechanical and fire destruction (Harvey and others 1999a; Mika and Moore 1990; Minore 1979; Moore and others 1991).

A positive change in the moist forests is the introduction of western white pines that are resistant to blister rust (Bingham 1983). Tree improvement programs that began in earnest in the 1950s started yielding rust-resistant seedlings for use in reforestation. By 2005, millions of rust resistant seedlings have been planted throughout the moist forests. In addition, some native resistance to the disease remained in non-harvested residual western white pines. As a result, western white pine has many opportunities to play both significant ecological and commercial roles in the moist forests (Graham and Jain 2005).

Cold Forest Changes

Depending on the physical setting, estimates suggest that the cold forests of the inland northwestern United States historically (1850 to 1900) burned in the range of 25 to 100 year intervals. Non-lethal surface fires burned approximately 10 percent of these forests, every 30 to 100 years. Lethal crown fires burned 25 to 30 percent of the cold forests every 30 to 100 years with the longer intervals occurring in moist areas. During the short fire season (~60 days), a mixed fire regime burned about 60 percent of the cold forests at 25- to 100-year intervals, with occasional large fires occurring every 100 years (Hann and others 1997). It is estimated that the proportions of the cold forests burned by non-lethal (10 percent), mixed fires (60 percent) and lethal fires (30 percent) is similar to the proportions historically burned by the different fire regimes. Therefore, fire exclusion has had minimal influence on these forests. However, local impacts on forest composition and structure are very likely the result of fire exclusion, as are the locations where the different fire regimes likely burned. In turn, the vegetative mosaic and its texture (size, shape, and location of vegetation patches) most likely differ from the historical characteristics of these forests (Bradley and others 1992a).

In the subalpine fir PVT, fire maintained 23 percent of the cold forest in early-seral vegetation primarily dominated by lodgepole pine. In these areas, lethal fires created ideal conditions (for example, ~10 acre openings and larger, burned-over and mineral soil surfaces) for the regeneration of lodgepole pine. Extremely dense stands of lodgepole pine would develop and, without subsequent disturbance, dominate settings for decades. Similarly, in some of the drier and colder portions of the cold forests, a single canopy of lodgepole pine could persist for over a hundred years.

Historically, mid-seral structures occupied about 53 percent of the cold forests (Hann and others 1997). A mixed fire regime in these forests, along with periodic wind, floods, snow, and other small-scale disturbances, allowed uneven-aged and patchy stands to develop. Engelmann spruce, western redcedar, grand fir, mountain hemlock, and subalpine fir would readily regenerate in a variety of canopy conditions (gaps), giving rise to dense stands with many canopy layers. In the absence of lethal fires, the warmer forest settings containing lodgepole pine were readily succeeded by Engelmann spruce and subalpine fir.

Prior to 1900, late seral, multi-storied forests occupied 15 percent of the cold forests (Hann and others 1997), and subalpine fir and Engelmann spruce dominated these forests with lodgepole pine and Douglas-fir as intermittent associates. Forest densities were often high (exceeding 29,000 trees/ha, 11,733 trees/ac) and dominated by subalpine fir and Engelmann spruce but, occasionally, a few large, fire resistant Douglas-firs would be interspersed throughout these multi-storied forests (Franklin and Mitchell 1967). Non-lethal surface fires would encourage single canopies of subalpine fir, Engelmann spruce, or lodgepole pine to develop over approximately 8 percent of the cold forests.

Due mainly to harvesting, and to a lesser degree lethal fires, the greatest vegetative changes occurring in cold forests over the last 100 years is the increase in the extent of early-seral structures consisting primarily of lodgepole pine. Large (100s to 1,000s of ha) expanses of lodgepole pine forests, especially those with large (greater than 20-cm, 8-in) diameters, are often killed by mountain pine beetle (fig. 8). Also Douglas-fir beetle, spruce budworm, tussock moth, and spruce bark beetle (*Dendroctonus rufipennes*) appear to be defoliating and killing trees more readily in the cold forests than they did historically. Singly, and in combination, the defoliation and mortality caused by these insects can provide large amounts of dead fuel for future fire events (Hann and others 1997) (fig. 8).

Dry Forest Changes

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

Before successful fire exclusion, temperature and precipitation patterns combined with natural and human ignitions that allowed fires to burn the dry forests at relatively frequent (for example, less than 40 years) intervals. Cultural burning by Native Americans augmented and even dominated burning in several locations (Barrett and Arno 1982; Stewart 1951). In the northern Rocky Mountains of Idaho and western Montana, dry settings (ponderosa pine and/or Douglas-fir PVTs) were historically burned by non-lethal wildfires at 15- to 23-year mean return intervals. These fires could be quite large often burning for weeks to months (Weaver 1943). Mixed fires frequently burned mesic forests containing ponderosa pine (grand fir and/or Douglas-fir PVTs) at mean return intervals extending to over 60 years. Non-lethal fires dominated the central and southern Rockies (ponderosa pine and/or Douglas-fir PVTs), although mixed severity fires also occurred, especially along the Front Range of the Rocky Mountains in Colorado (Bradley and others 1992a; 1992b; Fulé and others 1997; Kaufmann and others 2001). Fires tended to be few on the driest settings (ponderosa pine and/or woodlands) because of discontinuous surface fuels (Bradley and others 1992a). In contrast to other locales dominated by late-seral ponderosa pine, the forests of the Black Hills of Wyoming and South Dakota possibly experienced greater extents of lethal fires because of the abundant ponderosa pine regeneration that normally occurred (Shepperd and Battaglia 2002; Shinnen and Baker 1997). Nevertheless, historical wildfires likely burned through most ponderosa pine forests leaving in their wake a wide variety of species compositions and vegetative structures arranged in a variety of mosaics.

Within the dry forests, dense conditions often develop and are exacerbated by fire exclusion, increasing the abundance of insect and disease epidemics, which significantly altered the composition, and structure of these forests (Harvey and others 2000). Historically, western pine beetle (*Dendroctonus brevicomis*), pine engraver (*Ips* spp.), fir engraver (*Scolytus ventralis*), and Douglas-fir tussock moth were insects associated with regularly burned areas (Hessburg and others 1994). In most years, bark beetles occurred at endemic levels in ponderosa pine, Douglas-fir, and grand fir, killing large weakened trees that were struck by lightning, infected by root disease (*Armellaria* spp.), or too old to resist attack (Williams and others 1986; Wu and others 1996). Pine engraver and fir engraver beetles attacked young, densely stocked ponderosa pine, killing trees scorched by low-intensity surface fires and severely infected trees containing root rot or dwarf mistletoe.

Because of forest change, these same insects have reached epidemic levels in many forests (Gardner and others 1997; Hedden and others 1981). Today, ponderosa pine continues to be susceptible to the western pine beetle, and the mountain pine beetle is prevalent on Douglas-fir and grand fir PVTs. The pine engraver beetle is more abundant and destructive now with some of the severest outbreaks occurring on low-elevation ponderosa pine PVTs (Hessburg and others 1994). Pandora moth (*Coloradia pondora*) defoliates ponderosa pine and scattered outbreaks have occurred in Arizona, California, Colorado, and Oregon during the 20th century (Speer and others 2001). Dense stands of Douglas-fir and grand fir that developed on many grand fir and Douglas-fir PVTs are very susceptible to both defoliators and root diseases.

Harvesting western larch and ponderosa pine precipitated the regeneration and growth of grand fir/white fir and Douglas-fir in the dry forests, which subsequently facilitated the accumulation of both above-and below-ground biomass and associate nutrients close to the soil surface (Harvey and others 1986) (fig. 6). As a result, even low-intensity surface fires often consume the surface organic layers, killing tree cambiums and/or fine roots, volatilizing nutrients, killing trees, and increasing soil erosion potential (Debano 1991; Hungerford and others 1991; Robichaud and others 2000; Ryan and Amman 1996). In addition, the abundant fir in the understory creates nutrient-rich ladder fuels that facilitate crown-fire initiation, increasing the likelihood of nutrient loss (Harvey and others 1999a; Minore 1979; Van Wagner 1977). The risk of nutrient loss is greater on infertile sites because dense stands of late-seral species are more demanding of nutrients and water than the historical stands dominated by widely spaced early-seral species (Harvey and others 1999a; Minore 1979) (see Chapter 9).

With the advent of fire exclusion, animal grazing, timber harvest, and climate cycles, on the moist potential vegetation types (for example grand/white fir), ponderosa pine is being succeeded by Douglas-fir, grand fir, and/or white fir (fig. 6) (Arno and others 1997; Graham and others 2004; Gruell and others 1982). Fire intolerant vegetation, dense forest canopies, and homogenous and continuous horizontal and vertical structures are developing, thus creating forests favoring crown fires rather than low intensity surface fires that historically occurred in these forests (Arno and Brown 1991; Dodge 1972; Peterson and others 2005; Van Wagner 1977). Within the Inland Northwest, the extent of mid-seral (for example, Douglas-fir) vegetation has increased by nearly 3.2 million ha (8 million ac) and the extent of single storied mature vegetation (for example, ponderosa pine) has decreased by over 1.6 million ha (4 million ac) (Hann and others 1997). Another way to view these changes is that the successional processes (movement from one successional stage to another) in some locations have been shortened by a factor of at least 10. For example, ponderosa pine may or may not be succeeded by Douglas-fir in 300 to 400 years within forests historically burned by frequent fires, but in many locations Douglas-fir has succeeded ponderosa pine in less than 50 years (fig. 6) (Hann and others 1997; Harvey and others 1999a; Smith and Arno 1999).

Forest Floor Changes

The shift in species composition from western white pine, western larch, and/or ponderosa pine to Douglas-fir, grand fir/white fir, and/or western hemlock dominated forests (including the shrub and forb components) has changed litter (soil surface) type and quantity from that which occurred historically. In addition, the accumulation of both above- and below-ground biomass from roots, needles, and boles in fir forests is accelerating activities of decomposers by increasing and changing the basic substrate they use (Harvey 1994). Associated with these changes in litter type and quantity is a likely change in ectomycorrhizal relationships and soil surface chemistry, including allelopathic substances, with the potential to alter a variety of microbial activities (Rose and others 1983). In addition, fire exclusion, timber harvesting, and animal grazing have exacerbated these forest floor alterations in many locales singly, or in combination, by soil compaction and displacement. For example, decomposed true firs create white rotten wood, which rapidly disperses into the soil and is quickly consumed by decomposers. In contrast, decomposed ponderosa pine, western white pine, and western larch create brown rotten wood, which can persist in soil for centuries and in this condition, can retain nutrients and hold water (Harvey and others 1988; Larsen and others 1980). Western larch and ponderosa pine tend to be deep-rooted, in contrast to the relatively shallow-rooted western hemlock and grand fir, which have abundant feeder roots and ectomycorrhizae in the shallow soil organic layers (Harvey and others 1987; Minore 1979). The soil microbial activities in fir-dominated forests compared to pine-dominated forests may diminish the post-fire acquisition and cycling of nutrients (Neary and others 1999). Moreover, these changes in soil microbial activities may increase the likelihood of uncoupling any continuity between current and preceding vegetative communities (Amarnathus and Perry 1994).

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 nutrients were dispersed in the mineral soils (Harvey and others 1999b). In the absence of fire, bark slough, needles, twigs, and small branches can accumulate on the forest floor and when these layers are continuously moist, ectomycorrhizae and fine roots of all species tend to concentrate in the surface mineral soil and thick organic layers, making them vulnerable to disturbances (Harvey and others 1994). In addition, historical ponderosa pine forests were likely well matched to soil resources, relatively resistant to detrimental fire effects, well adapted to wide ranges of site and short-term climate variation, subject to modest (largely beneficial) insect and pathogen mortality, and could be considered long-lived and relatively stable. In contrast, forests that were dominated by ponderosa pine and are now dominated by Douglas-fir, grand fir, or white fir are probably not well matched to soil resources and are also not likely resistant to the wide range of site and climate variations found within the dry forests (fig. 6). In turn, they are often subject to high insect and pathogen mortality and are no longer considered either long-lived or stable (Harvey and others 1999a).

Forests as Fuel

Fire behavior and burn severity depend on the properties of the various fuel (live and dead vegetation and detritus) strata and the continuity of those fuel strata, horizontally as well as vertically. The fire hazard for any particular forest stand or landscape relates to its potential for the fuels to cause specific types of fire behavior and effects. Understanding the structure of fuelbeds and their role in the initiation and propagation of fire is the key to developing effective fuel management strategies. Fuelbeds are classified in six strata:

1. tree canopy,
2. shrubs/small trees,
3. low vegetation,
4. woody fuels,
5. moss, lichens, and litter, and
6. ground fuels (in other words, humus, fermentation layer, surface and partially buried rotted wood, etc. (Sandberg and others 2001) (fig. 10).

Modification of any fuel stratum has implications for fire behavior, suppression, and burn severity.

Ground Fuels

Ground fuels consist of duff (organic soil horizons), roots, and buried woody material (fig.10) (Sandberg and others 2001). Often, needle fall and bark slough will accumulate at the base of trees and eventually create deep organic layers in which fine roots and ectomycorrhizae of trees and ground level vegetation may accumulate (Graham and others 2000). Ground fuels typically burn by smoldering and may burn for many hours, days, or even weeks if initial moisture contents are high (Frandsen 1991; Hungerford and others 1991) (fig. 11). This long duration smoldering can often lead to soil damage, tree mortality (high severity), and smoke (Ryan and Noste 1983; Ryan and Reinhardt 1988; Wells and others 1979). Rotten material on the ground surface is particularly ignitable by firebrands (small twig segments or bark flakes supporting glowing combustion) falling ahead of an advancing fire front and increases the success of spotting.

Surface Fuels and Ladder Fuels

Surface fuels consist of grasses, shrubs, litter, and woody material lying on, or in contact with, the ground surface (fig.10) (Sandberg and others 2001). The bulk density (weight within a given volume) of surface fuels and size class distribution of fine fuels (sticks less than 7.62 cm (3.0 in) are critical to frontal surface fire behavior (spread rate and intensity) compared to fuel loading (weight per unit area) alone. Other characteristics of surface fuels that determine surface fire behavior are fuel depth, continuity, and chemistry. Surface fires burn in both flaming and postfrontal (smoldering or glowing)

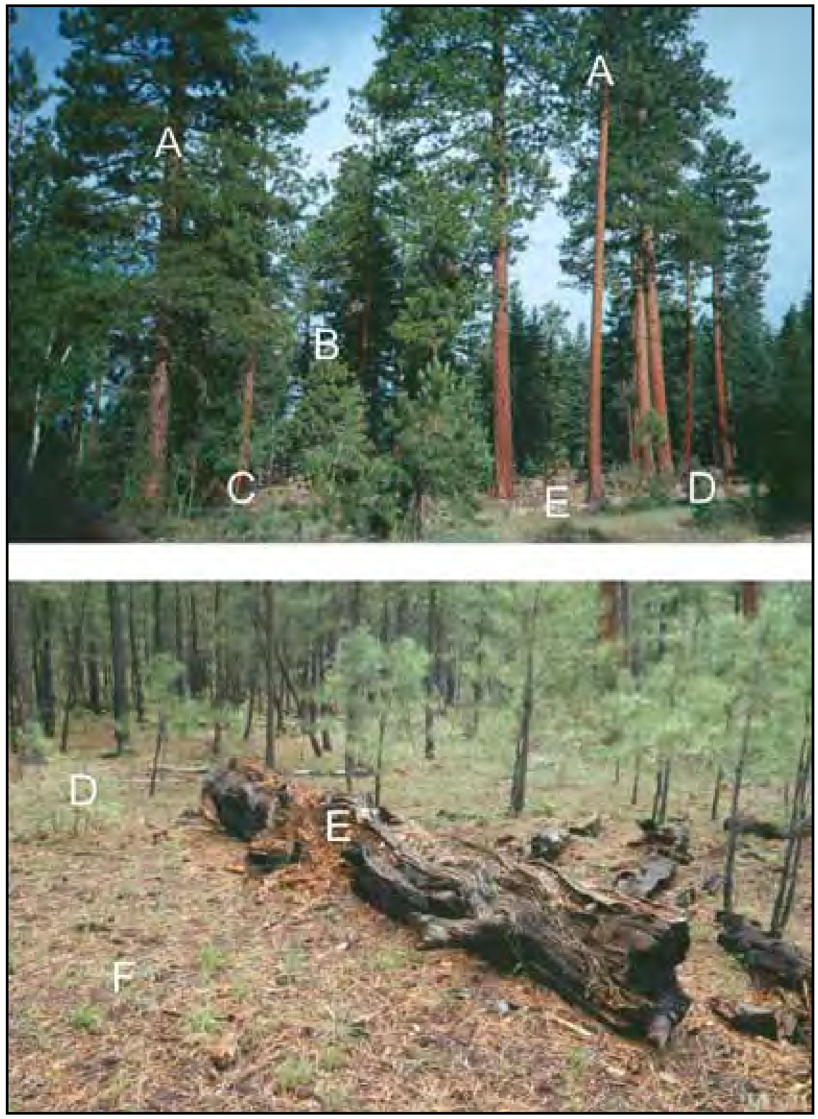


Figure 10. Fuelbed strata have different implications for combustion environment, fire propagation and spread, and fire effects. (A) The canopy, (B) ladder fuels and (C) shrub layers contribute to crown fires. (D) Low vegetation, (E) woody fuel, and (F) ground fuel contribute to surface fires. (E) Woody fuel and (F) ground fuels are most often associated with smoldering fires and residual combustion that can transfer large amounts of heat deep into the soil (Graham and others 2004; Sandburg and others 2001).

Figure 11. Large amounts of smoke can be produced from smoldering ground fuels after flames have subsided and large amounts of heat can be transferred to the soil.

Figure 12. Large amounts of fine fuels can be created when reducing canopy fuels. Without treatment, these fuels can increase the risk of unwanted wildfires and increase burn severity to both the high forest canopy and forest floor during prescribed fires and wildfires. Mechanical fuel treatments (for example piling and mastication) are often used to treat these fuel conditions.

phases (fig. 11). High-energy release rates occur during the relatively short flaming phase when fine fuels are consumed. Low energy release rates occur over longer periods by smoldering and glowing phases that consume larger (greater than 7.62 cm, 3 in diameter) fuels. Surface fuel complexes with high loadings of large material, such as slash left after timber harvesting or precommercial thinning operations (fig. 12), have long flaming residence times compared to fine fuels such as needles, shrubs, or grasses (fig. 5). High surface fire intensity usually increases the likelihood for igniting overstory canopy fuels, but surface fires with long residence times can contribute to drying aerial fuels that can lead to torching (when a tree or group of trees' foliage ignites and flares up, usually from bottom to top) (Alexander 1988).

Even in the dry forests, shrub and small tree regeneration can be abundant and frequent creating dense and robust layers of vegetation covering the forest floor (Pearson 1950) (fig. 5). In the moist and cold forests, tolerant tree and shrub regeneration is common even in forests with continuous canopy cover (Cooper and others 1991) (figs. 2 and 3). How these ground vegetative layers develop into mid-canopy layers depends on disturbance and how species differentiate as they develop based on their competitive and successional abilities (Oliver and Larson 1990). Stem differentiation can occur in even-aged, single species forests as stands self-thin because of inter-tree competition often in association with disturbances such as fires, diseases, and insects. Most often, these mid-canopy layers constitute the majority of the ladder fuels (Sandberg and others 2001).

Canopy Fuels

Crown fuels (also referred to as canopy fuels or aerial fuels) are those suspended above the ground in trees or vegetation (for example, vines, mosses, needles, branches etc.) (fig. 10). These fuels tend to consist mostly of live and fine materials less than 0.635 cm (0.25 in) in diameter. Crown fuels are the biomass available for a crown fire, which can ignite from a surface fire via understory shrubs and trees (ladder fuels) or from crown to crown. The shrub/small tree stratum is also involved in facilitating crown fires by increasing surface fireline intensity and serving as ladder fuels that provide continuity from the surface fuels to canopy fuels. These essentially bridge the vertical gap between surface and crown strata. The size of this gap is critical to ignition of crown fire from a surface fire below (Van Wagner 1977). Van Wagner (1977) identified two thresholds of crown fire activity: (1) crowns are ignited after the surface fire reaches critical fireline intensity relative to the height of the base of the aerial fuels in the crown; (2) crown ignition can become an "active" crown fire if its spread rate is high enough to surpass

Figure 13. Fires can progress from torching individual and groups of trees (A) until entire hillsides are a blaze (B) (K. Watenmaker photograph).

the second threshold based on the crown density (often referred as canopy bulk density, canopy weight for a given volume). Aerial fuels separated from surface fuels by large gaps are more difficult to ignite because of the distance above the surface fire, thus requiring higher intensity surface fires, surface fires of longer duration that dry the canopy before ignition, or mass ignition from spotting over a wide area (Byram 1966). However, once ignited, high-density canopy fuels are more likely to result in a spreading crown fire (active crown fire) than are low-density canopies (fig. 13).

The upper canopy is composed of leaves, branches, and boles of trees. Again, depending on the forest, its setting, and inherent disturbances, these layers may be simple and uniform such as those that occur in young (30 to 50 years) to mid-aged (80 to 120 years) early-seral species such as lodgepole pine and western larch. In contrast, dense and highly complex upper canopy layers often occur in late-seral, moist forests in which over five conifer species may occur (Haig and others 1941).

Wildfire

Several terms and concepts describe a wildfire, including fire intensity, fire behavior, fire severity, fire line intensity, burn severity, and fire effects (Scott and Reinhardt 2001; Peterson and others 2005). However, several of these terms are often used interchangeably and mean different things to different individuals and disciplines. Therefore, to further the understanding of wildfires and their effects, we suggest the circumstances concerning a wildfire be described as a continuum, beginning with the pre-fire environment, then fire environment, and finishing with the post-fire environment (Jain and others 2004).

Pre-Fire Environment

The pre-fire environment describes the condition of the forest before a fire occurs. It includes, but is not limited to, such descriptors as physical setting (for example, location, geology, soils, topography, landform, etc.), current vegetation, seral stage of vegetation, structure of vegetation, and fuel moisture content. Other information, such as the time since fuel treatment and time of year may also be included (Peterson and others 2005; Jain and Graham 2007) (figs. 4 through 6).

Fire Environment

The fire environment includes the state of the fuel, physical setting, weather (short- and long-term), and their interactions (Graham and others 2004; Jain and others 2004; Peterson and others 2005; Rothermel 1983). Weather characteristics, such as wind speed and direction, relative humidity, temperature, and atmospheric stability interact with physical attributes such as aspect, slope angle, topographic position, and landscape orientation. In turn these characteristics interact with the presence or absence of fuel (in other words, canopy, surface, or even buildings), their moisture contents, composition, and structure (for example, multiple canopies of conifers and deciduous trees and shake-roofed homes). As such, how a fire ultimately burns, and its effects are predicated on weather, and how the fuels are arranged within the physical setting (Finney 2001; Graham and others 2004; Peterson and others 2005; Scott and Reinhardt 2001) (fig. 10).

Fire intensity (fire line intensity) describes how a fire burns and the amount of energy it produces. Flame length, rate of spread, amount and location of torching, and spotting distance are frequently used to describe fire intensity (Jain and others 2004). The direct effects of a fire are often referred to as fire severity and/or as first-order fire effects. Fire severity is a function of how much heat a fire produces and the resulting consumption of plants and forest floor, heating of soils (which can volatize nutrients, create water repellent soils, distill organic matter, etc.), smoke production, and in some circumstances, the burning of homes (Cohen and Stratton 2003; Jain and others 2004; Ryan and Noste 1983; Turner and others 1999; Ulery and Graham 1993). For example, tree fire severity describes the amount of pre-fire tree crowns consumed, charred, or scorched, and the proportion of the tree cambium killed (Ryan and Reinhardt 1988). Similarly, how a community responds (for example, supportive, angry, frustrated) to fire suppression activities or smoke could also be a direct fire effect (Kent and others 2003).

Post-Fire Environment

The post-fire environment describes conditions when all the flames, smoldering, and heat are gone, or in simpler terms, what is left after the fire is out (fig. 14) (Jain and others 2004). Descriptors of the post-fire environment include, but are not limited to, the amount and condition of live and dead vegetation, condition of the forest floor and soils, and the state of the homes and buildings impacted by the fire (Cohen and Stratton 2003; Jain and others 2004; Peterson and Arbaugh 1986; Ryan and Noste 1983; Ryan and Reinhardt 1988; Skinner and Weatherspoon 1996). However, the post-fire environment does not describe what was consumed, nor does it describe the influence of the fire alone (Jain and others 2004). Instead it assumes that the pre-fire environment, fire environment, and combustion processes all contributed to creating the post-fire environment or burn severity (figs. 14 and 15).

Figure 14. A ponderosa pine stand burned by the highly intense Rodeo-Chediski Fire in Arizona that resulted in high tree burn severity (level 5) (Jain and Graham 2007).

Figure 15. Ground fires (especially those with long residence times) can severely burn soils, removing organic matter, volatilizing nutrients, killing tree roots, and creating water impermeable layers (soil burn severity level 6) (Jain and Graham 2007).

Burn Severity

How an ecosystem responds in the post-fire environment is often referred to as indirect or second-order fire effects (Reinhardt and others 2001). These effects include characteristics such as soil erosion and sedimentation of streams, the opening of serotinus lodgepole pine cones, colonization of a burned forest by woodpeckers, and the introduction of exotic plants. Burn severity may also include homeowners' response to burned houses or community reaction to post-fire rehabilitation activities (Kent and others 2003). These biological, physical, social, and economic responses to a fire vary over time and space and are interdependent. Therefore, we suggest the term burn severity best describes the post-fire environment because for a given fire intensity (behavior), depending on the pre-fire environment, a fire can create a variety of post-fire characteristics (Cohen and Stratton 2003; Graham and others 2000; Hungerford and others 1991; Reinhardt and others 1991; Robichaud and others 2003). For example, an intense canopy consuming wildfire can severely burn tree crowns but minimally affect the forest floor and surface soils. This wildfire outcome frequently occurs when the surface organic layers are too moist to burn, resulting in low soil burn severity. Similarly, a low intensity surface fire with long residence times can produce large amounts of heat that can alter mineral soil, kill vegetation, and predispose trees to insect attacks (figs. 11 and 15). As such, a low intensity fire can result in high burn severity. To this end, we have found that the relation between soil burn severity and tree burn severity is inconsistent. For example, tree death can be a function of soil burn severity, tree burn severity, or both severities combined. In addition, it appears that forest structure and composition can differentially influence both soil burn severity and tree burn severity.

Soil Burn Severity

We use six levels to classify soil burn severity that link fire intensity to the physical and biological responses to the fire (Jain and Graham 2007). The factors we chose to define the levels include: proportion of surface organic layers, mineral soil, and exposed rock present after a fire and the dominant char class defined as unburned and black char specific to both litter and mineral soil, and grey and orange char specific to mineral soil.

Level 1 soil burn severity occurs when surface organic horizons (litter, humus, rotten wood) cover approximately ≥85 percent of the forest floor after a fire, either in an unburned state or showing some evidence of black char (Debano and others 1998; Ryan and Noste 1983; Ulery and Graham 1993). This ≥85 percent threshold for organic horizons followed the soil quality standards used in Forest Service Regions 1, 2, 4, and 6 that define a 15 percent loss of soil organic horizons as detrimental to forest productivity (Page-Dumroese and others 2000). Not only do unburned surface organic layers maintain productivity they also provide refugia for microbes and other organisms (Hungerford and others 1991; Neary and others 1999).

Level 2 soil burn severity occurs when surface organic horizon cover is ≥40 and <85 percent of the forest floor, indicating minimal consumption occurred (Wells and others 1979). However, much of the mineral soil could be covered by black charcoal, and small and isolated gray to orange colored mineral soils may be present. Soils exhibiting this burn severity would contain nutrients in the surface horizons and microbes would be living in the humus and/or mineral soil layers (Hungerford and others 1991).

Level 3 soil burn severity occurs when forest floor conditions exhibit less than 40 percent surface organic horizons and black char dominates the site. Little or no areas are unburned, but a considerable amount of black, charred litter exists (Hungerford and others 1991; Neary and others 1999). Although some volatilization of soil nutrients would occur when large logs burned, many nutrients remain in the upper mineral soil and residual surface organic materials (organic cover tends toward 40 percent rather than 5 percent). If the pre-fire environment favors soil erosion (for example, slope angle, soil texture, parent material, etc.), this level of soil burn severity would indicate the potential for rill erosion; however, soil heating and duration may be insufficient to create hydrophobic soils (Debano 2000; Johansen and others 2001; Wondzell and King 2003).

Level 4 soil burn severity occurs when organic layers comprise less than 40 percent of the forest floor and gray or orange colored char dominates the mineral soil. The organic layers covering the site would be dispersed, and isolated amounts of surface organics and nitrogen volatilization would most likely occur. Refugia for organisms would be absent except in isolated areas, water repellent soil could exist, all areas would show some evidence of fire, and temperatures that create these types of conditions would have ranged from 20 to 400 ^{o}C (Debano 2000).

Level 5 soil burn severity occurs when no litter is present and black soil dominates the area. Depending on the site, potential soil erosion could be high and nitrogen in the surface organic horizons would likely be volatized, but nutrients in the mineral soil would likely remain (Hungerford and others 1991).

Level 6 soil burn severity describes a forest floor condition where no surface organics are present and gray to orange char dominates the mineral soil appearance. This soil burn severity indicates that minimal or if any nutrients or microorganisms are present in the soil surface (Debano 2000). Soil erosion and water repellent soils are possible and the abundance and presence of exposed rock may influence overland water flow (fig. 15).

Tree Burn Severity

Tree burn severity can be characterized two ways (Jain and Graham 2007). The first describes the vertical distribution of post-fire tree condition and is typically used to estimate post-fire tree mortality (Peterson 1985; Peterson and Arbough 1986; Wyant and others 1986). For example, ponderosa pines with 30 percent green crown have a 0.38 probability of dying after a fire and grand firs with similar crowns have a 0.87 probability of dying (Ryan and Reinhardt 1988). Secondly, we define horizontal tree burn severity as the proportion of trees and their condition remaining in a patch, stand, or landscape after a fire (Broncano and Retana 2004; Turner and others 1999) (fig. 14).

We identified five levels of tree burn severity. Because in most wildfires there are areas that do not burn, we defined a level 0 in our tree burn severity, as did several other authors (for example, Hutto 1995; Pollet and Omi 2002; Weatherspoon and Skinner 1995). Level 1 tree burn severity is defined as trees having greater than 60 percent of the residual crown remaining green. Defined horizontally, level 1 burn severity would have all trees within an aerial extent containing trees with greater than 60 percent green crowns. Mixed green burn severity (level 2) is highly variable with trees containing <30 percent green crowns to those having over 60 percent residual green crowns. Completely brown (scorched) trees can be present within level 2, but no trees with completely black crowns can occur. Typically, the site is dominated by trees containing some green crowns (Hutto 1995; Pausas and others 2003; Skinner and Weatherspoon 1996). Mixed brown crown severity (level 3) also contains green trees; however, trees with black crowns (needles consumed by the fire) occur as well as some trees with partially consumed crowns (brown needles and no needles present). In this situation, the abundance of green trees tends to be much less when compared to the abundance of completely dead trees (black or brown trees).

Level 4 tree burn severity occurs when the entire spatial extent contains trees without green needles and the majority are brown. However, within a given patch, some trees could be black (all needles consumed). When brown needles fall, they can decrease inter-rill and rill soil erosion, and because of the organic matter input, soil productivity can be enhanced (Harvey and others 1987; Pannkuk and Robichaud 2003). Level 5 tree burn severity occurs when an intense crown fire consumes all foliage (needles), leaving black branches, stems, and boles (figs. 13 and 14).

The tree and soil burn severity levels we defined are a series of classes that partition a continuum from everything remaining after a wildfire (as in the pre-fire state), to where all of the foliage and forest floor are consumed leaving blackened boles and bare soil. As such, the soil and tree burn severity we present is highly flexible allowing for the severity levels to be combined or kept separate depending on the data resolution required to meet a particular purpose or need.

Fuel Treatments

Crown fires are generally considered the primary threat to forests and human values; however; low intensity surface fires can degrade forest soils, kill trees, and burn domestic structures (Cohen and Stratton 2003; Graham 2003; Graham and others 2004) (fig. 11). In the moist, cold, and dry forests, crown fires are also the primary challenge for fire management. Our current understanding of fire behavior in most forests indicate that a crown fire begins with a transition from a surface fire to the ignition of the canopy. Therefore, crown fire development depends on the sequence of available fuels beginning with surface fuels, followed, in order, by woody fuel, low vegetation, shrubs, ladder fuels, and canopy fuels (figs. 5, 6 and 10).

There is a wide range of treatments that can be used to modify forest fuels (vegetation) that in turn influence both fire intensity (behavior) and burn severity (what is left). As a result, fuel treatments not only influence how a fire burns, but they can also influence how a fire affects water quality, timber quantities, wildlife habitat, scenery, and other forest elements that society values. Treating forests to produce commodities and values that society favors is not a new endeavor. Forest treatments that produced desired conditions over time existed by the mid-1600s (Evelyn 1664). These treatments have been improved over the years, especially the last 100 years, with continued research and experience in the practice and art of silviculture (Nyland 2002; Smith and others 1997).

In general, silviculture is the art and science of controlling the establishment, growth, competition, health, and quality of forests and woodlands to meet the diverse needs and values of landowners and society on a sustainable basis (Helms 1998). The plan and execution of forest and fuel treatments over time fall under the umbrella of a silvicultural system. There are not two or three systems, but rather an infinite number that can be developed and implemented over time and space integrating biology, management, and economic knowledge to treat forest fuels (Nyland 2002; Schlich 1906; Smith and others 1997). Even though silvicultural methods historically stressed the production of wood crops, treatments can be designed to create and maintain a variety of forest compositions and structures relevant and effective for modifying fire intensity and burn severity. The challenge is to develop systems to manage forests as fuel yet create variable and complex forest structures and compositions that address other values such as maintaining the sense of place or maintaining wildlife habitat (Graham and Jain 2004).

The most effective strategy for reducing crown fire occurrence and tree and soil burn severity is to:

1. reduce surface fuels,
2. increase height to live crown,
3. reduce continuity of the forest canopy, and
4. reduce canopy bulk density (Cruz and others 2002; Graham and others 1999a; Scott and Reinhardt 2001; Van Wagner 1977) (fig. 10).

The documentation of treatments in a fuel treatment strategy is written in a silvicultural prescription that describes the treatments and the resulting composition and structure of dead and live vegetation through time.

Forest Floor Treatments

Forest floor treatments, possibly more than any other, influence how forest vegetation establishes and develops and are major determinants of both fire intensity and burn severity. No matter what method is used to treat the forest floor, a key to treatment success is ensuring that highly diverse (for example, diverse amounts of CWD, bare soil, litter depths and compositions etc.) forest floor conditions remain after a treatment to facilitate forest recovery after a wildfire. By doing so, a fire burning these fuels would burn heterogeneously and result in soil burn severity dominated by levels 1 through 4.

Mechanical Treatments

Machines through their proper use can create and maintain desired forest floor conditions in many forest settings. However, if not used properly they can compact and/or displace mineral soils and reduce soil organic content (Harvey and others 1996; Page-Dumroese and others 1997). As such, they are usually limited to operating on gentle slopes (approximately <40 percent). Machines capable of separating slash of different sizes, often called grapple machines, displace less soil than rakes attached to the front of tractors, which in turn conserve and protect the soil surface layers (fig. 16). After piling the slash, although usually burned, piles can also be chipped, used for bioenergy, or used for domestic firewood.

Machines can rearrange, compact, chip, masticate, or otherwise change the fire hazard without reducing the fuel loads (fig. 17). The effects such mastication and similar machines have depends on the size, composition, and location of the residual fuels they leave (Graham and others 2000). For example, thin layers of wood chips spread on the forest floor tend to dry and rewet readily, and deep layers of chips and chip piles may have insufficient air circulation creating poor decomposition conditions. In addition, these layers of small woody material on the forest floor can insulate the soils and when decomposition does occur, the decomposing organisms utilize large amounts of nitrogen, which reduces its availability to plants. Therefore, use of any of these crushing, chipping, or mulching treatments needs to consider the impact on the decomposition processes and the potential contribution to smoldering fires.

Figure 16. Grapple machines have the ability to separate coarse from fine fuels more readily than tractors and tend to displace and compact soils less than tractors.

Figure 17. Masticator used to treat ladder and surface fuels in the moist forests near Lake Tahoe, California.

Prescribed Fire

Prescribed fire is commonly used to treat and manage forest floor conditions throughout the western United States. Fire can reduce fine fuels, reduce ground-level vegetation, preserve surface organic layers, and maintain appropriate amounts of coarse woody debris (fig. 18). However, unless fuel and weather conditions at the time of ignition are appropriate, fire can also create conditions adverse to vegetative development and impair soil productivity (Debano 1991; Hungerford and others 1991). The amount of forest floor consumed by a fire is dependent on its moisture content, particularly in the lower humus and fermentation layers. Within the Inland West, when the moisture content of these lower layers exceeds 100 percent when a fire occurs, the majority of these layers are generally conserved (Ryan 1982). When burning occurs under these conditions, nutrients (P, N, K) in the litter and fine fuels (≤7.6 cm, 3 in) have the opportunity to condense in the lower layers (for example humus) and, therefore, are not lost from the site (Harvey and others 1989) (soil burn severity less than or equal to level 3).

Figure 18. (A) Fire being used to decrease the amount of organic material that developed, most likely because of fire exclusion, at the base of this large ponderosa pine located in southern Idaho. Fire was applied early in the spring when the temperature of the lower organic layers was below 4.4 °C (40 °F) (when fine root activity is minimal) and when their moisture contents exceeded 100 percent. (B) The application of a low intensity prescribed fire treating the entire free selection area after three spring "snow well" treatments used to reduce the organic layers at the base of the trees.

Some ground-level vegetation responds vigorously to heat, such as *Ceanothus* spp., which has seeds buried in the forest floor. In addition, many ground-level species sprout aggressively in response to fire (fig. 19). The amount, kind, season, and type of fire used to treat the forest floor can create a particular soil burn severity. Therefore, the relation between the desired soil burn severity and the expected response of vegetation for a given forest, PVT, and biophysical setting can be highly variable (Baumgartner and others 1986, 1989; Bradley and others 1992a, 1992b).

Prescribed fire can reduce horizontal fuel continuity (shrub, low vegetation, woody fuel strata), which in turn disrupts growth of surface fires, limits their intensity, and reduces the potential of spot fire ignition. In addition, by reducing fine fuels, duff, large woody fuels, and rotten material, their continuity changes the fuel energy stored on the site and potentially reduces both fire intensity and burn severity (figs. 13, 14 and 15). Also, prescribed fire can directly consume low ladder fuels (shrubs, dead trees, needle drape, and small trees) and scorch and kill the lower branches of overstory trees, effectively raising the live crown above the ground surface.

Prescribed fire is an excellent tool for treating the forest floor; however, there are also considerable risks and uncertainties with its use (Biswell and others 1973; Cooper 1960; Fernandes and Botelho 2003; Weaver 1955, 1957). For example, fire cannot readily create precise stand structures and compositions compared to controlled mechanical treatments. Climatic and fuel moisture conditions can severely restrict when fire can be used, especially in forests with large amounts of fuel or located in areas sensitive to smoke production. Also, fires may escape and cause unintended resource and economic damage. Even with these short-comings, prescribed fire, by influencing multiple fuels, can effectively modify both fire behavior and burn severity.

Ladder Fuel Treatments

Precommercial thinning and other intermediate forest treatments can be designed to target specific fuel strata and disrupt, (1) the vertical progression of fire from surface

Figure 19. Ponderosa pine saplings growing among *ceanothus*, a ground-level shrub that responds aggressively in response to fire. The clearcut is located on a grand fir potential vegetation type in eastern Oregon.

fuels to ladder fuels to canopy fuels and (2) the horizontal progression of fire through individual fuel strata. These treatments can be designed and applied to reduce both the continuity and density of shrubs and trees and disrupt fire spread, and prevent the fire from producing sufficient heat to detrimentally affect the surface soils (Biswell 1960; Biswell and others 1973; Cooper 1960; Fisher 1988; Martin and others 1989; Pollet and Omi 2002; Scott 1998a; 1998b; Scott and Reinhardt 2001; Weaver 1955). Depending on the forest and its condition, these tending activities can occur at a variety of time intervals and intensities, thereby creating an infinite number of stand structures and compositions (Graham and others 1999a) (figs. 17 and 20).

Treatments emphasizing the smaller trees and shrubs (ladder fuels) can effectively reduce vertical fuel continuity that fosters crown fire initiation (fig. 21). During this period in the life of a forest, the structure and composition are the most plastic (in other words, responsive to treatments) and future stand dynamics are largely determined (Graham 1988; Haig and others 1941; Pearson 1950). In addition, thinning small material and pruning branches are more precise methods than prescribed fire for targeting ladder fuels and specific fuel components in the ladder-fuel stratum. The net effect of removing ladder fuels is that surface fires burning through treated stands are less likely to produce enough energy to ignite the overstory fuels.

The issue in many forests is excessive regeneration, which makes weeding and cleaning young stands essential to develop fire resilient forests (Helms 1998; Smith and others 1997). For example, in the ponderosa pine forests of the Black Hills (northeastern Wyoming and western South Dakota), ponderosa pine regeneration is often prolific, creating a fire hazard and compromising stand development (Shepperd and Battaglia 2002) (fig. 22). Similarly, in both the cold and moist forests, abundant regeneration is often the norm; again requiring tending operations to develop stand characteristics that reduce fire intensity and burn severity (Deitschman and Pfister 1973; Graham 1988; Haig and others 1941; Johnstone 1985; Johnstone and Cole 1988) (fig. 23). Such ladder fuel treatments can be most often accomplished mechanically and, in rare situations, fire can be used (Saveland and Neuenschwander 1988).

Canopy Treatments

Classically, the term thinning was applied to stand treatments aimed at redistributing growth on remaining stems, but often, any kind of partial cutting (tending) such as

Figure 20. A thinned mixed conifer stand located in northern Idaho.

Figure 21. The first entry of a free selection system being accomplished using a masticator. Depending on stand conditions, the majority of the ladder fuels and small trees (less than 20 cm, 8 in) were chunked, leaving a clumpy high forest canopy. The objective of the system is to create and maintain a wildfire resilient ponderosa pine forest located in southern Idaho, similar to those that occurred historically.

Figure 22. Ponderosa pines in the Black Hills of western South Dakota tend to regenerate prolifically using shelterwood regeneration methods.

Figure 23. In moist forests, western hemlock (a late-seral species) readily regenerates in relatively closed canopy conditions to create multiple fuel layers.

liberation, preparatory, improvement, sanitation, and selection cuttings could be termed thinning (Graham and others 1999a). Thinnings can be designed to affect canopy bulk density, which in turn determines whether a crown fire could be sustained.

Low thinning occurs when trees are removed from the lower canopy, leaving large trees to occupy the site. This method mimics mortality caused by inter-tree competition or surface fires, and primarily removes small and suppressed trees (ladder fuels). Crown and selection thinnings can be used to reduce canopy density and continuity within the main forest canopy and alter forest composition (Nyland 2002) (fig. 20). Usually, different tree species have characteristic development rates that result in individual species dominating specific canopy layers. For example, in many dry forests, ponderosa pine primarily occupies the dominant canopy layers, whereas shade-tolerant grand fir, white fir, or Douglas-fir occupy the intermediate and suppressed layers (fig. 6). In this situation, low thinning favors the development of the dominant and codominant ponderosa pine, often a desirable forest fuel condition. Thinnings can remove few to many trees and need not create regularly spaced forests, but the number, clumping, and juxtaposition of residual trees can be varied (Long and Smith 2000; Nyland 2002; Reynolds and others 1992). As such, thinnings can precisely create targeted stand structures and compositions that will influence both fire intensity and burn severity (Agee and others 2000; Miller and Urban 2000; Stephens 1998; van Wagtendonk 1996; Weatherspoon and Skinner 1996).

A clearcut may be an appropriate fuel treatment, especially for fuel breaks or in other areas in which the spatial continuity of fuels needs to be drastically decreased (Agee and others 2000; Graham and others 2004). Disposal of hazard fuels (in other words, activity fuels, tree limbs, and boles less than 7.6 cm, 3 in diameter etc.) can occur by using prescribed fire and/or mechanical means. This amount of canopy and fuel removal will have the greatest impact on watershed, wildlife, and wildfire responses. Clearcuts can be modified by not removing all trees but retaining a few to many. Traditionally for timber production, seed-tree and shelterwood regeneration methods created such conditions. This condition might be similar to those created by a mixed severity fire (Graham and Jain 2005; Graham and others 2005). The retained trees will modify a site's environment, influencing fuel moistures and how fires would burn. The number, location, juxtaposition, disposition (longevity), and species of the residual trees can be specified. By doing so, a variety of forest structures and compositions and landscapes can be developed (Long and Smith 2000; Reynolds and others 1992).

Selection systems are another set of forest treatments that have applicability for treating all forest (fuel) layers (Graham and Jain 2005; Graham and others 1999b; Graham and Smith 1983; Long and Smith 2000; Marquis 1978; Nyland 2002; Reynolds and others 1992; Smith and others 1997). Depending on the design, they can maintain high forest cover, which is an important component to many current management objectives. Individual tree, group, and free (irregular) selection systems have been described. As traditionally applied, they maintain uneven-aged (diameter distributions) forest structures by planning for and executing frequent (in other words, 10- to 20-year) entries (treatments). However, by varying the opening sizes, canopy gaps, tree clumping, species preferences, and their juxtapositions, highly heterogeneous forest structures and compositions can be created and maintained (Jain and others 2008). An integral part of all selection systems is the tending of all canopy (fuel) layers over time and space to ensure the desired conditions are created and maintained (Graham 1990; Graham and others 1999b). This is of particular importance in treating the ladder fuels and multiple canopies using selection systems. Canopy gaps, highly variable tree arrangements, and forest floor treatments can be used in concert, as to ensure the lethal fire risk is not exacerbated (Graham and Jain 2005) (fig. 18).

Fuel treatments can increase the probability of modifying fire behavior and burn severity during most weather conditions. However, extreme weather conditions (low fuel moisture contents, low humidity, high winds) can create fire behavior that can burn through or breach most fuel treatments (Finney and others 2003) (fig. 24). A realistic objective of fuel treatments is to reduce the likelihood of crown fire and severely burned soils that would lead to a loss in value or undesirable future conditions and not necessarily guarantee crown fire elimination.

Figure 24. The intense Rodeo-Chediski Fire that burned in Arizona in 2002. This ponderosa pine forest was historically burned by low intensity surface fires, but singly and in combination, fire exclusion, timber harvest, climate change, and livestock grazing contributed to forest changes that facilitated this uncharacteristically intense fire. Often, fuel treatments will locally impact such fires but most treatments are readily breached.

Canopy Treatments Combined With Forest Floor Treatments

The most effective and appropriate sequence of fuel treatments depends on the amount of surface fuel present; the density of understory and mid-canopy trees (Fitzgerald 2002); long-term potential effects of fuel treatments on vegetation, soils, and wildlife; and short-term potential effects on smoke production (Huff and others 1995). In forests that have not experienced fire for many decades, multiple fuel treatments are often required to achieve the desired fuel conditions. Canopy treatments, followed by prescribed burning, reduce canopy, ladder, and surface fuels, thereby providing maximum protection from intense fires in the future (Peterson and others 2005). Potential fire intensity and/or burn severity in treated stands is significantly reduced only if canopy treatments are accompanied by reducing the surface fuels (woody fuel stratum) created from the thinning operations (Alexander and Yancik 1977; Graham and others 1999a) (fig. 18). Given current accumulations of fuels in some stands, multiple prescribed fires—as the sole treatment or in combination with thinning—may initially be needed, followed by long-term maintenance burning or other fuel reduction (for example, mowing, mastication, scarification, etc.) to reduce crown fire hazard and the likelihood of high burn severity (Jain and Graham 2004; Peterson and others 2005). (see Chapter 4).

The most appropriate fuel treatment strategy is often thinning (removing ladder fuels and decreasing tree crown density), followed by prescribed fire, piling and burning, mastication of fuels, or other mechanical treatments that reduce surface fuel amounts and often promote decomposition (figs. 18 and 21). This approach reduces canopy, ladder, and surface fuels, thereby reducing both the intensity and burn severity of potential wildfires. Restoring forests to a condition in which fire alone can maintain the desired conditions will take time (fig. 18). Wildland fire use (that is, allowing certain wildfires to burn under certain conditions and locations) offers some hope once homes, communities, and key resources are protected through thinning, prescribed fires, or other treatments.

Post Treatment Environment

Thinning and prescribed fires can modify understory microclimate that was previously buffered by overstory vegetation (Scott and Reinhardt 2001; Pollet and Omi 2002; Weatherspoon 1996). Thinned stands (open tree canopies) allow incoming solar radiation to penetrate to the forest floor increasing surface temperatures, decreasing fine fuel moisture, and decreasing relative humidity compared to unthinned stands—conditions that facilitate intense fires (Countryman 1955; Pollet and Omi 2002). An increase in surface fire intensity may increase the probability of a fire exceeding the critical threshold needed to initiate a crown fire (Van Wagner 1977). Therefore, it is important that the gap between the surface and crown fuels is increased by either prescribed fire or pruning (fig. 18) so if a fire should occur, the potential for crown fire initiation is minimized. Rothermel (1983) found significant differences in fire behavior between a closed stand (no harvest) and an open stand (thinned). Changing stand structure, while ignoring surface fuels, will only affect the likelihood of active crown fires—it will not necessarily reduce the likelihood of surface fires intense enough to damage soils or cause significant overstory mortality. For example, figure 25 illustrates the build-up of organic layers as a function of fire exclusion. If only ladder and crown fuels are treated ignoring the ground fuels, in this situation, a fire would most likely lead to high soil burn severity and killing of trees through cambium kill. Moreover, surface fire intensity may be greater in thinned stands compared to untreated stands depending on whether the thinning activity adds to the surface fuels (Alexander and Yancik 1977; van Wagtendonk 1996; Stephens 1998). Therefore, it cannot be emphasized enough that all fuel layers need to be managed (over time and space) to minimize the unwanted consequences of wildfires (fig. 10).

Treatment Longevity

Very few specific experiments have evaluated the longevity of treatments and their effectiveness in altering fire behavior and burn severity. However, there is considerable information on forest growth and development, and this information is useful in providing estimates on the longevity of potential treatments. There are good models available (Reinhardt and Crookston 2003; Wykoff and others 1982) that predict vegetation development over time and provide estimates of treatment longevity.

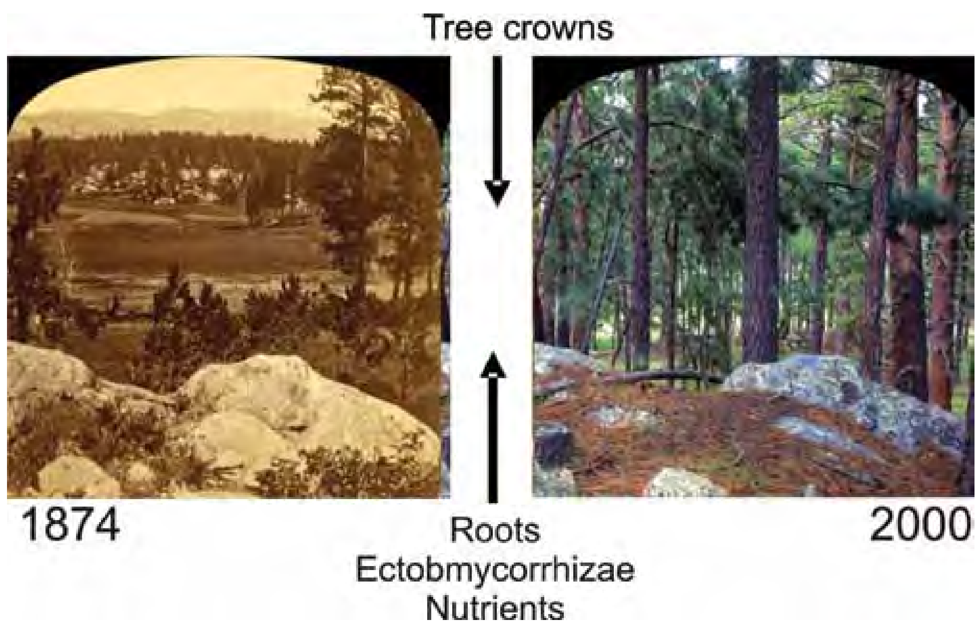


Figure 25. When fires are excluded from ponderosa pine forests, organic layers tend to accumulate and tree roots, ectomycorrhizae, and nutrients also tend to concentrate in these layers. Note the contrast between the amount of organic material on the forest floor when General Custer came through the Back Hills in 1874 and the amount that has accumulated around the rocks in the photo in 2000. Photograph courtesy of Paul Horsted/custertrail.com (Grafe and Horsted 2002).

There are limited amounts of information on the relation between canopy structure and ground-level vegetation, or the relation between vegetation development and fuel moisture. For short time periods (months) after treatment, fuel changes can produce dramatic differences in fire behavior. Biswell and others (1973) showed that the effectiveness of prescribed fire treatments in maintaining desired fuel conditions decreased significantly over 2 decades in a ponderosa pine forest. Van Wagtendonk and Sydoriak (1987) directly examined fuel accumulation following prescribed burning and found that fuel amounts reached 67 percent of their pre-burn loading after 7 years. Many of the prescribed fires they used were the first fuel treatments that occurred in these stands in decades and would potentially kill many small trees that would contribute to the woody fuel load. Repeated burns were not studied, but the elimination of small trees using a series of burns would be expected to retard fuel accumulation compared to the amounts they reported. Van Wagtendonk and Sydoriak (1987) concluded that prescribed burning would be required at least every 11 years to maintain fuel loads below their preburn condition. Van Wagtendonk (1995) also reported reductions in fire spread and intensity of fires up to 14 years after previous burns within the mosaic of large fires in the mixed-conifer forests of Yosemite National Park.

The duration of treatment effectiveness will vary with climate, PVT, soils, and other factors that influence productivity and the nature of the fuel treatments (Keyes and O'Hara 2002). For example, the longevity of thinning slash is greater on drier sites, particularly for finer woody material compared to fine fuels occurring in wetter forests (Christiansen and Pickford 1991). Treatment effects will likely last longer in areas in which vegetation development is slower than in highly productive areas where vegetation development is more rapid and lush (Weatherspoon and Skinner 1996). Few data exist, but inferences from fire history research and modeling show that the length of treatment effectiveness will vary with forest type (general fuel characteristics) and fire regime (Heyerdahl and others 2001; Miller and Urban 1999; Taylor and Skinner 1998; Taylor and Skinner 2003).

Fuel Treatment on Landscapes

The spatial patterns of fuel treatments in landscapes will most likely determine their effectiveness in modifying wildfire behavior (Hessburg and others 2000), because multiple stands and fuel conditions are involved in large fires (Finney 2001). Fire behavior under extreme fire weather may involve large areas of fuels, multiple fires, and spotting, so a "firesafe" landscape needs to populate hundreds to thousands of hectares (acres) with strategically located fuel treatments (Finney 2003). Treating small or isolated stands without assessing the broader landscape will most likely be ineffective in reducing wildfire extent and severity.

There are limited examples of how fuel treatments have altered subsequent fire behavior and burn severity (Helms 1979; Martin and others 1989; Pollet and Omi 2002). However, despite small-scale modification of fire behavior, none of these studies demonstrated that spread or behavior of a large fire was significantly altered, probably because the units were relatively small and were surrounded by areas containing vegetation favoring continued fire growth. In the mixed-conifer forests of northern California, fire intensity varied with dominance of short-needle or long-needle conifers in the same fire regime (frequent, low-moderately intense surface fires). Under similar burning conditions in a retrospective study of the widespread fires of 1987, stands dominated by Douglas-fir sustained significantly less damage than did stands dominated by ponderosa pine (Weatherspoon and Skinner 1995). Given current fuel accumulations across the Interior West, small areas (unknown threshold) favoring low intensity fires will probably be irrelevant to fire behavior (Dunn 1989; Salazar and Gonzalez-Caban 1987). Therefore, treatments that alter vegetation to favor low intensity fires must consider spatial arrangement of fuel structures to alter wildfire behavior.

Large-scale, frequent mosaic burning may maintain many portions of some landscapes in a treated condition and disrupt growth of the inevitable wildfire (Brackebusch 1973). Evidence that mosaic patterns reduce fire spread comes from natural fire patterns that have fragmented fuels across landscapes. This spatial pattern produces self-limiting fire growth and behavior by management of natural ignitions, as shown in Yosemite National Park and Sequoia National Park (van Wagtendonk 1995, 1996). The spatial arrangement of vegetation influences the growth of large fires (Brackebusch 1973; Finney 2001). Patches of vegetation that burn relatively slower or less intensely than surrounding patches may force the fire to move around them by flanking (at a lower intensity), which locally delays the forward progress of the fire. Such strategically placed treatments create landscape fuel patterns that collectively slow fire growth and modify behavior while minimizing the amount of treated area required (Finney 2001) (fig. 26). The importance of spatial pattern is emphasized by findings that random fuel treatment arrangements (Finney 2003) are extremely inefficient in changing fire behavior (fig. 26)—requiring perhaps 50 to 60 percent of the area to be treated compared to 20 percent in a strategic fashion (Finney 2001). If fuel treatments are to be effective at changing the growth of large fires, then strategic placement of treatment areas must incorporate land ownership, endangered species, riparian buffers, and other concerns. The costs and maintenance levels needed to maintain this forest pattern would vary depending on forest type, PVT, access, and public acceptance.

An alternative to a landscape approach to altering fuels is to create fuel breaks, which modify easily accessible portions of the vegetation in strategic locations across a landscape (Agee and others 2000; Weatherspoon and Skinner 1996). The purpose of fuel breaks, which are typically placed in defensible locations like a ridge, is to aid suppression efforts of firefighters to stop fire spread (Green 1977). The benefits of a fuel break are successful only if the fire suppression activities anchored to the fuel break limit the size or perimeter of the fire. If fire suppression does not occur, a fire can continue to burn through the fuel break with little or no effect on fire size. Moreover, fuel breaks (clearcuts) most often require long-term maintenance and repeated treatments.

Treatment Efficacy

Fire behavior and burn severity (see section on burn severity, page 42) are strongly influenced by stand structure as it relates to live and dead fuel loadings and ladder fuels. The type and abundance of surface fuels have an effect on the abundance of falling embers, which can ignite distant fuels and spread, thus influencing fire behavior. Reducing both ladder fuels and surface fuels is essential to effectively change fire behavior and burn severity (Graham and others 1999a; Omi and Kalabokidis 1991; Pollet and Omi 2002). Examples from the Hayman Fire in Colorado illustrate these interactions (Finney and others 2003). The Polhemus prescribed burn in November 2001 removed most surface fuels and pruned lower live branches from trees in a ponderosa pine forest but did

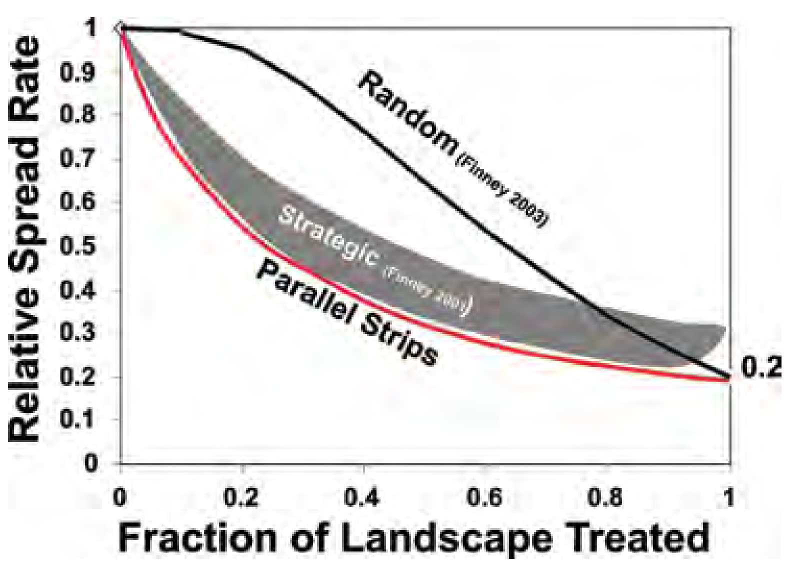


Figure 26. Comparison of large fire growth rate among different spatial fuel patterns (Finney 2003).

not significantly reduce overstory density (tree burn severity less than or equal to level 2). These changes were sufficient to stop the Hayman Fire when it burned into the area in June 2002 even though intense fire behavior was present, facilitated by high winds (48 kmph, 30 mph and greater) and low relative humidities (near or below 10 percent). This treatment was applied within a few months of the fire, thus decreasing the surface fuels substantially (soil burn severity less than or equal to level 3). In this case, the time since treatment, plus the treatment contributed to the change in fire behavior and subsequent burn severity. On the Manitou Experimental Forest (Hayman Fire), mechanical harvesting (selection silvicultural system) reduced the density of all sizes of trees in a pure ponderosa pine forest and concentrated logging slash in large piles. These actions resulted in an easily suppressed surface fire when the Hayman Fire burned into the area. On the other hand, all trees were killed in the Sheepnose Fuels Reduction Project (shelterwood) within the Hayman Fire. Although the stand was heavily thinned from below, heavy surface fuels from non-merchantable logging slash allowed the fire to burn intensely through this stand, potentially damaging the soils and scorching and killing the trees left after the treatment (Finney and others 2003) (fig. 27).

Another example of fuel treatment effectiveness is the Cone Fire (September 2002) in northern California that burned into the Blacks Mountain Experimental Forest. The fire burned approximately 809 ha (2,000 ac) of a study area that was designed to evaluate the effect of varying forest structures (thinnings) on wildlife. When the fire encountered forest structures in which the surface fuels had been burned and the canopy density was reduced, the fire dropped from a crown to a surface fire within the first few yards of entering the treatment units. In areas where the surface fuels were not treated, the fire continued through the unit as a surface fire with variable intensity. There was considerable crown scorch and bark charring in these treatment units with areas of up to 1 ha (2.47 ac) where all trees were dead (Graham and others 2004). These examples show that variability in weather, physical setting, and forest fuels (composition, structure,

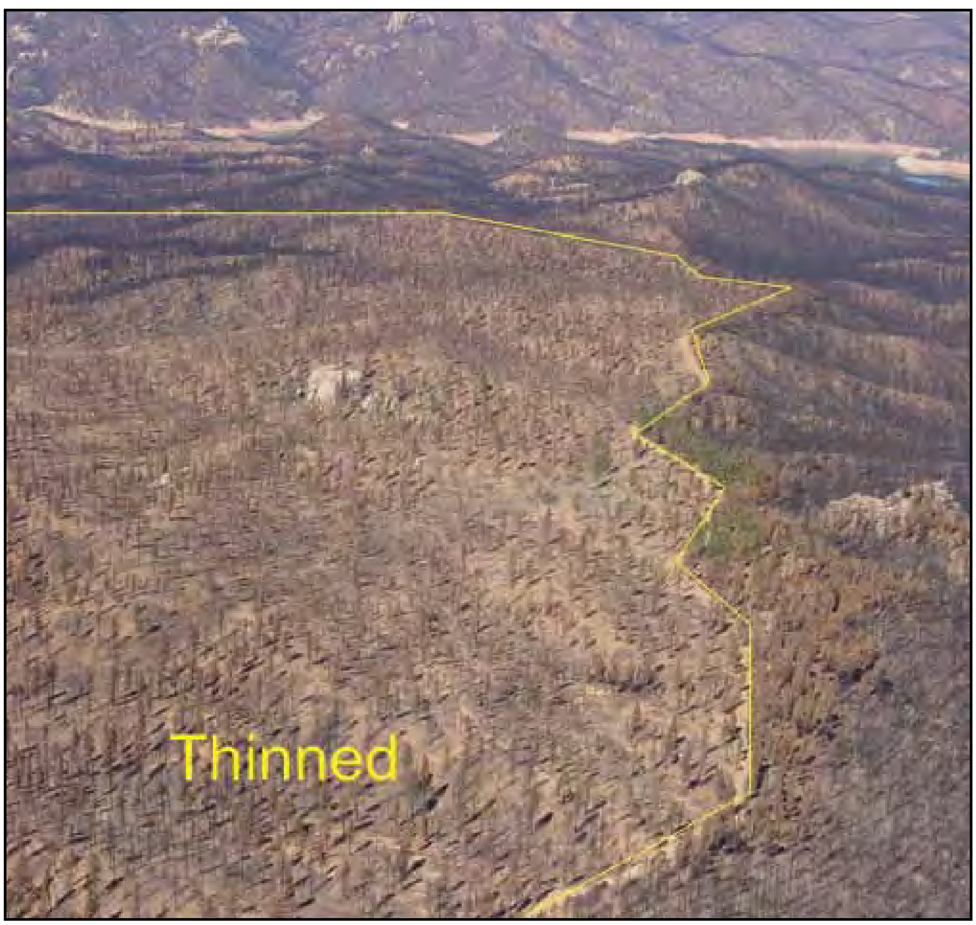


Figure 27. A thinned ponderosa pine stand in which the surface fuels remained after harvest, which resulted in an intense surface fire (tree burn severity level 4) and high soil burn severity (level 5 to 6) (Jain and Graham 2007) when the Hayman Fire burned near Colorado Springs, CO, in 2003.

etc.), influences fire behavior and effects, making it difficult to generalize the effects of treating forests to alter fire behavior and burn severity. However, a key point from these examples is that in many cases, particularly if combined with fire suppression efforts, reduced surface fuels and thinning can significantly limit fire spread and influence burn severity to both vegetation and soils.

Fuel Treatments and Cumulative Watershed Effects

The cumulative effects (either positive or negative) of a fuel treatment are the environmental consequences of the activity when added to the existing landscape condition and any reasonably foreseeable future actions or disturbances. An environmental consequence has both spatial and temporal dimensions and short-term and long-term effects (Reid 1988). Fuel treatments should cumulatively influence vegetation (forest composition and structure function) and be considered as wildfire fuel (Finney 2003; Graham and others 1999a; Graham and others 2004). The cumulative impacts that fuel treatments have on burn severity and fire behavior can range from local to the landscape and watershed levels. As Finney (2003) suggests, fuel treatments can be strategically located and designed for disrupting a fire's progression as it burns through watersheds and landscapes. The cumulative impact of such treatments on potential fire behavior and burn severity would depend on how much of a watershed or landscape was treated, the treatment locations, the timing of the treatments, the kind and intensity of the treatment, and the length of time before both dead and live vegetation would return to pretreatment conditions (for example, the longevity of the fuel treatments and their impact on the fuels complex). In addition, the physical character of the watershed (in other words, slopes, aspect, geology, soils, orientation, and elevation), forest type, PVT, structural stages, seral stages, and patch sizes and juxtapositions would interact with current, planned, and continued maintenance fuel treatments to influence wildfire intensity and burn severity.

Similar to targeting the vertical distribution of fuels to modify fire outcomes, treating different fuel strata would most likely differentially affect an array of wildlife species that are dependent on the vertical forest strata for habitat or habitats of their prey (Reynolds and others 1992; Theobald and others1997; Thomas 1979). Fuel treatments that affect the accumulation and disposition of snags and coarse woody debris, and the retention, disposition, juxtaposition, size, and amount of canopy cover, seral stages, and structural stages occurring on a site, would most likely impact wildlife (Reynolds and others 1992; Thomas and others 1979). These wildlife implications would be important to current planning and the execution of fuel treatments, but future and maintenance treatments would need to include their potential impact on wildlife. Also, depending on the wildlife species and the extent, number, and location of fuel treatment, their effects would most likely be cumulative and could encompass watersheds and large landscapes (Thera and Wildman 2001). Canopy cover also affects water relations within and across watersheds similarly to the potential impacts it has on wildlife. Through evapotransporation, shading, snow interception, snow retention, and other watershed impacts, canopy cover can affect watershed stability (Ziemer and Lisle 1998).

Canopy cover can influence watershed processes (for example, sedimentation and peak flow) through altering rainfall intensity on established snow packs and raindrop intensity on the forest floor; however, these effects must be placed with the context of the soil type, geology, and other biophysical characteristics to understand the cumulative impacts to water quality and quantity. Even though they are not part of fuel treatments, present and future roads are integral to forest management actions. Their location and use for managing fuels inherently affects watersheds and the cumulative effects of fuel treatments on watersheds (Berg 1989; Elliot 2000) (see Chapter 5).

The effect fuel treatments have on visual quality is often an important component of a cumulative effects analysis. Fuel treatment appearance and juxtaposition within a landscape are integral to their visual effects (Bergen and others 1995). The visual quality of a watershed is highly intertwined with the road network along with the character (in other words, prescribed fire, piling slash, timber harvest, etc.) of the treatments and their visual quality. Canopy closure—its shape, location, size, and other attributes—has significant impact on visual quality, both in the short- and long-term, and the cumulative visual impact of such treatments (Brown and Daniel 1986). In addition, the visual quality attributes of a fuel treatment are predicated by whether the treatments are viewed from the foreground, middle ground, or the background.

Fuel treatments should treat forests using silvicultural systems. The silvicultural system documented in a silviculture prescription can disclose the cumulative effects fuel treatments have on vegetation regeneration and development. In particular, silvicultural systems designed for treating fuels should integrate the cumulative and interactive role that insects, diseases, and wildfires play in forest development. Moreover, by designing a silvicultural system and documenting it in a silvicultural prescription, the prescription could display the dynamics of a forest and all of its components over time, which can facilitate the understanding of the cumulative effects of fuel treatments among a wide array of disciplines and stakeholders. By disclosing the effects in a prescription, the effects of fuel treatments on the sustainability of a forest, along with their risks and uncertainties, are identified and documented. In addition, silvicultural systems and their documentation provide a framework for understanding cumulative watershed effects and can be developed into visualizations and other communication tools applicable to a wide range of disciplines and stakeholders.

Conclusion

The cold, moist, and dry forests are all inherently different (figs. 1 through 5). Each forest has a unique suite of forest vegetation, seral and structural stages, and compositions. With this uniqueness comes distinctiveness in where wildfires burn, what they burn, and the effects they have as well as the effect and latitude of forest treatments that modify fire behavior and burn severity. The fuels that wildfires burn range from high canopy fuels to those located on and below the soil surface (fig. 10). Fires can be rather short-lived (minutes), producing little heat to long-lived (months), producing large amounts of heat. The spatial extent of fires can be small (a few m^2, ft^2) to large (100s km^2, mi^2). The burn severity within these extents can range from homogeneous to extremely heterogeneous, leaving a wide variety of vegetation and soil conditions in the after fire (figs. 11, 14, and 15).

Lethal fires (stand replacing), non-lethal (low intensity and severity) surface fires, mixed fires, or a combination, burn through the moist, cold, and dry forests at various intervals and intensities resulting in a variety of burn severities. Fire exclusion has affected all forests, but in general, it has had the largest impact on the dry ponderosa pine/ Douglas-fir forests by changing species compositions, structures, and forest floor constituents in many locales. Fire regimes have minimally changed in the moist and cold forests because of fire exclusion but they have greatly altered dry forests.

White pine blister rust (an exotic stem disease) has altered the moist forests by killing western white pine and is progressing to severely alter the cold forests by killing white bark pine (*Pinus albicalus*) (fig. 9). In addition, because of the uniformity of age and size of lodgepole pine and Engelmann spruce in many cold forests, bark beetles are killing millions of trees (fig. 8).

Forests that were once dominated by vegetative structures and compositions relatively resilient to native insects, diseases, and fire regimes are now (2009) more prone to epidemics of insects and diseases and uncharacteristically large and severe wildfires. The changes in forests occurred through a variety of components, ranging from ground level vegetation to high forest canopies and the forest floor and mineral soil that support the vegetation. In addition, all forest vegetative components—live, dead, and in various stages of decay—are part of the fuel matrix, and their composition, development, structure, and juxtaposition at both fine and broad temporal and spatial scales influence how fires burn. These forest characteristics, in concert with physical setting and weather (observable both at fine and broad spatial and temporal scales), ultimately determine fire behavior and burn severity.

The final item influencing the scope and impact of wildfires is the efficiency of suppression activities that, in concert with locale, fuels, and weather, determine the extent and severity of wildfires. Fuel treatments are the only management activity that can influence fire behavior and burn severity. In addition, depending on forest development and values at risk, the frequency of treatments to maintain desired fuel conditions could occur often. It is recommended that surface fuels be treated first, followed by ladder and crown base height treatments and canopy treatments.

Fuel treatments that affect canopy openings most likely impact the amount of water produced within a watershed, and treatments that disturb the forest floor subsequently influence water sedimentation. Also, the location, number, size, age, intensity, and vegetative and forest floor recovery (development) will determine the cumulative effects of fuel treatments within and among watersheds. Because they tend to occupy locales at the higher elevations relative to the surrounding landscapes, cold forests would contain more headwater stream locations. Both the dry and moist forests occupy the lower elevations and often the dry forests border grass and/or shrublands, especially in the central and southern Rocky Mountains. Therefore, the affects that fuel treatments would cumulatively have on watersheds is highly variable among forest types and highly variable depending on the location and juxtaposition of the forests and treatments within and among watersheds.

Forests die, regenerate, and develop in response to disturbances or the lack thereof. Fuel treatments are yet another disturbance that influences how wildfires behave and the subsequent outcomes they produce (figs. 17, 18 and 21). Along with influencing wildfires, such treatments influence both the physical (soils, water, and air) and vegetative properties of a setting. Cumulatively fuel treatments affect many forest attributes, including wildlife habitat, sense of place, invasive species, cultural resources, sensitive plants, and ecosystem services (air and water quality, water quantity, and soil productivity). Moreover, forests are dynamic and, therefore, fuel treatments should change over time and space. We suggest that it is very useful to display fuel treatments and associated outcomes spatially and temporally in addition to identifying the associated risks and uncertainties. At a minimum, these silvicultural systems should demonstrate their cumulative impacts over time on both wildfires and the soil and water resources within and among watersheds. This is the essence of silvicultural systems and forest planning.

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