Chapter 2

Ponderosa Pine, Mixed Conifer, and Spruce-fir Forests

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Introduction

Before European settlement of the interior west of the United States, coniferous forests of this region were influenced by many disturbance regimes, primarily fires, insects, diseases, and herbivory, which maintained a diversity of successional stages and vegetative types across landscapes. Activities after settlement, such as fire suppression, grazing, and logging significantly altered these disturbance regimes. As a result, forest structure and species composition have departed from historical conditions on many landscapes and this has led to increased forest densities, forest type conversions, and greater contiguity of many western forests. These forests are now more susceptible to large-scale insect infestations, disease outbreaks, and severe wildland fires than in the past, possibly endangering overall forest ecosystem health. The purpose of this paper is to address the historical and current conditions of coniferous forests of southern Utah to aid in the development of treatments to restore the ecological composition, structure, and function of these ecosystems.

The distribution of coniferous forests in southern Utah is mainly influenced by both climate and disturbance regime. Climatic factors, such as temperature and precipitation, determine where certain forest types can grow. Ponderosa pine (*Pinus ponderosa*) forests are the lowest elevation coniferous forest type in southern Utah occurring just above the warmer, drier Colorado pinyon-Utah juniper (*Pinus edulis-Juniperus osteosperma*) woodlands (Youngblood and Mauk 1985). As elevation increases, mixed conifer forests consisting of ponderosa pine, white fir (*Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), blue spruce (*Picea pungens*), limber pine (*Pinus flexilis*), Engelmann spruce (*Picea engelmannii*), and subalpine fir (*Abies lasiocarpa*) occur. Species dominance in these mixed conifer forests is mainly determined by disturbance history and microclimate. In the cool, moist, higher elevations above 3,048 m (10,000 ft), Engelmann spruce-subalpine fir forests dominate when there is minimal disturbance.

Ponderosa Pine Forests

Historical Conditions

Historically, ponderosa pine was found on warm, dry sites on plateaus and mountains of central and southern Utah at elevations ranging from 1,981 to 2,743 m (6,500 to 9,000 ft) (Madany and West 1980; Youngblood and Mauk 1985). Ponderosa pine forests bordered the shrub and woodland communities on its lower elevation range and mixed with Douglas-fir, white fir, blue spruce, and aspen at higher elevations (Powell 1879; Stein 1988a). Historical accounts of ponderosa pine acreage in southern Utah at the time of settlement are limited. The Fishlake National Forest estimates that ponderosa pine forests once occupied 9 percent, or 54,632 ha (135,000 acres), of the 640,212 ha (1,582,000 acres) analyzed within its boundaries (USDA Forest Service, Fishlake National Forest 1999).

Historical accounts of ponderosa pine forests for southern Utah and the southwestern United States indicate that these forests were open with a large diversity of grasses and flowers, with scattered pockets of shrubs (Alter 1942; Cooper 1960; Ogle and DuMond 1997; Powell 1879). Several of these historical descriptions commented on the abundance of thick, lush, and high grass that would provide excellent forage for livestock.

The large diversity and amount of grasses in ponderosa pine forests was due in part to the structure of the forest. Pre-settlement ponderosa pine forests were irregularly spaced, uneven-aged stands with trees growing together in small even-aged groups and grassy meadows between the groups (Covington and Moore 1994a; Dutton 1882; Mast and others 1999; Schubert 1974). Although some groups could be overstocked (Schubert 1974), estimations for tree density and basal area in pre-settlement ponderosa pine forests are low – approximately 99 to 148 trees/ha (40 to 60 trees/acre) with basal areas of 11.5 m^2/ha (50 ft^2/acre) (Covington and Moore 1994b; Rasmussen 1941) – indicating the great size of the mature ponderosa pine trees. For example, a 1911 survey on the Manti-LaSal National Forest indicated that mature ponderosa pine trees grew as large as 1.2 to 1.5 m (4 to 5 ft) in diameter, although it was more common to see trees that were 1.1 m (3.5 ft) in diameter and many of these large trees showed evidence of fire scars (Peterson 1971). Other descriptions noted ponderosa pine trees over 30.5 m (100 ft) tall and 30.5 to 46 cm (12 to 18 inches) in diameter (Ogle and DuMond 1997).

The prevalence of large diameter ponderosa pine trees and the open structure of the forests are attributed to fire and ponderosa pine's insulative bark, which allows it to survive low intensity, surface fires. Frequent, low intensity fires created a diversity of vegetative structures by creating gaps, thinning seedlings, releasing nutrients, encouraging light-intolerant pine germination, and reducing the invasion of fire-intolerant, shade tolerant species. In pre-settlement forests, frequent fires spread easily through fine fuels such as grasses and needles. During dry spells, these fine fuels would allow surface fires to spread over large areas. The surface fires would kill seedlings, saplings, and shrubs and consume the large fuels such as branches and logs. With frequent surface fires, most woody fuels were consumed and large fuel loadings rarely accumulated; therefore, severe fires were rare because of low fuel volumes (Bradley and others 1992). Estimates for fuel loads in pre-settlement ponderosa pine forests are limited. However, Covington and Moore (1994b) estimate that forest floor and woody fuel loadings on North Kaibab ponderosa pine forests were less than 2.24 Mg/ha (1 ton/acre).

Fuels loads and climate were the driving force for pre-settlement fires in ponderosa pine forests. Fires occurred primarily in dry years following wet years (Fulé and others 2000; Swetnam and Betancourt 1990) during the early growing season (Heyerdahl and others 2005; Heyerdahl and others 2006) suggesting that wet pre-fire years allow fine fuel production and buildup that facilitate burning. Mean fire return intervals (MFRI) in ponderosa pine forests of southern Utah vary according to site topography and elevation (Heyerdahl and others 2005; Heyerdahl and others 2006). For example, in three canyons on the Paunsaugunt plateau of the Dixie National Forest, composite MFRI ranged from 15 to 18 years (Stein 1988b). On the Old Woman Plateau of the Fishlake National Forest, the MFRI was 27 years with a range of 6 to 62 years (Heyerdahl and others 2005). Buchanan and Tolman (1983) reported MFRI of 4 to 7 years for Bryce Canyon National Park with a fire occurring at least once a decade within a different area of the forest. They suggest that the fires were small in extent because few of the fire-scarred trees in the same immediate vicinity were burned in the same year. In Zion National Park, MFRI ranged from 2.7 to 25 years on the Horse Pasture Plateau, but within the park on the isolated Church Mesa surrounded by a barren expanse, MFRI was significantly longer, averaging 69 years with a range of 56 to 79 years (Madany and West 1980; Madany and West 1983). Since fire could not start somewhere else on the landscape and spread to the isolated mesa, lightning strikes that hit the mesa were the only source of ignition, resulting in a longer fire return interval. MFRI in a ponderosa pine/Gambel oak forest on the North Rim of the Grand Canyon in Arizona was 3 to 4 years with a range of 1 to 11 years. The frequency decreased slightly to 6.8 years with a range of 2 to 24 years when calculations were made for fires that scarred 25 percent or more of the sample size within the same year, indicating a larger scale fire (Fulé and others 2000).

In general, most pre-settlement fires in ponderosa pine forests in the interior west and Rocky Mountains were of low to moderate severity surface fires (Barrett 1981; Brown and others 1999; Brown and Sieg 1999; Cooper 1960; Covington and others 1997; Fulé and others 1997; Heyerdahl and others 1994; Swetnam and Baisan 1996). Crown fires were rare in open, lightly stocked stands because crowns did not overlap to allow running crown fires and ponderosa pine self-prune their branches, which keeps the foliage separate from the surface fuels. However, on moist sites that did not burn as frequently, more surface fuel probably accumulated and ingrowth of Douglas-fir and white fir served as ladder fuels, thereby increasing the chance for stand replacement and mixed-severity fires (Bradley and others 1992).

Although frequent periodic fire was the major driving force that influenced ponderosa pine forest structure, other disturbances also contributed to the thinning of forests and accumulation of surface fuels for the fires. Mountain pine beetle (*Dendroctonus ponderosae*), western pine beetle (*Dendroctonus brevicomis*), roundheaded pine beetle (*Dendroctonus adjunctus*), and pine engraver (*Ips pini*) all probably attacked stressed ponderosa pine trees in southern Utah. Endemic populations most likely reproduced in stressed or weakened trees, as they do today, and killed a few trees per acre. Epidemics most likely occurred when tree densities became greater than 27.5 m^2/ha (120 ft^2/acre) of basal area (Schmid and others 1994), which would only occur in the absence of fire. Outbreaks of roundheaded pine beetles were probably sporadic and short-lived (Negron and others 2000). On the Kaibab plateau in northern Arizona, Blackman (1931) estimated that pre-settlement outbreaks of mountain pine beetle occurred approximately every 20 years. Even if the ponderosa pine trees were able to resist an insect attack, the tree would still be tree more susceptible to fire due to the exposed resin on the bark around the attacked areas (Bradley and others 1992).

Diseases, such as Armillaria root disease (*Armillaria mellea*) and dwarf mistletoe (*Arceuthobium* spp.), also existed in pre-settlement ponderosa pine forests, but it is unknown to what extent and severity. Woolsey (1911) reported that 1 to 2 percent of ponderosa pine surveyed in Arizona and New Mexico were infected with dwarf mistletoe. Fire often determined the distribution and intensity of dwarf mistletoe infection in coniferous forests (Alexander and Hawksworth 1976). Dwarf mistletoe infections were probably kept to a minimum by frequent fires because severe infections lead to high accumulations of dead trees, highly flammable witches' brooms, and other surface fuels that would have burned more severely (Parmeter 1978). If stands were heavily infected, a fire would be more severe and kill the entire stand, thereby removing the infection source. However, a partial burn that left scattered infected trees could actually lead to rapid infection of regeneration (Alexander and Hawksworth 1976). Armillaria root disease was probably more common since it resides in overmature trees (Schubert 1974). The diseased trees often occur in groups of trees and are susceptible to wind breakage, which adds to the surface fuel loads. With frequent fires, however, the accumulation of such large fuels was minimal, but could probably result in severe fires in affected areas.

Herbivory in pre-settlement ponderosa pine forests was probably kept to a minimum since herbivore populations were quite low in southern Utah and ponderosa pine is not utilized for browse by most wild ungulates. Kay (1995) argues that elk (*Cervus canadensis*) and deer (*Odocoileus hemionus*) in the 1800s were rare in the western United States, including Utah, due to the efficiency of Native American hunters and the predation of the ungulates by wolves (*Canis lupus*), coyotes (*Canis latrans*), and other carnivores. Photographs taken in 1872 by early surveyors showing multi-aged regenerating aspen stands suggest low population levels of elk and deer (Kay 1995). In contrast, bighorn sheep (*Ovis canadensis*) existed in all the mountain ranges of Utah (Dalton and Spillet 1974) and utilized the open ponderosa pine forests (Smith and others 1999) due to the high visibility they provided (Wakelyn 1987). Most likely, herbivores influenced ponderosa pine forest structure by eating the grasses that carried fire, which prompted Native Americans to light fires to enhance the habitat for hunting purposes.

Disease, insects, and fires worked in concert to shape ponderosa pine forest community characteristics. These disturbances helped create heterogeneous landscape structure by creating gaps in the forest canopy, maintaining various age classes, and reducing forest density (Lundquist 1995a; Lundquist 1995b; Lundquist and Negron 2000). Reduction in forest density decreased competition for water and nutrients and subsequently increased tree vigor, which lowered susceptibility to insect and disease attack (Christiansen and others 1987; Kegley and others 1997; Larsson and others 1983; Wargo and Harrington 1991). Frequent, low-intensity fires reduced encroachment of ponderosa pine into meadows and influenced species composition by reducing the invasion of shade-tolerant and fire-intolerant species (Weaver 1967; Wright 1978). The frequent fires prevented surface and ladder fuel buildup, released nutrients (Covington and Sackett 1984; Covington and Sackett 1992), and encouraged germination (Bailey and Covington 2002). Frequent fires also allowed ponderosa pine forests to reach high elevations by reducing the competition with other tree species that were less fire resistant, such as Douglas-fir and white fir.

Current Conditions

The settlement of southern Utah has drastically altered the ponderosa pine ecosystem. The combined effects of fire suppression, logging, and grazing have altered the extent, location, and structure of ponderosa pine forests. Ponderosa pine still borders the Colorado pinyon-Utah juniper woodland (Youngblood and Mauk 1985), but forest inventories have shown a significant decrease in its quantity and extent (Stein 1988a; USDA Forest Service, Fishlake National Forest 1999). Ponderosa pine forests have gained some acreage from riparian zones, aspen, sagebrush, and mountain brush, but have lost significant acreage to Douglas-fir and white fir invasion (USDA Forest Service, Fishlake National Forest 1999; Heyerdahl and others 2005; Heyerdahl and others 2006). In the entire state of Utah, ponderosa pine now covers 240,560 ha (594,436 acres), or 3.7 percent, of the forested land (O'Brien 1999). Comparison of historical versus current acreage is only available for the Fishlake

National Forest where ponderosa pine once occupied 54,632 ha (135,000 acres). Today, ponderosa pine occupies 16,716 ha (41,307 acres), which is a 69 percent decrease in coverage (USDA Forest Service, Fishlake National Forest 1998).

The current stocking of the ponderosa pine understory varies from open to dense thickets (fig. 1). Many ponderosa pine stands in southern Utah now have shrub-dominated understories, most likely a result of continued fire suppression and grazing practices (fig. 2) (Bradley and others 1992). The most common shrubs in the understory include curlleaf mountain-mahogany (*Cercocarpus ledifolius*), greenleaf manzanita (*Arctostaphylos patula*), black sagebrush (*Artemisia nova*), Gambel oak (*Quercus gambelii*), mountain snowberry (*Symphoricarpos oreophilus*), bitterbush (*Purshia tridentata*) and common juniper (*Juniperus communis*). Mountain muhly (*Muhlenbergia montana*), a graminoid, is also a common habitat type in southern Utah (Bradley and others 1992; Youngblood and Mauk 1985). Forbs do not contribute much to the understories of current southern Utah ponderosa pine forests (Bradley and others 1992).

Figure 1—Ponderosa pine (*Pinus ponderosa*) with an understory dominated by muttongrass (*Poa fendleriana*) and rubber rabbitbrush (*Chrysothamnus nauseosus*).

Figure 2—Well-developed understory of greenleaf manzanita (*Arctostaphylos patula*), Gambel oak (*Quercus gambelii*), and Rocky Mountain juniper (*Juniperus scopulorum*) beneath an open canopy of ponderosa pine (*Pinus ponderosa*).

The increase in shrub density is dramatic in some areas. For example, Fulé and others (2002a) studied changes in a ponderosa pine/Gambel oak habitat on the north rim of the Grand Canyon and reported that Gambel oak density has substantially increased since settlement. Gambel oak densities in this area pre-settlement were approximately 1 to 6 percent of the total density, whereas current density of Gambel oak is 20 to 70 percent of total plot density. On these same plots, total tree density (ponderosa pine included) has increased 155 to 486 percent. This increase in Gambel oak populations is a direct result of fire suppression. After fire, Gambel oak can resprout, but oak densities were historically kept low due to frequent fires. With fire suppression, Gambel oak was able to grow into sapling and pole thickets (Fulé and others 2002a).

The overstory structure of ponderosa pine forests has been drastically altered compared to pre-settlement forests (Heyerdahl and others 2006). Reynolds and others (in press) compared current southwest-wide data on ponderosa pine forests from Arizona (Conner and others 1990), New Mexico (Van Hooser and others 1993), and northern Kaibab to Woolsey's (1911) inventory of forests in 1910. Current tree densities averaged 329 trees/ha (133 trees/acre) compared to 89 trees/ha (36 trees/acre) for forests in 1910. Basal areas in 1910 averaged around 18.8 m^2/ha (82 ft^2/acre), while current basal areas average around 13.3 m^2/ha (58 ft^2/acre) (Ogle and DuMond 1997; Woolsey 1911). The main difference, however, is the size of the trees. Due to heavy logging, current forests lack large old growth trees (Ogle and DuMond 1997), but instead have many seedlings, saplings, and small saw-log sized trees.

The current age distribution of ponderosa pine forests has also changed from pre-settlement forests, especially at higher elevations. The age structure in lower elevation ponderosa pine forests follows an uneven-aged distribution with numerous seedlings, saplings, and pole-sized ponderosa pines (Stein 1988a), often in a clumped spatial pattern. Although this distribution has not changed much from pre-settlement forests, the magnitude in the density of young trees is much higher. Furthermore, the mortality of older trees has increased due to the competition for nutrients and water from the dense post-settlement trees and the stagnated nutrient cycling in the absence of fire (Covington and Moore 1994b). The younger age classes of ponderosa pine are missing in higher elevation ponderosa pine forests due to inability to regenerate under more shade-tolerant species such as Douglas-fir and white fir, which have invaded around the widely scattered mature ponderosa pine (Stein 1988a; Mast and Wolf 2004).

Surface fuel loading measurement estimates for southern Utah ponderosa pine forests are scarce. Estimates of total dead fuel loading for several ponderosa pine forests across the southwest range from 44.8 to 69.5 Mg/ha (20.0 to 31 tons/acre) (Bastian 2001a; Covington and Moore 1994b; Sackett 1979). Sackett's (1979) survey of 62 ponderosa pine stands across the southwest United States indicated that 58 percent of all the dead surface fuel was less than 2.54 cm (1 inch) in diameter. This highly flammable fuel, combined with the increased density of live shrubs and small trees that provide ladder fuels to carry fires from the surface into the trees (Cooper 1960; Covington and others 1994b; Madany and West 1980), have increased the chance of crown fire.

Herbaceous production in ponderosa pine forests of southern Utah is currently quite low and these stands lack the graminoid undergrowth that characterizes high quality range (Youngblood and Mauk 1985). A simulation study of the North Kaibab estimates that in pre-settlement ponderosa pine forests, herbage production averaged 0.67 Mg/ha (600 lbs/acre). Current estimates place the production at around 0.10 Mg/ha (100 lbs/acre) (Covington and Moore 1994b). Decreases in herbaceous production are a result of heavy grazing and an increase in both tree and shrub density, which increases competition. In addition, grazing reduces fine fuels to carry surface fire, exacerbating the problem of no fires, which results in stagnated nutrient cycling and ingrowth of more shade-tolerant tree species (Belsky and Blumenthal 1997).

Forests play an important role in supplying water to the majority of southern Utah communities. Descriptions of water yield from pre-settlement ponderosa pine forests are unknown. However, a simulation study of the North Kaibab estimates that in pre-settlement ponderosa pine forests, stream flows were around 17.5 cm (6.9 inches) and have decreased by 4.6 cm (1.8 inches), a 26 percent reduction post-settlement (Covington and Moore 1994b). Since ponderosa pine typically grows on drier sites than other conifers (for example, Douglas-fir, white fir, Engelmann spruce), the majority of water runoff comes from higher elevations where snow accumulates in the winter and melts in the spring. Annual water yield for the different drainage areas within southern Utah ranges between 2.5 to 5 cm (1 to 2 inches) of water, or 7.5 to 13.5 percent of the total precipitation (Hemphill 1998). The use of overstory removal treatments to increase water yield from ponderosa pine forests is short-lived due to the increase in understory herbaceous and shrub cover (Bojorquez-Tapia and others 1990).

Wildfires can greatly affect the hydrology of a forest for several years post-fire depending on the severity. Removal of forest litter in the understory and overstory vegetation increases runoff, peak discharge, soil erosion, sedimentation, and loss of soil nutrients (Baker 1990; Campbell and others 1977; Dunford 1954; Rich 1962). Water runoff was eight times greater on a severely burned (majority of trees killed by fire) watershed than on an unburned watershed the year preceding the wildfire. Higher incidences of runoff events increased for several years in the moderately and severely burned watersheds due to the removal of litter cover and hydrophobic soil. A year after the wildfire, more than 1.4 Mg/ha (1,254 lbs/acre) of sediment was lost in runoff from the severely burned watershed, although within another year, amount of sediment production returned to that of the unburned watershed (Campbell and others 1977).

Current Disturbances

Fire—The exclusion of fire since the late 1800s and early 1900s has greatly reduced periodic fires in southern Utah (Buchanan and Tolman 1983; Heyerdahl and others 2005; Heyerdahl and others 2006; Madany and West 1980; Madany and West 1983; Stein 1988b). Increased densities of small, young trees, build-up of surface fuels, and large areas of contiguous forests increase the likelihood of large-scale, severe fires. Lightning-caused wildfires in the southwest are getting larger over time, with some reaching tens of thousands of hectares (and getting bigger), in contrast to the 40 to 400 ha (100 to 1,000 acres) surface fires of pre-settlement times (Heyerdahl and others 2006; Swetnam 1990). Since 1972, in the entire Fishlake National Forest, 70 wildfires have burned with an average size of 650 ha (1,607 acres), but with some fires up to 7,440 ha (18,385 acres) in size (Fishlake National Forest GIS data). Estimates for all federal lands in the entire state of Utah from 1986 to 1996 indicate that there were 8,335 fires with an average size of 50.5 ha (125 acres), but some reached 28,781 ha (71,120 acres) (Schmidt and others 2002).

Insects and Disease—Tree vigor has also declined as a result of fire suppression and the subsequent increase in forest density and competition. This decline in tree vigor increases the potential for more insect infestations. Forest inventory reports for the Dixie, Fishlake, and Manti-La Sal National Forests indicate that between 73 to 93 percent of ponderosa pine trees within these forests are at moderate to high risk of attack by bark beetles (O'Brien and Brown 1998; O'Brien and Woudenberg 1998; O'Brien and Waters 1998). Since 2002, infestations of mountain pine beetle have increased in acreage in these forests (Matthews and others 2005). The North Kaibab Plateau in Arizona has experienced several outbreaks of mountain pine beetle in the past century (Blackman 1931; Parker and Stevens 1979; Wilson and Tkacz 1995). The roundheaded pine beetle was at epidemic levels in 1995 in the Dixie National Forest (Negron and others 2000). Outbreaks of the roundheaded pine beetle have caused considerable mortality across the southwest (Lucht and others 1974; Massey and others 1977), reducing basal area and total numbers of ponderosa pine trees by up to 50 percent (Stevens and Flake 1974) and increasing large diameter woody fuel loadings up to eight times of what was already on the ground (Negron 2002). Increased surface fuel loads as a result of bark beetle epidemics increases the potential for fire hazard and the probability for higher-intensity fires for several years (Schmid and Amman 1992).

Lower tree vigor has also made ponderosa pine trees susceptible to several diseases. Over 20 percent of ponderosa pine trees in Utah are infected with dwarf mistletoe (Matthews and others 2005). Dwarf mistletoe increases the chance of a surface fire to transition into a crown fire due to the flammable witches broom and lower crowns (Harrington and Hawksworth 1990). Armillaria root disease is also infecting and killing mature and immature ponderosa pine trees in Utah (Forest Health Protection 2000) creating additional surface fuel loads.

Ungulates—Livestock grazing is one post-settlement disturbance that has indirectly contributed to increased ponderosa pine and shrub densities. Utah was severely overgrazed in the late 1800s and throughout the 20th century (Ogle and DuMond 1997). Several large ranches were established in Utah and Arizona as early as 1863 (Altschul and Fairley 1989), and cattle and sheep have heavily grazed the Paunsaugunt Plateau since 1866 (Rathburn 1971). Guidelines for range allotments were non-existent in the 1800s, but by 1920, the Forest Service started to enforce them (Stein 1988a).

Overgrazing and trampling by livestock changed understory species composition, increased the amount of bare ground, decreased water storage, increased runoff, compacted soil, and increased erosion (T. Alexander 1987; Belsky and Blumenthal 1997). Heavy grazing by livestock promoted the establishment of tree seedlings due to the reduction of herbaceous ground cover and an increase in bare soil (Madany and West 1983). Grazing also decreased the competition of grasses with shrubs and increased the density and extent of shrubs such as

Gambel oak, bigtooth maple, Utah serviceberry, and greenleaf manzanita (Mitchell 1984). With the loss of fine fuels (grass) for frequent fire, ponderosa pine was able to expand into ecotonal communities such as sagebrush and mountain brush.

Increased numbers of wild ungulates, such as deer, elk, and moose (*Alces alces*), have reduced herbaceous vegetation and aspen suckering in many stands. Pre-settlement populations of elk and deer were very low (Kay 1995; Rawley 1985), but once Native Americans populations were removed and wolves, mountain lions (*Felis concolor*), coyotes, and other predators were extirpated, ungulate populations exploded (Rasmussen 1941). Deer populations on the Kaibab Plateau had reached 100,000 before declining in 1924 due to starvation and the creation of government hunting programs. Elk were transplanted into Utah from 1912 to 1915 (Rawley and Rawley 1967), and moose have recently been transplanted into south-central Utah (Kay and Bartos 2000). Current populations in southern Utah are approximately 116,000 deer, 23,000 elk, and 65 moose (Utah Division of Wildlife 2005a,b,c).

Exotic Weeds—Overgrazing has also lead to the invasion of exotic species such as cheatgrass (*Bromus tectorum*). Cheatgrass originated from Eurasia where it coevolved with heavy grazing and has adapted well to heavy grazing regimes in the west where native species are at a disadvantage (Stebbins 1981). Cheatgrass grows in dense stands and cures by mid-June, about two to four weeks earlier than native grasses (Devine 1993). Once cheatgrass is dry, it can carry a fast moving fire and cause more frequent, intense and early-season wildfires. Cheatgrass has already increased fire frequency in lower elevation ecosystems such as sagebrush (Kitchen and McArthur, this volume). Cheatgrass and other exotics, such as mullein (*Verbascum thapsus*), toadflax (*Linaria dalmatica*), and thistle (*Circium pulchellum*), are often found in severely burned areas (Keeley 2003; Phillips and Crisp 2001; Sackett and Haase 1998). Cheatgrass has been observed in understories throughout the ponderosa pine range, from California (Keeley 2003) to Colorado (Fornwalt and others 2003).

Fire Regime Condition Classes

Fire Regime Condition Class 1 (FRCC 1)

Ponderosa pine forests functioning within the historic range of variability contain trees of all sizes and ages. Stands usually occur in even-aged groups or clumps. Periodic surface fires have pruned the branches of large trees well above the ground, creating an open appearance with long sighting distances under the forest canopy. Such fires also have prevented large accumulations of surface fuels and kept the density of smaller trees and woody understory species low, creating a diverse understory of grasses and forbs. Periodic fires have created openings of up to several acres as a result of the torching of clumps or groups of trees. Such openings increase the spatial diversity of the forest and lessen the occurrence of landscape-wide stand replacement crown fires by serving as fuel breaks. Frequent surface fires do not consume the soil's organic layer that helps stabilize the soil surface and prevent excessive erosion. Smoke production is low in volume and short in duration, but regional landscapes would likely have been more smoky than today.

Fire Regime Condition Class 2 (FRCC 2)

Ponderosa pine forests existing under moderately altered fire regimes are denser and contain higher numbers of young trees in the understory than a historical stand. Large pines are still a component of the forest. Smaller, shade tolerant conifer associates are present at some places in the understory and as occasional larger trees. Smaller openings have been invaded by ponderosa pine, which creates a more contiguous forest canopy. Small pines also have branches closer to the ground. Understory species abundance and diversity is less than that in FRCC 1.

A wildfire occurring in a current FRCC 2 ponderosa pine forest is a mixture of severity and intensity, larger in size than historically observed, and difficult to suppress. Smoke production is probably moderate in volume and duration. Understory plants and small trees burn and provide a pathway for the fire to reach overstory foliage, which leads to large areas of torched trees. Soil productivity is more severely impacted (in other words, reduced) as a result of litter and upper duff consumption. Areas of bare ground are subject to some erosion, and sedimentation of streams decreases water quality. Biodiversity of herbaceous plants increases for the first few years post-fire, but with an increased probability of exotic species. Regeneration of ponderosa pine is possible in areas with surviving trees.

Fire Regime Condition Class 3 (FRCC 3)

Ponderosa pine forests are considerably altered from those in FRCC 1, either through repeated harvests or through vegetative succession as a result of altered fire regimes. Where harvests have occurred, most large pre-settlement trees have been removed. If post-harvest regeneration was successful, dense, evenly-spaced forests of younger pines now exist. If pine regeneration was not successful, oak shrubs or other conifers replaced the logged pine. This created either mixed species forests or open pine forest with dense shrub understories. In either case, understory grass and forb production has declined, along with populations of animals and birds that depended upon the diversity of FRCC 1 ponderosa pine forests.

A wildfire occurring in a current FRCC 3 ponderosa pine forest is extremely damaging to the ecological integrity of an area. The wildfire is high in severity and intensity, covers large acreages, and is extremely difficult and costly to suppress. Smoke production is extremely high in volume and duration. Understory plants, small trees, and even large trees burn, resulting in a completely altered landscape. Nutrients are volatized, soil microbes killed, and soil organic matter consumed where accumulations of woody fuels burned. Soil is more likely to become hydrophobic and increased erosion and sedimentation into streams is likely to impact regional water quality. Exotic species invasion is likely in areas of high fire severity. Regeneration of ponderosa pine is limited to areas that are not completely burned.

Ponderosa pine forests in southern Utah are somewhat of an anomaly compared to those in other regions. Inventory data indicate that the average ponderosa pine forest in southern Utah is relatively poorly stocked, containing mostly young, small diameter trees (O'Brien 1999; O'Brien and Brown 1998; O'Brien and Waters 1998; O'Brien and Woudenberg 1998). In terms of stocking, ponderosa pine forests would initially appear to be in FRCC 1. However, the lack of large old growth trees (O'Brien 1999; O'Brien and Brown 1998; O'Brien and Waters 1998; O'Brien and Woudenberg 1998) and 120+ years of fire exclusion would suggest that the majority of southern Utah ponderosa pine forests are not within the historical range of variation and are likely in FRCC 2 or 3.

We suspect that repeated timber harvests, heavy grazing, and wildfire suppression have significantly altered the species composition and fuel (surface and canopy) loadings of many ponderosa pine forests in southern Utah. Expansion of oak, pinyon, and juniper into the understories of lower elevation ponderosa pine and the increased presence of Douglas-fir and true fir in pine forests at higher elevations (Mast and Wolf 2004) are likely to have created conditions that will require restoration treatments before fire is reintroduced. A 69 percent post-settlement decrease in ponderosa pine on the Fishlake National Forest (USDA Forest Service, Fishlake National Forest 1999) indicates that type conversion from ponderosa pine to other forest types is occurring.

The increased presence of these associated forest species compared to pre-settlement conditions has resulted in an increased risk of stand replacement wildfire, even though the stocking of ponderosa pine is low. The lack of large thick-barked, fire-pruned ponderosa pines only increases the chances that all pines will be eliminated from the forest if fires should occur. Shade tolerant conifer seedlings such as Douglas-fir and true fir, and oak in the understory, provide live fuel ladders that will allow fire to easily reach into the crowns of ponderosa pine.

Recommended Treatments

In contrast to strict thinning from below where all small trees are removed, we advocate the use of an uneven-aged approach in maintaining open ponderosa pine stands that are at less risk to crown fire. We recommend converting the understocked ponderosa pine stands common to southern Utah to irregularly structured uneven-aged stands by reducing or removing shade tolerant conifers and oak and re-introducing frequent prescribed surface fires. Development of stand structures should focus on creating forests with basal areas and densities that occurred historically, which includes a component of large-diameter trees that can withstand wildfires. Historic ponderosa pine forest structure in the Grand Canyon and the Kaibab Plateau in nearby Arizona averaged 80 to 160 trees/ha (32 to 65 trees/acre) with basal areas ranging between 5.9 to 20.5 m^2/ha (25.7 to 89 ft^2/acres) (Fulé and others 2002a, 2002b, 2006) and quadratic mean diameters between 47 to 52.7 cm (18.5 to 20.7 inches) (Fulé and others 2002b). The estimates for historic southern Utah ponderosa pine forests had 0 to 297 trees greater than 20 cm dbh per ha (0 to 120 trees/acre) (Heyerdahl and others 2006).

Control of stocking or density under uneven-aged management can be achieved using either the Stand Density Index (SDI) or BDQ method. SDI utilizes upper and lower diameter limits, but assigns stocking evenly across all diameter classes except for a reduction in relative density for the smallest diameter class (see Long and Daniel 1990; Long 1995). As in mixed conifer forests, ponderosa pine forests should contain 30 percent maximum SDI stocking or less and contain trees of all sizes and ages (Long 1995). Several papers provide detailed examples for calculating SDI to regulate stocking in ponderosa pine forests (Long and Daniel 1990; Long 1995; Shepperd 2006). Shepperd (2006) describes a methodology (SDI-Flex) by which an infinite variety of stand configurations can be maintained.

BDQ relies on basal area, diameter distribution, and a "Q" ratio. Under the BDQ system, managers must first select the upper and lower target diameters that define the range of tree sizes to be managed. Second, a "Q" factor must be chosen. The "Q" factor is the ratio between the number of trees per acre in one diameter class and those in the next smaller class (Alexander and Edminster 1987). Lastly, the residual basal area that the growing stock is reduced to following each cutting cycle must be chosen. Examples of calculating desired stocking using the BDQ method are presented in Guldin (1996) and in Alexander and Edminster (1987). It is important to note that for the success of either method, managers must be diligent in monitoring regeneration and levels of growing stock across all diameter classes during each cutting cycle. Marking and thinning of such forests should be done to create a grouped or clumped appearance and be irregular in both spatial and vertical structure. Silvicultural prescriptions must include a flexible time table for future thinnings and prescribed burn treatments.

In southern Utah, much of the above-mentioned activity will need to be done in the future, rather than at present. Regional inventory data (O'Brien 1999; O'Brien and Brown 1998; O'Brien and Waters 1998; O'Brien and Woudenberg 1998) indicate many ponderosa pine forests are poorly stocked and will require time to grow into desirable ponderosa pine stocking conditions. Initial manipulation may still be required to reduce stocking of other species such as white fir or Douglas-fir. Both of these species would reduce available light and resources for successful ponderosa pine establishment and growth. Furthermore, white fir and Douglas-fir would increase the risk for a stand replacing wildfire.

In areas where white fir and Douglas-fir are present in ponderosa pine stands, they should be removed in the first uneven-aged entry to set the successional stage back to a purer ponderosa pine stand. A prescribed burn to kill fir seedlings should follow thinning. Similarly, where aspen occurs in conjunction with ponderosa pine, the aspen should be encouraged to come in under the pine. If aspen is a desired component and is already present in the stand, SDI stocking or BDQ guidelines may need to be lowered to create a more open stand to allow aspen to thrive. Normally, marking using a group selection technique should be adequate to maintain small clones of aspen interspersed among ponderosa pine. Periodic prescribed burns and/or harvest entries can be used to maintain the presence of aspen in these landscapes. Where ponderosa pine grows in conjunction with Gambel oak or other hardwoods species, periodic prescribed fire is needed to keep those shrub species in check.

Maintaining ponderosa pine forests as open, irregularly spaced forests has the additional benefit of increasing forage production for domestic livestock and wildlife. Because allotment stocking rates have not been adjusted in many years, the gradual loss of grasses and forbs due to shrub and shade-tolerant conifer ingrowth has actually reduced herbaceous production compared to a century ago.

A primary goal in restoration treatments in ponderosa pine stands should be to increase the presence of large trees through silvicultural practices. Repeated timber harvests in many areas of southern Utah selected only the larger trees and have reduced the average tree size considerably from that of the past. In terms of uneven-aged management, the largest pine stumps found on a site can serve as upper diameter limit targets when developing stocking guidelines.

If managers desire to use the BDQ or SDI-Flex (Shepperd and others 2006; Shepperd 2007) approach in achieving stocking goals for fuels reduction and forest restoration, diameter distributions should be chosen that do not result in an over abundance of young trees in the forest. For instance, increasing the Q factor (or flex factor) will increase the density of smaller trees, and decreasing the Q factor will lower the density of smaller trees (Shepperd and Battaglia 2002; Shepperd and others 2006; Shepperd 2007). Typically, if 5-cm (2-inch) diameter classes are used, a Q of 1.2 or less is desirable in ponderosa pine forests and no more than 1.3 in mixed conifer. Higher values will result in young trees that will have to be removed in subsequent entries.

Modeling and experience has shown that using uneven-aged management techniques with periodic entries on sites in both ponderosa pine and mixed conifer forests similar to those in southern Utah can be accomplished every 30 years and be economically sustainable (Skog and others 2006). A commercial timber harvest could be used to remove some medium and large diameter material, taking care to leave a sufficient number of large trees to create conditions similar to historic stocking and basal area levels. However, biomass or other markets are needed for smaller diameter materials to ensure they are also removed from the site to reduce competition and surface fuel buildup. As with all restoration cuts, finding markets for the small diameter wood biomass in southern Utah to offset costs is as critical an issue as anywhere else in the west. Much of what needs to be done cannot be accomplished without markets for small diameter material.

Mixed Conifer Forests

Historical Conditions

Pre-settlement mixed conifer forests contained ponderosa pine, Douglas-fir, white fir, aspen, blue spruce, Engelmann spruce, and subalpine fir. Distribution of these species depended upon disturbance history, aspect, elevation, and available moisture (Bradley and others 1992). Powell (1879) observed a mixture of Douglas-fir, white fir, ponderosa pine, and blue spruce growing from 2,133 to 2,743 m (7,000 to 9,000 ft). Lower elevations were dominated by ponderosa pine with Douglas-fir as the second most common tree. Ponderosa pine was prevalent on drier, southern aspects at middle and upper elevations up to 2,591 m (8,500 ft). At elevations above 2,591 m (8,500 ft), limber pine, Douglas-fir, and especially white fir performed best (Buchanan and Harper 1981). Detailed acreage estimates of pre-settlement mixed conifer forests for southern Utah are not available.

Lang and Stewart (1910) described the mixed conifer forests of the Kaibab Plateau as open and subjected to multiple wildfires that created partially denuded landscapes. A reconstruction of a mixed conifer forest in Little Park (>2,650 m [8,700 ft]) on the North Rim of the Grand Canyon provides a small picture of the species composition, age structure, and forest structure of a mixed coniferous forest in 1880 (Fulé and others 2003; Fulé and others 2002c). Tree density in this area averaged 40 trees/ha (98 trees/acre) with an average basal area of 17.7 m^2/ha (77 ft^2/acre). Ponderosa pine, white fir, and aspen each made up 24 to 27 percent of the tree density, followed by Douglas-fir (19 percent), and small amounts of Engelmann spruce (5 percent) and subalpine fir (1 percent). Although aspen made up 25 percent of the tree density, the species only contributed 4 percent to the basal area, indicating high numbers of sprouts. In contrast, ponderosa pine, white fir, and Douglas-fir each made up 31 to 32 percent of the basal area, suggesting the presence of larger, older trees (Fulé and others 2003; Fulé and others 2002c).

At another lower elevation site (2,400 to 2,500 m [7,962 to 8,300 ft]) within the Grand Canyon, Swamp Ridge, Fulé and others (2002a) reconstructed the 1880 mixed conifer forest and found no Engelmann spruce or subalpine fir. Average tree densities were similar to Little Park, but with higher average basal area (28.5 m^2/ha [124 ft^2/acre]). Ponderosa pine was dominant in this stand making up 75 percent of the basal area and 53 percent of the tree density. White fir (13 percent) had twice the tree density of Douglas-fir (6 percent), but its basal area was only 20 percent higher, indicating an ingrowth of white fir seedlings. As with the Little Park area, aspen contributed about 27 percent of the tree density, but only 4 percent of the basal area. Again, this suggests there were a high number of sprouts (Fulé and others 2002a). Of interest is the wide range of basal areas (15 to 54 m^2/ha [66 to 235 ft^2/acre]) found on the Swamp Ridge plots. This suggests both open and dense stands were present in pre-settlement mixed conifer forests as a result of mixed severity fires (Fulé and others 2003).

The diverse stand structure of pre-settlement mixed conifer forests contributed to a variety of understory conditions ranging from dry, open, grassy understories to moist, closed canopy understories with a diverse mixture of plant lifeforms. Although no quantitative pre-settlement descriptions of shrub and herbaceous conditions for mixed conifer forests were found in the literature, some conclusions can be drawn from Buchanan and Harper's (1981) comparison of a 1959 and 1980 botanical survey of Bryce Canyon National Park. This study reported a 30 to 44 percent decrease in average understory coverage in four mixed conifer community types and a 21 percent to 24 percent decline in understory diversity in three mixed conifer community types due to an increase in forest overstory density. Community types with white fir showed the largest decline in shrub coverage. Based on these data, we suggest that shrubs (especially sprouters) in pre-settlement mixed conifer forests, which had even lower overstory densities than those observed in 1959 and were subjected to fire, were more prevalent, and more grasses and forbs were present due to a greater availability of light.

Based on the varied densities and composition of pre-settlement mixed conifer forests, surface fuels loads in these stands were probably quite variable. The fuel complex of pre-settlement mixed conifer forests in Bryce Canyon National Park has been described as a timber overstory with a grass or herbaceous surface stratum (Anderson 1982; Jenkins and others 1998; Roberts and others 1993). Although there are no detailed inventories of pre-settlement surface fuel loads, Dieterich (1983) estimates that surface fuel loads were probably less than one-fourth to one-third of present-day loadings.

Mean fire return intervals (MFRI) reported in the literature for mixed conifer forests of the southern Utah region range from 2 to 129 years depending upon location, species composition, and methodology used (Buchanan and Tolman 1983; Chappell and others 1997; Fulé and others 2003; Heyerdahl and others 2005, Heyerdahl and others 2006; Jenkins and others 1998; Stein 1988b; Touchan and others 1996; White and Vankat 1993; Wolf and Mast 1998). In general, MFRI lengthened with elevation and composition of shade-tolerant species (Fulé and others 2003; Heyerdahl and others 2005; Heyerdahl and others 2006; Wolf and Mast 1998). For example, Wolf and Mast (1998) reported lower elevation mixed conifer forests had MFRI ranging from 5 to 7.25 years, while higher elevation forests with a spruce component had longer return intervals of 10 to 19 years, with the lower value including all sampled trees and the higher value calculated for fires that scarred 25 percent or more of the sample size within the same year. Stands sampled on the Fishlake National Forest showed similar patterns with lower elevation mixed conifer forests having MFRI ranging from 15 to 30 years and higher elevation mixed conifer forests with an Engelmann spruce-subalpine fir component ranging from 45- to 57-year mean fire interval (Chappell and others 1997; Heyerdahl and others 2005).

In general, most pre-settlement fires in mixed conifer forests were low to moderate intensity surface fires (Buchanan and Tolman 1983; Dieterich 1983; Fulé and others 2003; Heyerdahl and others 1994; Swetnam and Brown 1992), especially on southern and western aspects (Fulé and others 2003). In areas of high tree density or areas with high insect and disease-caused mortality, patchy crown fires and high intensity fires were possible (Bradley and others 1992). At lower elevations, fires burned during the dormant and early growing season, while higher elevations burned during the dormant and late growing season (Heyerdahl and others 2005; Heyerdahl and others 2006).

Several types of insects influence the structure and composition of mixed coniferous forests in southern Utah because they are host specific and their mode of attack differs (Swetnam and Lynch 1989; Swetnam and Lynch 1993). For instance, mountain pine beetles (*Dendroctonus ponderosae*), western pine beetles (*Dendroctonus brevicomis*), pine engravers (*Ips pini*), and roundheaded pine beetles (*Dendroctonus adjunctus*) all attack ponderosa pine, while Douglas-fir beetles (*Dendroctonus pseudotsugae*) only attack Douglas-fir. The Douglas-fir tussock moth (*Orgyia pseudotsugata*) and fir engravers (*Scolytus ventralis*) prefer both white fir and Douglas-fir. Western spruce budworm (*Choristoneura occidentalis*) attacks Douglas-fir, white fir, subalpine fir, and Engelmann spruce, but usually ignores ponderosa pine. Spruce beetle (*Dendroctonus rufipennis*) attacks only Engelmann spruce.

Although there are few descriptions of insect or disease attacks on pre-settlement mixed conifer forests in southern Utah, other regions of the western United States have experienced evidence of episodic outbreaks of Douglas-fir tussock moth (Wickman and Swetnam 1997) and western spruce budworm (Swetnam and Lynch 1989; Swetnam and Lynch 1993; Veblen and others 1994). In the Blue Mountains of Oregon, Wickman and Swetnam (1997) reported Douglas-fir tussock moth outbreaks occurred over large areas and were synchronous. Western spruce budworm outbreaks in southern Colorado and northern New Mexico were shown to occur at irregular intervals over the last 300 years with intervals between outbreaks ranging from 14 to 58 years with an average duration of 12.9 years (Swetnam and Lynch 1989; Swetnam and Lynch 1993). Mountain pine beetle epidemics reported in the previous ponderosa pine section of this chapter probably have attacked the ponderosa pines found in mixed conifer forests. Spruce beetle outbreaks in mixed conifer forests would have been limited to the upper elevations where more Engelmann spruce was present.

Diseases, such as Armillaria root disease (*Armillaria mellea*), annosus root disease (*Heterobasidion annosum*), and dwarf mistletoe (*Arceuthobium* spp.), existed in pre-settlement mixed conifer forests, but it is unknown to what extent. Outbreaks in pre-settlement mixed conifer forests were probably not as detrimental as they are today for several reasons. First, frequent fires would have maintained low to moderate basal areas. These fires decreased competition and increased tree vigor, lowering the susceptibility to insect and disease attacks. Additionally, insects and diseases are host specific and the pre-settlement mixed conifer forests maintained a mixture of tree species. This combination would allow tree species not susceptible to a certain insect or disease attack to regenerate while the susceptible tree species declined.

Disturbances in mixed conifer forests affect community characteristics in several ways. Species composition and tree density are somewhat regulated by fire due to the different fire resistance characteristics of each species. Mature ponderosa pine and Douglas-fir have thick bark that is very resistant to fire. However, seedlings and saplings of Douglas-fir are vulnerable to surface fires, while young ponderosa pine can maintain a presence on sites with fire intervals as short as 6 years if fire severity is low (Bradley and others 1992). Young white fir trees are also vulnerable to fire, but as they mature, the bark becomes thicker and more resistant, although its low branching habit increases its susceptibility to crown fires. Fire resistance in young and mature blue spruce, subalpine fir, and Engelmann spruce is very low and these species are often killed by fire. Aspens are also easily killed by fire, but can revegetate a site quickly with new sprouts produced through root suckering (Bartos, this volume).

Fire frequency and severity are the major factors that determined the species composition of pre-settlement mixed coniferous forests. Most areas were probably dominated by ponderosa pine and Douglas-fir in the overstory because of the frequent fire regime in the pre-settlement era. In areas where fire intervals were longer, white fir was able to develop thick enough bark to resist low intensity surface fires and contribute to post-fire regeneration. At higher elevations, where fires were not as frequent, a greater proportion of spruce and fir were present. Aspens were prevalent in pre-settlement mixed conifer forests under both frequent and mixed mode fire regimes and served as natural firebreaks. In areas with longer fire intervals, aspen populations were much lower, but could increase in response to fire, serving as a nurse crop for conifers.

Current Conditions

Species composition, forest density, structure, and disturbance regimes have been altered in many mixed conifer forests of southern Utah since settlement. Interruption of natural fire regimes has allowed succession to move these forests toward more shade-tolerant species. As a result, ponderosa pine is no longer dominant in mixed conifer forests and aspen populations have declined dramatically (fig. 3). Ponderosa pine has lost acreage to both Douglas-fir and white fir, and in turn, Douglas-fir has lost acreage to white fir (fig. 4) (USDA Forest Service, Fishlake National Forest 1998).

For the entire state of Utah, Douglas-fir covers 456,647 ha (1,128,400 acres) and white fir covers 163,374 ha (403,707 acres) (O'Brien 1999). Within southern Utah, the Douglas-fir cover type comprises about 5 percent and white fir makes up 2.5 percent of the forested land (O'Brien and Brown 1998; O'Brien and Woudenberg 1998; O'Brien and Waters 1998).

Mixed conifer stands in southern Utah are dominated by a variety of tall and low shrubs (Bradley and others 1992; Youngblood and Mauk 1985). The most common shrubs found in the Douglas-fir, white fir, and blue spruce habitat types of southern Utah include ninebark (*Physocarpus malvaceus*), curlleaf mountain mahogany (*Cercocarpus ledifolius*), greenleaf manzanita (*Arctostaphylos patula*), mountain mahogany (*Cercocarpus montanus*), Gambel oak (*Quercus gambelii*),

Figure 3—Colorado blue spruce (*Picea pungens*) gaining dominance in a stand of trembling aspen (*Populus tremuloides*) with an understory dominated by Kentucky bluegrass (*Poa pratensis*) and mountain snowberry (*Symphoricarpos oreophilus*).

Figure 4—Mixed conifer forest with white fir (*Abies concolor*) and ponderosa pine (*Pinus ponderosa*). The most dominant understory species is greenleaf manzanita (*Arctostaphylos patula*).

Oregon grape (*Berberis repens*), mountain snowberry (*Symphoricarpos oreophilus*), mountain maple (*Acer glabrum*), and common juniper (*Juniperus communis*). The majority of these shrubs resprout after fire at various degrees, with the exception of common juniper and curlleaf mountain mahogany (Bradley and others 1992). The majority of herbaceous stratums in the habitat types of southern Utah are depauperate, although some of the white fir and blue spruce areas do support a small amount of graminoids and forbs (Buchanan and Harper 1981; Youngblood and Mauk 1985).

The most dramatic change in mixed conifer forests is the increase in basal area, tree density, and species composition shift toward white fir at lower elevations and Engelmann spruce and subalpine fir at the higher elevations (Bastian 2001b; Fulé and others 2003; Fulé and others 2002a; Heyerdahl and others 2005; Heyerdahl and others 2006; Jenkins and others 1998; Mast and Wolf 2004). For instance, in one area of Bryce Canyon, the largest and oldest trees (200 to 250 years old) are ponderosa pine, Douglas-fir, and white fir, but the regeneration for the past 100 years is mostly white fir and Douglas-fir (Jenkins and others 1998). In another area of Bryce Canyon National Park, white fir overstory density is 33 trees/ha (81.7 trees/acre) compared to a ponderosa pine overstory density of 9 trees/ha (22.8 trees/acre). More striking is the regeneration layer where white fir seedlings density is 649 trees/ha (1,604 trees/acre) and ponderosa pine seedling density is 15 trees/ha (36.8 trees/acre) (Bastian 2001b). At Swamp Ridge (2,400 to 2,500 m [7,962 to 8,300 ft]) on the Northern Rim of the Grand Canyon, there has been a 283 percent increase in tree density and a 45 percent increase in basal area (Fulé and others 2002a). Although ponderosa pine still dominates basal area, it only makes up 17 percent of the tree density. The majority of pre-settlement aged trees are ponderosa pine, with a substantial increase in white fir and aspen regeneration after fire regimes were disrupted (Fulé and others 2002a). At a higher elevation site within the same area, average basal area is 38.8 m^2/ha (169 ft^2/acre) with a tree density of 143 trees/ha (353 trees/acre) (Fulé and others 2002a). Although white fir density has not increased since pre-settlement in this stand, there has been a substantial increase in the density of Engelmann spruce and subalpine fir at the expense of ponderosa pine and Douglas-fir. Furthermore, Engelmann spruce and subalpine fir now contribute 19 percent to the total basal area in contrast to the 3 percent they contributed in the pre-settlement stand (Fulé and others 2002a).

Fuel loadings estimates for southern Utah mixed conifer forests are limited. In Bryce Canyon National Park, current total surface fuel loading estimates are around 69.5 to 71.7 Mg/ha (31 to 32 tons/acre), a 200 percent increase since 1900 (Bastian 2001b; Roberts and others 1993; Jenkins and others 1998). At higher elevations, mixed conifer forests average total surface fuel loading is 118.8 Mg/ha (53 tons/acre), the majority

(87 percent) of which is 1000-hour rotten and sound fuels (Fulé and others 2002c). However, Sackett's 1979 survey of 16 southwestern mixed conifer stands indicated total surface fuel loadings averaged 98.6 Mg/ha (44 tons/acre), half of which was from 1000-hour fuels.

Although no formal forest hydrology studies exist in southern Utah mixed conifer forests, results can be extrapolated from research in mixed conifer forests in Arizona (Rich and Thompson 1974). Water yields were inversely related to the amount of forested area of the watershed. Clearcutting increased water yields in proportion to the percent of the forested area clearcut. While most of the yield was accounted for in the Arizona study by reductions in evapotranspiration, snow interception also likely plays a role in mixed conifer ecosystems (Troendle and others 1988). Current knowledge of the relationship between water yield and forest structure holds that water yield in mixed conifer forests is a function of the basal area/leaf area within the watershed (Shepperd and others 1992). Because snow packs do accumulate in mixed conifer forests, we can expect them to be hydrologically similar to spruce-fir forests (see following spruce-fir section). Peak discharge rates should not be appreciably altered by removal of vegetation, but duration of flow lengthens. Therefore, any removal or increase in forest vegetation (through fire, harvest, insect attack, or succession) would potentially translate into an increase in water yield.

Wildfires can greatly affect sediment production in mixed conifer forests. A 60-acre wildfire in a mixed conifer forest occurred on the South Fork watershed of Workman Creek in Arizona, destroying 74 percent of the basal area (Rich and Thompson 1974). Average post-wildfire sediment production in this area was 50.6 m^3/ha (726 ft^3/acre), compared with 0.009 to 0.98 m^3/ha (0.14 to 14 ft^3/acre) in unburned mixed conifer forest within the same watershed.

Current Disturbances

Fire—Since settlement, fire-free intervals have increased in mixed conifer forests similar to ponderosa pine forests. Shade tolerant species now dominate in many areas of the mixed conifer forest and the continuity of forest vegetation has increased in many landscapes. Instead of a mixed mode of surface fire and patchy crown fire, mixed conifer forests may now burn in large landscape crown fires (Fulé and others 2004). Loss of pure aspen stands, which served as natural fire breaks within mixed conifer landscapes, through succession, fire suppression, and overbrowsing has further exacerbated the situation.

Insects and Disease—Occurrences of western spruce budworm, mountain pine beetle, Douglas-fir beetle, and Douglas-fir tussock moth have likely increased with increasing density and continuity of southern Utah mixed conifer forests. Forest inventory reports for the Dixie, Fishlake, and Manti-LaSal National Forests indicate that between 61 and 74 percent of all Douglas-fir trees within the forests are at moderate to high risk of attack by bark beetles (O'Brien and Brown 1998; O'Brien and Woudenberg 1998; O'Brien and Waters 1998). In fact, outbreaks of Douglas-fir beetle have substantially increased on each these National Forests since 2000 (Forest Health Protection 2000; Matthews and others 2005). Douglas-fir beetle epidemics seem to be more synchronous on a larger scale than in pre-settlement forests. Epidemics of this beetle have occurred in the Front Range of Colorado approximately every 15 to 35 years in the 20th century (Schmid and Mata 1996). Douglas-fir beetle epidemics typically occur following an extensive windthrow event or fire (Furniss and Carolin 1977). In addition, Douglas-fir epidemics in Colorado and Wyoming have taken place following western spruce budworm epidemics (Schmid and Mata 1996).

Western spruce budworms defoliate Douglas-fir, white fir, subalpine fir, and to a lesser degree, Engelmann spruce. Fire suppression and past management practices have created multi-storied, dense, and continuous forests that provide abundant food sources for all larval stages of western spruce budworm and reduce larval dispersal loss (Carlson and others 1985). The changes in forest structure have shifted the spatial and temporal pattern of budworm outbreaks. Although frequency of outbreaks has not changed since the pre-settlement era (20 to 33 year intervals), recent outbreaks of western spruce budworm have become more extensive (Swetnam and Lynch 1989), more severe, and synchronous (Swetnam and Lynch 1993). Activity of western spruce budworm typically increases with periods of high moisture and decreases in drier periods (Swetnam and Lynch 1993). Outbreaks can cause substantial mortality and create increased surface fuel loading in a short period. The increased activity during wetter periods, in combination with greater fine fuel production during the same time, could increase the fire hazard in subsequent drier years. Currently, western spruce budworms are attacking trees on the Dixie, Fishlake, and Manti-LaSal National Forests. The outbreak has increased in acreage since 1999 with up to 6,014 ha (14,861 acres) impacting the Dixie National Forest in 2004 (Forest Health Protection 2000; Matthews and others 2005).

The increased densities in susceptible host species has likely increased the incidence of Armillaria root disease, annosus root disease, and dwarf mistletoe in southern Utah mixed conifer forests as it has in other areas of the western United States (Hagle and Gohenn 1988). This increase in infection has lead to increased susceptibility to insect attack and greater surface fuel loadings due to mortality of susceptible trees.

Ungulates—The increased density of wild and domestic ungulates has significantly reduced the herbaceous vegetation in mixed conifer stands. Aspen regeneration in many areas of the Rocky Mountains has been severely reduced due to herbivory by ungulates (Hart and Hart 2001; Kay 2001a; Kay 2001b; Romme and others 1995), although few studies have addressed this issue in Utah. Because aspen stands are natural firebreaks, the reduction in aspen stand coverage has increased the probability of larger fires in mixed conifer landscapes. Furthermore, the lack of fine fuels in the understory to carry surface fires has lengthened the fire frequency and has allowed greater quantities of larger diameter woody fuel accumulation.

Selective Harvesting and Fire Exclusion—Selective harvesting and fire exclusion has caused dense, multi-storied Douglas-fir and white fir to largely replace the ponderosa pine component in mixed conifer stands. As a result, mixed conifer forests are now very susceptible to western spruce budworm, root disease, bark beetles, dwarf mistletoe, and stand-replacing fires (Swetnam and Lynch 1989). The higher densities and contiguity of forests has led to large regional insect outbreaks that are more severe than in the past. Larger outbreaks will result in continued changes in forest structure, composition, and function, including creation of openings, depletion of large diameter trees, and an increase in fire hazard due to greater surface fuel accumulations (Wilson and Tkacz 1995). With continued fire exclusion in mixed conifer forests, surface and ladder fuels will continue to coalesce with crowns of overstory trees. This change in vertical fuel structure will further increase the probability of severe stand replacement crown fires.

Fire Regime Condition Classes

Under pre-settlement fire regimes, stocking of mixed conifer landscapes ranged from components containing shade-intolerant, early-successional species, such as aspen and pine, to those containing shade-tolerant, late-successional species, such as spruce and fir. Mixed severity fire regimes operating at a variety of scales were constantly affecting successional pathways creating landscapes of infinite diversity and character.

Fire Regime Condition Class 1 (FRCC 1)

The majority of FRCC 1 mixed conifer landscapes contains early successional species such as aspen, Douglas-fir, and ponderosa pine. They are more diverse in spatial structure, containing openings and only patches of mature, late successional species. Vertical structure is also more diverse. Some patches are very open and late successional species are represented predominately in seedling and sapling size class. Understory vegetation is more abundant in these patches. These landscapes have experienced recent fire within the period of mean fire return interval. Surface fuel loads are patchy due to the nature of mixed severity fire regime. Heavier woody fuels are present in areas that experienced crown fire, while lighter woody fuel loadings are present in areas experiencing surface fire. Continuity of surface and ladder fuels is diverse due to the patchiness of fire-induced mortality.

Fire Regime Condition Class 2 (FRCC 2)

Mixed conifer forests in FRCC 2 have overstories dominated by Douglas-fir and ponderosa pine, but the regeneration layer and midcanopy is dominated by white-fir, Engelmann spruce, and subalpine fir. Aspen patches are maturing and smaller in extent due to invasion by conifers. Understory herbaceous composition is not as diverse as in FRCC 1. These forests have been impacted by fire suppression and exclusion, logging, and grazing, but not to the extent as observed in FRCC 3. Surface fuels and ladder fuels are more contiguous throughout the landscapes with fewer openings present. Surface fuel loadings are dependent on management activities. Areas that have not been harvested contain heavier surface fuel loadings due to large diameter snags that have fallen and accumulated on the forest floor. Many of these large diameter snags are not present in stands that were managed since settlement due to fuel management activities during harvest.

Fire Regime Condition Class 3 (FRCC 3)

We suggest the majority of mixed conifer forests are currently in FRCC 3. Since settlement, the impacts of fire suppression and exclusion, logging, and heavy grazing have brought extensive changes to mixed conifer forests in southern Utah. These activities have converted forests that once were dominated by a mixture of ponderosa pine, aspen, and Douglas-fir trees into forests dominated mostly by white-fir, Engelmann spruce, and subalpine fir. Prolific regeneration of these shade-tolerant species has created abundant ladder fuels, which allow surface fires to travel into the crown fostering crown fires. Mixed conifer forests are higher in basal area and density than pre-settlement forests, which have lead to higher susceptibility to insect and disease attack and a substantial increase in surface fuel loads. Natural fuel breaks once provided by aspen are now decreasing in size due to aging aspen groves and the lack of aspen regeneration. Fire regimes have departed from historical frequencies by up to three or four intervals (Heyerdahl and others 2005; Heyerdahl and others 2006). The combination of all these factors may lead to large landscape crown fires instead of mixed mode fires (Fulé and others 2003; Fulé and others 2004).

Recommended Treatments

The historical fire regimes of mixed conifer forests are more complex than that of nearly pure ponderosa pine forests or spruce-fir forests. Because mixed conifer forests are made up of a number of species, the mix and distribution of each these species across landscapes will determine what approach is needed to restore them to a proper functioning condition. The goal in most mixed-conifer forests should be to return stands back to earlier stages of succession and increase the age and spatial diversity within most landscapes. Our goal should be to remove the more shade-tolerant species in the understory and leave species in the overstory, such as ponderosa pine and Douglas-fir, which can withstand periodic fires. As with spruce-fir, some complete removal of portions of the landscape may be needed to provide natural firebreaks and reintroduce early successional species. If an aspen component is present, the goal may be to increase aspen regeneration to rejuvenate the aspen stands. Techniques outlined in the aspen chapter of this document should be used to accomplish those goals (Bartos, this volume). Such fuel breaks can be accomplished by cutting or prescribed burning to create openings in the forest. Aspen may re-occupy those areas, creating effective fuel breaks without creating permanent openings. Because Douglas-fir and ponderosa pine are not as susceptible to windthrow as Engelmann spruce, subalpine fir, or white fir, these species should be left whenever a thinning or biomass reduction operation occurs within stands. This also helps set succession back to an earlier stage by allowing Douglas-fir and ponderosa pine the light and growing conditions they need to regenerate. Development of stand structures should focus on creating forests with basal areas and densities that occurred historically, which includes a component of large-diameter trees that can withstand wildfires. Historic mixed conifer forests in the Grand Canyon and the Kaibab Plateau in nearby Arizona averaged 190 to 265 trees/ha (77 to 107 trees/acre) with basal areas ranging between 17.6 to 28.5 m^2/ha (77 to 124 ft^2/acre) (Fulé and others 2003, 2004, 2006). Ponderosa pine dominated the basal area in each stand (Fulé and others 2004, 2006) or shared dominance with Douglas-fir and white fir (Fulé and others 2003). In historic southern Utah forests, trees greater than 20 cm (8 inches) ranged from 0 to 396 per ha (0 to 160 per acre) (Heyerdahl and others 2006).

Once a harvest, non-commercial thinning, or mechanical mastication treatment has removed the undesirable trees, a prescribed burn can be used to clean up surface fuels and ensure that few white fir or subalpine fir seedlings survive. Existing fir seedlings will soon overwhelm the understory and quickly grow into the overstory if this is not done. Experience in southwestern Colorado on the San Juan National Forest has shown that due to their rapid growth in partially shaded conditions, understory subalpine fir saplings can dominate the canopy within 25 years (data on file, RMRS, Wayne Shepperd).

After treatment, growing stock should be less than or equal to 30 percent of maximum stand SDI when calculated on a diameter class basis. The most shade-tolerant species should be removed first, leaving the less shade-tolerant, most fire resistant species, such as ponderosa pine and Douglas-fir. Treatments should not be uniform across the landscape, but should range from complete removal of all trees in some patches, thinning in others, and leaving intact forested patches to emulate the effects of mixed-severity fire regimes that created the natural spatial diversity of mixed conifer forests. The 30 percent maximum SDI stated above should therefore be an average of stocking across the entire landscape. The tools and techniques for developing treatment prescriptions discussed earlier in the ponderosa pine forests section of this chapter also can be used in mixed conifer forests. However, the diverse nature of mixed conifer forests may require that the prescriptions be developed at the landscape scale rather than the stand scale. Any mechanical removal of trees will result in some scarification of the forest floor that will provide ideal seedbeds for the recruitment of new seedlings. Subsequent thinning or prescribed burning will periodically be needed to keep such regeneration from restocking and overstocking the forest. It is important to keep in mind that silvicultural prescriptions must include a flexible time table of future thinnings and prescribed burn treatments.

Biomass reduction thinnings do not necessarily have to be evenly spaced. The same goals can be accomplished through irregular spacing and grouping of trees, provided that the groups are not contiguous to larger trees in a manner that they would provide live fuel ladders. Using an irregular spacing allows biomass fuel reduction goals to be met while retaining wildlife habitat attributes, such as hiding cover, and the juxtaposition of various vegetation structural stages classes within the forest canopy (Reynolds and others 1992), thereby leading to conditions more similar to pre-settlement.

The authors recognize that not all the acreage in southern Utah will be accessible for mechanical treatment. Prescribed burning is the most feasible method of creating diversity within many landscapes. In addition to prescribed under-burning, prescribed crown fire and wildland fire use can be used in late seral, mixed conifer, and some spruce-fir forests where aspen is present. Burning landscapes with a mixed severity fire that includes stand replacement patches can achieve the diversity that was present historically. When repeated through time, such burns can essentially recreate the vegetation mosaics that once existed within southern Utah mixed conifer landscapes. Recent wildfires in southern Utah that have resulted from attempts at such burns should not prevent their future use. Such setbacks are to be expected as we learn how to safely re-introduce fire into these complex systems.

Using prescribed under-burning to reduce surface fuel loadings will likely create a conflict between achieving the fuel reduction goals, maintaining healthy forests, and achieving a sound silvicultural treatment. If the prescribed burns occur under dry enough conditions such that the surface fuels are completely consumed in the understory, the fires are likely to be so severe that they will harm the roots of the living trees and plants. This has been demonstrated in ponderosa pine (Sackett and Haase 1998) and can be expected to be true for other shallow-rooted species such as Douglas-fir, aspen, and spruce. Avoidance of root damage is best accomplished by burning in late spring when soil moisture is at its highest and larger diameter woody fuels are not completely consumed because of high moisture content (Shepperd 2004). Even though the larger diameter woody fuels and some live fuels are not completely consumed, such burns can apparently still have a beneficial effect. A portion of the recent Hayman fire in Colorado in June 2002 was slowed and essentially stopped by the Polhemus prescribed burn that occurred in Fall 2001 (Martinson and others 2003). Although there were ample live crown fuels left following the earlier prescribed burn, the reduction of surface fuels was such that it stopped the Hayman crown fire within the treated area. While we recognize that this would not occur in all instances, even an incomplete prescribed burn has some beneficial effect on preventing or slowing the spread of crown fires. We believe that combining mechanical treatments where possible with prescribed burning will allow us to better choose the portions of the landscape where we can expect prescribed crown fires to be contained.

In summary, any silvicultural technique that will result in a diversity of mixed conifer successional stages within southern Utah landscapes will probably be beneficial in reducing the risk of large landscape stand replacing crown fires in mixed conifer forests. Removing portions of forests or trees from landscapes will increase the amount of forage that is produced on those landscapes for wildlife and livestock. This is especially important in southern Utah where livestock numbers have not been reduced as landscapes have been filled in with trees and understory production has decreased. Reduction of forest biomass from all forests within southern Utah is also likely to have a beneficial effect on the water balance within those landscapes (Shepperd and others 1992). We believe that good wildfire management is also good ecosystem management. It would be ideal to treat every acre to reduce the risk of wildfires operating outside the historical range of variation; however, the reality is that not every acre will be treated. Instead, we should prioritize treatments in some areas and allow wildfires to burn in others.

Engelmann Spruce and Subalpine Fir Forests

Mixed Engelmann spruce and subalpine fir forests comprise the upper extent of forest vegetation in southern Utah, occupying the coldest and wettest sites in the altitudinal continuum of ecologic conditions in the area. Precipitation regimes in these forests are dominated by snow, which can occupy these sites for 6 to 8 months of the year. Spruce-fir forests can exist on-site for extremely long periods, on average 500 to 600 years (R. Alexander 1987), with reports of even longer periods (Brown and others 1995). Harsh climates and short growing seasons result in infrequent, but large-scale disturbances including fire, insect attacks, wind, and avalanches, which historically interacted to create coarse-scaled mosaics of different aged patches on the landscape (Baker and Veblen 1990).

Historical Conditions

Historical descriptions of spruce-fir forest structure and density are limited. One study on the North Rim of the Grand Canyon suggests that compared to contemporary forests, historic spruce-fir forests in the southwest were significantly less dense (16 to 24 percent) and had lower basal area (36 to 46 percent) with densities about 150 trees/ha (60.7 trees/acre) and basal areas only 10 m^2/ha (43.5 ft^2/acre) (Fulé and others 2003). Reconstructed ca. 1860 spruce-fir forests on the Fishlake and Dixie National Forest show increases in density similar to the Grand Canyon site (Heyerdahl and others 2006). Most of the recruitment has occurred in the past 100 years suggesting these trees are smaller in diameter and height than pre-settlement forests (Heyerdahl and others 2006). A 1911 survey on the Manti-LaSal National Forest indicated that spruce reached 24.4 m (80 ft) tall and 61 cm (24 inches) in diameter, but the average size was 18 m (60 ft) tall and 46 cm (18 inches) in diameter (Ogle and DuMond 1997). Spruce-fir forest structure contained a variety of age classes and successional stages in varying patch sizes (Ogle and DuMond 1997).

Fuel structure under spruce-fir forests often promotes stand replacement fires. Surface fuel loads were probably much higher than those found at lower elevation montane forests due to slower decomposition rates (Uchytil 1991). Needles are small and fine and form a compact fuel bed in which fire spreads slowly and fuel beds accumulate under the narrow-crowned trees (Uchytil 1991). Large diameter woody debris and snags are also prevalent in spruce bark beetle outbreak areas. Mielke (1950) reported that 84 percent of spruce bark beetle killed trees were still standing after 25 years, mostly in large diameter classes. Relatively few trees fell within the first 10 years after beetle attack, but trees did fall after being infected with basal and root rots (*Fomes pinicola*).

Spruce-fir forests are predominately subject to long-interval stand replacement fire regimes rather than the more frequent low-intensity surface and mixed-mode fires occurring in lower elevation southern Utah coniferous forests. Mean fire return intervals (MFRI) in spruce-fir do appear to vary by region. Arno (1980) estimates MFRI of 50 to 130 years in the Northern Rockies. Veblen and others (1994) reported a MFRI of 202 to 241 years in northwestern Colorado, while Peet (1981) reported a MFRI of 200 to 400 years for the Front Range of Colorado. In northern Utah, the overall MFRI was 41.3 years, however, this average is somewhat deceptive, as no fires occurred between 1700 and 1855 (pre-settlement), followed by a 9-year fire interval from 1856 to 1909 (settlement), and then a 79-year interval from 1910 to 1988 (suppression period). White and Vankat (1993) reported a MFRI of 70 to 250 years on the North Rim of the Grand Canyon. Lang and Stewart (1910) reported that the North Rim contained "vast denuded areas, charred stubs and fallen trunks." The old fires extended over large areas at higher altitudes covering several square miles on either side of Big Park, with numerous smaller irregular areas over the remainder of the park (Lang and Stewart 1910: 18 to 19 cited in Fulé and others 2003).

Fires were predominately of three types: 1) lightning-ignited fires that consumed individual trees or small patches of forests (the most common type), 2) crown fires that killed most overstory trees as well as saplings and seedlings, (Arno 1980; Baker and Veblen 1990), or 3) patchy fires that burned as surface fires for a short distance and then burned the overstory trees for a short distance resulting in a coarse-grained mosaic of dead trees or open areas with alternating patches of surviving trees (Baker and Veblen 1990).

Several morphologic and ecologic factors of Engelmann spruce and the true firs contribute to the long-interval stand replacement fire regimes. Spruce and fir species have thin bark that is not well insulated from fire damage, which allows the trees to easily be killed by low intensity fire. The species have long, dense crowns that typically reach close to the ground. This creates abundant live fuel ladders that allow surface fires to climb into the upper forest canopy. This is especially prevalent in older stands and where spruce bark beetle outbreaks have initiated a regeneration response (due to openings in the canopy) creating multi-storied structures. The accumulation of snags and heavy surface fuels resulting from beetle outbreaks persist for a long period of time (Brown and others 1998), and further contributes to a stand replacement fire regime.

Long fire return intervals in spruce-fir forests are primarily a result of their occurrence at high elevations that receive more precipitation and cooler temperatures throughout the growing season. In these snow dominated precipitation regimes, it is difficult for fire to burn in a normal year. Following snowmelt in the spring, there is abundant soil moisture throughout the growing season and cool temperatures keep understory fuels moist and hard to ignite. Therefore, fires are more likely to occur in the fall, late summer, or after an unusually dry winter instead of in the spring.

Windthrow is second to fire as the landscape-wide disturbance affecting spruce-fir forests. The amount and degree of damage varies depending upon the wind event, stand structure, species composition, topography, and stand's fire history (Kulakowski and Veblen 2002; Veblen and others 2001). For instance, older spruce trees are shallow-rooted and prone to blowdown (R. Alexander 1987); however, young post-fire stands are less affected due to shorter trees, less canopy gaps, and often a greater component of aspen (Kulakowski and Veblen 2002). With severe blowdowns, huge surface fuel loadings can result that increase the severity of any fire occurring in the area for years after the event. In addition, outbreaks of spruce bark beetle are often triggered by blowdowns (Schmid and Frye 1977).

Spruce bark beetles play a major role in the development and maintenance of high elevation coniferous forests in Utah. The beetles, in association with fire, help maintain a variety of successional stages and age classes of spruce-fir forests across the landscape. Endemic populations live in windthrown trees and probably have little effect on forest dynamics. In areas with an extensive windthrow event, however, ample food supplies allow rapid buildup of beetle populations (Massey and Wygant 1954; Wygant 1958). These epidemic levels of spruce bark beetle populations are reported to have occurred on average every 116 years (since 1700) in presettlement spruce-fir forests of central Colorado (Baker and Veblen 1990; Veblen and others 1994). Epidemic populations attack mature, larger diameter (>46 cm [18 inches]) Engelmann spruce trees (Schmid and Frye 1977) and as a result, average tree diameter, height, age, and densities of stands are reduced, dominance in basal area shifts from Engelmann spruce to subalpine fir, and accelerated growth of residual trees are observed (Schmid and Frye 1977; Veblen and others 1991).

Ungulates were likely to have had minimal influence on subalpine forests prior to European settlement. The presence of large predators probably kept deer and elk populations at much lower levels than today and the lack of domestic livestock further insured minimal impact.

Disturbances existing prior to settlement affected community characteristics of subalpine forests and their position on the landscape in several ways. High elevation fires set back succession from forest to meadow and maintained existing meadows. Some fires in forest communities near timberline created herb or shrub dominated seral communities, which are slow to regenerate back to spruce (Huckaby 1991). Fires occurring at the lower elevation interface with mixed conifer forests allowed those sites to regenerate to aspen and Douglas-fir, thus maintaining a wider range for those forests than occurs today. Fires that occurred in spruce-fir beetle-killed or windthrow areas were likely to have been extremely severe, insuring that those areas remained open for long periods of time.

Current Conditions

Spruce-fir forests have expanded into the mixed coniferous forests, as well as into high elevation meadows of southern Utah (Fulé and others 2003; Heyerdahl and others 2005; Heyerdahl and others 2006; USDA Forest Service, Fishlake National Forest 1999; White and Vankat 1993). Expansion into the lower elevations is a result of succession from aspen forests to mixed conifer forests due to fire suppression and because aspens provide suitable habitat for the establishment of shade-tolerant conifers (Shepperd and Jones 1985). Expansion into higher meadows over the past 100 years is largely due to grazing by domestic animals or wildlife, which scarified seedbeds and reduced competition between tree seedlings and the herbaceous community (Allen 1989; Moir and Huckaby 1994). In addition, if climatic warming is occurring, it would increase the length and warmth of the growing season, possibly improving seedling survival (Moir and Huckaby 1994; Moir and others 1999). Furthermore, fire suppression has allowed seedlings to establish on the edges of meadows and reduce the extremely high soil moisture making it easier for additional seedlings to establish in the center of the meadows.

The structure of spruce-fir forests in southern Utah is predominately uneven-aged (fig. 5) (Hanley and others 1975; Mielke 1950; O'Brien and Brown 1998; O'Brien and Waters 1998; O'Brien and Woudenberg 1998; Pfister 1972). Engelmann spruce is the major species, followed by subalpine fir and aspen (Bradley and others 1992; Fulé and others 2003). Pure Engelmann spruce stands and spruce-fir forests (where spruce and subalpine fir are codominant) consist of all ages, although the majority of these trees are 51 to 150 years old (O'Brien and Brown 1998; O'Brien and Waters 1998; O'Brien and Woudenberg 1998). In addition, there are some Engelmann spruce trees 151 to 250 years old in the Dixie National Forest (O'Brien and Brown 1998). Surveys on the Markagunt and Aquarius plateaus in southern Utah in the early 1970s revealed that subalpine fir stocking was uneven-aged, but not all-aged. A 50- to 70-year old prolific regeneration component existed in the understory, which corresponded with a spruce beetle outbreak in the 1930s. Subalpine fir was also present in the 70- to 130-year old age class indicating its ability to maintain itself under the Engelmann spruce canopy for long

Figure 5—Uneven-aged Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) stand with an understory dominated by heartleaf arnica (*Arnica cordifolia*) and tall bluebell (*Mertensia arizonica*).

periods. The predominance of subalpine fir seedlings in the understory of southern Utah forests (Hanley and others 1975) is indicative of its ability to successfully reproduce on duff-covered seedbeds, but spruce's average longevity over subalpine fir keeps it dominant in the overstory. Fulé and others (2003) found 73 percent of their plots on the North Rim had fire-initiated groups of trees dominated by Engelmann spruce dating back to 1788.

Diameter distributions for Engelmann spruce and subalpine fir in southern Utah are not available, but can be found for the entire state of Utah (O'Brien 1999). On a landscape level, the diameter distribution of Engelmann spruce and spruce-fir forests exhibit an uneven-aged structure with a large numbers of trees less than 12 cm (5 inches) in diameter.

Fuels/biomass data collected by Fulé and others (2002c) from North Rim spruce-fir forests report duff depths of 2.3 cm (0.9 inches) and litter depths of 0.8 cm (0.3 inches) with downed wood loadings of 141 Mg/ha (63 tons/acre), the majority of which was split between 1000-hour rotten and sound fuels. Shrubs and small trees predominate in southern Utah spruce-fir forests. *Abies lasiocarpa/Ribes montigenum* habitat type occupies approximately half of the spruce-fir in Utah (Pfister 1972). Annual herbaceous production ranged from 1,201 kg/ha (1,072 lbs/acre) to 2,541 kg/ha (2,267 lbs/acre) in similar communities in central Utah with forbs contributing to over 80 percent of the production (Ralphs and Pfister 1992).

Due to the long history of water yield research in subalpine forests (Leaf 1975; Troendle and others 1988), more is known about the hydrologic function of spruce-fir forests than any other western forest type. Streamflow and water yield in these snow-dominated precipitation regimes is driven by the density of the forest, which principally affects interception of snow by the canopy and subsequent sublimation back to the atmosphere (Troendle and others 1988). Therefore, any disturbance that removes trees from the forest will affect the hydrologic water balance of the system. Potential effects due to fire have been studied in a subalpine dominated forest in the Shoshone National Forest in Wyoming after a fire (Troendle and Bevenger 1996). Since most high elevation watersheds have peak annual discharge at snowmelt, they found peak annual discharge unchanged by the fire. However, water yields increased 25 percent on the burned watershed. Suspended sediment production in terms of concentration was two times greater in the burned watershed and suspended load was four times that of the unburned watershed. The highest concentration occurred during an individual summer storm event. These findings indicate that large-scale crown fires in southern Utah spruce-fir forests could significantly alter the water balance of the system until sufficient vegetation recovery occurs to reduce sediment transport.

Current Disturbances

Post-settlement spruce-fir forests are still subjected to the same disturbances observed in pre-settlement times, mainly stand-replacing fires, blowdowns, and insect attacks. However, new disturbances such as grazing, logging, increased fire ignitions, and fire suppression have emerged in these high-elevation coniferous forests.

Ungulates—Between the time of settlement and the early 1900s, settler activities impacted spruce-fir forests. These activities continue to influence the disturbance regime of present times. First, the introduction of livestock grazing in the high elevation meadows had several impacts on the ecosystem. During early settlement, overgrazing led to loss of topsoil that created unproductive areas because no vegetation could reestablish. This led to several summer-time floods and sheet erosion (T.Alexander 1987; Keck 1972; Ogle and Dumond 1997). Although overgrazing reduced fine fuel loads, the presence of sheepherders and loggers actually increased the fire frequency in some spruce-fir forests (Bird 1964; Roberts 1968; Wadleigh and Jenkins 1996) until the Forest Service implemented fire suppression after 1910. Early logging of spruce-fir forests was limited to accessible areas and only the best species were removed leaving the subalpine fir (Ogle and Dumond 1997). Today, we are left with large acreages of young, small diameter subalpine fir and Engelmann spruce.

Spruce-fir forests continue to be heavily use as summer range in southern Utah. The increased acreage and density of spruce-fir discussed earlier translates to a decrease in overall forage available on allotments in spruce-fir forests. Furthermore, species composition of herbaceous plant communities continues to change as a result of preferential selection of specific species by the livestock. Continuation of traditional livestock grazing rates in these allotments only puts more pressure on a dwindling grazing resource and reduces potential for fire ignition.

Fire—Although fire intervals for spruce-fir forests are typically several hundred years, a portion of spruce-fir forest acreage would probably have burned in years of severe drought if not for fire suppression efforts. These fire suppression effects have only stalled the inevitable large stand-replacing fire. Large stand-replacing fires during drought years are normal for spruce-fir forests. One notable fire is the Outlet fire, which occurred within Grand Canyon National Park in May 2000. The extreme drought and occurrence of numerous other fires in subalpine forests throughout the western United States in recent summers are strong evidence that the potential exists for more large scale fires to occur in southern Utah.

Insects and Disease—Insect activity has also dramatically increased in southern Utah in recent years (fig. 6). Spruce bark beetle populations have been at epidemic levels since 1991 on the Dixie National Forest

Figure 6—Engelmann spruce (*Picea engelmannii*) at Cedar Breaks National Monument, UT, killed by recent spruce beetle (*Dendroctonus rufipennis*) activity.

and since 1989 (Knapp and others 1991) on the Manti-LaSal National Forest (Knapp and others 1989). Large areas of the Dixie, Fishlake, and Manti-LaSal National Forests have experienced severe disturbances caused by spruce bark beetle. Most spruce trees greater than 15 to 20 cm (6 to 8 inches) were killed during these outbreaks (Dymerski and others 2001; Forest Health Protection 2000; Matthews and others 2005). For example, 73 percent of spruce trees greater than 12 cm (5 inches) in diameter on the Manti-LaSal National Forest were killed over a period of about 5 to 7 years – a 47 percent decline in the spruce component (Dymerski and others 2001; Samman and Logan 2000). From 2000 to 2004, spruce bark beetles have killed over 366,000 trees on over 40,469 ha (100,000 acres) of southern Utah National Forests (Forest Health Protection 2000; Matthews and others 2005). Hazard ratings indicate that 62 percent of spruce-fir forest types in the Manti-LaSal National Forest, 45 percent in the Dixie National Forest, and 42 percent in the Fishlake National Forest are at moderate to high risk of attack by bark beetles (O'Brien and Brown 1998; O'Brien and Woudenberg 1998; O'Brien and Waters 1998).

Mortality of subalpine fir trees in the National Forests in southern Utah is also on the increase (Matthews and others 2005). From 2000 to 2004, western spruce budworm had defoliated over 3,287 ha (8,000 acres) on the Fishlake National Forest and over 24,281 ha (60,000 acres) on the Dixie National Forest (Forest Health Protection 2000; Matthews and others 2005). Defoliation on the Manti-LaSal National Forest remains low with approximately 243 to 485.6 ha (600 to 1,200 acres) per year since 2002 (Matthews and others 2005). Decline and mortality of subalpine fir resulting from a complex of secondary biotic agents have affected over 24,281 ha (60,000 acres) in the National Forests of southern Utah from 2000 to 2004 (Forest Health Protection 2000; Matthews and others 2005). These secondary agents include twig beetles, woodborers, engraver beetles, secondary bark beetles, root diseases, cytospora canker, and fir broom rust (Forest Health Protection 2000).

Fire Regime Condition Classes

The long fire return interval in spruce-fir forests and the wide range of possible forest structure conditions that may naturally develop between stand replacing fires pose somewhat of a dilemma in assigning FRCC classes to these forests. Because most existing forest conditions fall within the past range of natural variability, we believe most spruce-fir forests could be classified as FRCC 1. What may move these forests into FRCC 2 or 3 is the distribution of forest structural stages across the landscape. We believe the continuity of fuels at the landscape scale is what differentiates the FRCC class in spruce-fir forests. Fire exclusion and subsequent ingrowth into gaps have resulted in more contiguous surface and crown fuels at the landscape scale. Although these surface and crown fuel accumulations are not out of the historical range on a stand level scale, at landscape scales, fires would likely be larger than what occurred in the past. Therefore, we believe that FRCC 2 and 3 landscapes would be those that are currently completely forested as opposed to being partially forested 200 to 300 years ago. A hundred or more years of fire exclusion in spruce-fir forests since settlement has also likely altered the distribution of age classes over the landscape, which has moved a greater portion of forests toward fuels conditions associated with older forests that are described below.

Fire Regime Condition Class 1 (FRCC 1)

Spruce-fir forests classified as FRCC 1 span a variety of successional stages and age classes distributed across the landscape. These forests include young post-fire forests, forests after insect epidemics, middle-aged and mature multi-storied forests, and overmature old growth forests. Each stage is a result of past disturbance history and is dispersed on the entire landscape intermixed with open meadows.

Young stands, less than 100 to 150 years old, established after a stand replacement fire. Such forests are clumpy with canopy gaps and openings between stands. Low surface fuel accumulations and a rich and diverse understory exist in the openings between the clumps. Heavy surface fuel loadings occur within the clumps and consist of partially decomposed and burned, fallen snags from the previous stand. However, these surface fuels are not contiguous.

As the canopy in these young forests begins to close, clumps begin to merge and expand. The outer edges of the clumps have younger fully crowned trees, but the inside of the clumps (approximately one ha [2.5 acres] in size) have relatively young trees with different diameter distributions (due to density). These forests do not have an appreciable buildup of surface fuels because mortality is low at this stage in their life cycle.

As the spruce matures, the stand is susceptible to disturbances such as windthrow, spruce beetle attacks, and disease. These disturbances help maintain the canopy gaps within the forest and create multi-storied stands. Subalpine fir also becomes an important component of the forest species composition as the spruce forest matures. Subalpine fir increases the presence of ladder fuels and increases surface fuel accumulation due to its short life span.

Overmature and old growth spruce-fir forests are subjected to repeated disturbances (fire, insects, and disease) that result in partial mortality and heavy surface fuel accumulation. These forests typically have fewer canopy gaps and contain a significant component of younger fir and numerous standing snags.

If the mature, overmature, or old growth spruce-fir forests have been recently attacked by spruce beetle, the majority of large diameter trees are dead. The accumulation of snags and heavy surface fuels resulting from beetle outbreaks persist for a long period of time (Brown and others 1998) and further contributes to a stand replacement fire regime. The dominant species of these forests changes from spruce to fir and creates abundant live fuel ladders that allow surface fires to climb into the upper forest canopy.

Fire Regime Condition Classes 2 and 3 (FRCC 2 and FRCC 3)

Spruce-fir forests classified as FRCC 2 or FRCC 3 consist of a landscape with contiguous forest stocking, a lack of meadows, and a lack of successional and age class diversity. These forests have species composition and fuel accumulation patterns that are not consistent with patterns observed after a natural disturbance. The degree of departure of these factors determines the FRCC classification. For example, the expansion of spruce-fir forests into meadows or the upper elevations of mixed conifer forests due to fire exclusion could be classified as FRCC 2. Multi-watershed contiguous forests that are mature and contain no canopy gaps or other natural features that could interrupt the continuity of fuels over an extremely large area might be classified as FRCC 3.

Past harvesting techniques impact FRCC characteristics on a stand level. If they were implemented on a large portion of the landscape, the FRCC classification could shift toward 2 or 3. Past harvests often removed the larger diameter spruce trees, which left the smaller spruce and fir trees behind and effectively changed the age structure and species composition of the forest. Similar structural changes occur after a spruce beetle infestation except there is a major difference in the post-disturbance fuel complex since all of the dead trees remain on-site. After a spruce beetle infestation, large diameter snags are present in the stand, eventually fall, and then contribute to the nutrient pool, surface fuel loading, and eventually to high intensity wildfire. Salvage logging removes the large diameter snags and effectively reduces large diameter surface fuel loadings and potential nutrient inputs. In any case, harvested stands would have higher loadings of fine fuels if activity fuels were not removed or treated. Modern whole-tree harvest methods alter fuels differently. Since all unmerchantable trees and tops are removed, surface fuels are not as heavy as in past harvests. Early logging of spruce-fir forests in Southern Utah was limited to accessible areas (Ogle and Dumond 1997), so fuels were not altered on all portions of landscapes.

Recommended Treatments

Although most existing spruce-fir forests in southern Utah are likely to be in FRCC 1, or possibly FRCC 2, losing key ecological components that define spruce-fir forests is still possible if extremely large patches of these forests are burned completely in a single stand replacement fire event. Spruce and fir are generally shade tolerant species that establish best in partial shade. They do not establish well or quickly in the full sunlight existing after a stand replacing fire. Because Engelmann spruce seeds require mineral soil to germinate, subalpine fir seedlings usually outnumber spruce seedlings in the understory of spruce-fir stands (Alexander and others 1984).

Successional pathways following fire can vary dramatically (Stahelin 1943). Since aspen acts as a nurse crop for young spruce and fir seedlings, if an aspen component is present, spruce-fir can recover in 100 to 150 years providing a residual seed source is present (Shepperd and Jones 1985). If aspen is missing and the burn is severe, a meadow opening is created that will take much longer to recover to spruce-fir. In high elevation forests, it takes over 40 years for spruce trees to reach dbh (1.37 m [4.5 ft tall]) under open conditions. This growing period increases on south and west facing slopes and decreases on north and east aspects where water is not a limiting factor (R. Alexander 1987). Spruce and fir seedlings are dependent on wind dispersal, so if fires are large in scale, it takes a long time for seedlings to establish and create a new forest. Effective seeding distance is four to five times tree height (R. Alexander 1987). Very severe fires that create large openings will result in a type conversion to meadow or aspen for hundreds of years. Moderate fires will have some spruce and fir survival, which will increase the reestablishment rate of spruce-fir into burned openings.

As mentioned above, past logging in spruce-fir forests will likely affect fuels and fire behavior. Increasing the spacing of trees and allowing more light to reach the understory will further affect fire behavior by increasing understory vegetation production and wind speeds to promote a warmer, drier microclimate. However, a counteracting effect of logging to decrease crown fuels would be the increased accumulation of woody fuels in the understory, which would increase the likelihood of high severity fires. Initially, any disturbance resulting in the partial removal of trees would likely decrease fire spread rate. However, the tendency of spruce-fir to develop a multi-storied structure following disturbance would eventually negate this effect.

A possible management goal in spruce-fir landscapes should be to allow stand replacement fires to occur while reducing the risk of entire landscapes succumbing to such fires. Techniques that could be used to accomplish this goal include thinning to reduce stocking of younger forests, complete cleanup of slash with whole tree harvests or mechanical harvesting, or using techniques such as block or strip cutting that add fuel breaks in the forested landscape. All of these techniques would break up the continuity of fuels and increase the chance that a portion of the forest would remain after a fire.

Use of prescribed fire in spruce-fir forests is less likely to be beneficial in this ecosystem because of the sensitivity of the species to damage by fire and the heavy accumulation of woody fuels and duff layers. Even cool, slow burning fires would damage the roots and bark of the trees (Bradley and others 1992) and would likely get into the crowns. Shelterwood systems, if used in some portions of the landscapes, would break up the age class structure, create gaps between crowns, and possibly lessen the likelihood that stand replacing fires would sweep over large areas. However, use of mechanical treatment is limited to areas that are roaded and accessible. Many spruce-fir forests are on steep slopes in inoperable terrain, or are in wilderness areas where management options are quite limited.

Spruce-fir Treatments at the Landscape Scale

Because spruce and fir are species that evolved with stand replacement fire regimes, underthinning or reduction of density is not recommended to protect individual stands from catastrophic replacement. Almost any spruce-fir forest is capable of burning. Even young forests can burn because their crowns generally reach down to the ground. Breaking up landscapes of spruce-fir into different successional stages is one way to increase the chances of spruce-fir survival at large scales. This would be akin to practicing uneven-aged management at a large scale, similar to what occurred naturally in the past. This would allow fires to burn through some areas and not others resulting in spruce-fir stands interspersed with pure aspen stands and openings such as alpine meadows or shrublands.

Openings created by harvesting large areas in spruce-fir forests will take decades to regenerate if they are greater than four to five times tree height wide (the effective seed distance) (R. Alexander 1987). Openings this large could be used as fuel breaks since it would take a long time to regenerate. Because of the heavy surface fuels under many spruce-fir stands, ensuring that openings be maintained in a meadow-like appearance might require broadcast burning recently cut units to reduce surface fuels and eliminate residual small trees. Ample evidence of the utility of this technique is evident from older, large spruce-fir cut blocks created in the central Rockies in the late 1950s to early 1960s. Some of these areas are still inadequately restocked 40 years after clearcutting and broadcast burning.

The creation of such openings should consider the landscape form so that a natural appearance is maintained and management activities do not appear to be geometric, irregular, or artificial in appearance. Areas selected for spruce-fir forest removal to create openings should be where former openings likely once existed. Examples include sides of valleys, bottom areas, and other areas that were likely to be unstocked at sometime in the past. One way to examine the landscape for spatial distribution patterns might be to use a technique similar to that of Moir and Huckaby (1994). They determined age cohorts on the Fraser Experimental Forest in central Colorado by identifying where past disturbances have occurred. Mangers could utilize such patterns in planning where to create fuel breaks in landscapes.

In some stands, the fir component of spruce-fir forests in southern Utah will affect fuel loadings and susceptibility to crown fire over time. Subalpine fir is much shorter lived than spruce and is likely to cycle in and out of spruce-fir stands through time (Aplet 1988; data on file RMRS). Assuming similar dynamics occur in southern Utah, spruce-fir forests are likely to have increased surface fuel loadings when the fir dies out and creates forests that are more susceptible to stand replacement fires. A similar relationship exists where aspen is a component of spruce-fir forests. As aspen dies out, stands accumulate large buildups of surface fuels. This creates areas that are more susceptible to stand replacement fire. However, most higher elevation spruce-fir forests are more likely to contain higher portions of spruce and be stocked with very old trees (Brown and others 1995), this indicates that fire is a rare occurrence at or near timberline. Removing the fir component from mixed stands is unlikely to reduce long-term fire risk. Unless all fir seed sources are removed, fir will quickly reestablish and contribute to a live fuel problem in the future.

Spruce-fir Treatments at the Stand Scale

We believe target stand structures are inappropriate in spruce-fir because FRCC is really applicable at the scale of landscapes instead of stands. However, management activities that add age classes and spatial diversity to the landscape would ultimately maintain FRCC 1 stands and benefit general ecosystem sustainability. Any management activity should leave a similar surface fuel complex that would naturally have occurred in a given stand configuration. The complete cleanup of slash mentioned above for fuel break creation would not apply here since these areas are part of the general forested matrix in which an almost infinite variety of fuel loadings could naturally occur. If the decision is made to treat fuels at the stand scale for reasons other than restoration (for example, protection of infrastructure), there are still limitations to what can be done in spruce-fir forests.

Thinning from below in mature spruce-fir forests, similar to what is done under even-aged shelterwood or seed tree regeneration in other forest types, is of limited usefulness in lessening the risk of catastrophic wildfire. The amount of material that can be safely removed without increasing risk of windthrow (R. Alexander 1987) is such that it is doubtful sufficient space could be maintained between crowns to appreciably lessen crown fire risk. At the same time, logging would likely increase the amount of downed woody fuel on the forest floor. Careful cleanup of logging slash, as well as monitoring of the overstory to remove subsequent windthrow, would be essential to keep surface fuel loadings at reasonable levels; however, these types of treatments are very costly. Since such techniques are normally used to regenerate spruce-fir forests, ample regeneration could be expected within a few years. The resulting increase in live fuel loadings would then require precommercial thinning to reduce risk of crown fire, further increasing the cost.

Uneven-aged management that maintains overall health and vigor of spruce-fir forests by periodically reducing stocking levels through time seems to be a reasonable way to meet a variety of ecosystem objectives for spruce-fir forests. Although nothing is likely to prevent crown fire, uneven-aged management can prevent excessive mortality and reduce both surface and crown fuels from levels that currently exist in many stands.

Summary

Pre-settlement coniferous forests of southern Utah were influenced by various disturbance regimes, including fire, insects, diseases, and herbivory, which maintained a heterogeneous landscape structure consisting of a diversity of successional stages and vegetative types. The combined effects of fire suppression, logging, and grazing have altered the extent, location, and structure of these forests. Identifying pre-settlement disturbance regimes and forest structures can serve as a reference to return the landscape to a more ecological sustainable system.

It is imperative that managers understand that each forest type in southern Utah has coevolved under different disturbance regimes that shaped their structure and community dynamics. Subsequently, hazardous fuel reduction and restoration treatments should differ for each coniferous forest type. Treatment area prioritization should focus on the ponderosa pine and mixed conifer forests types where fire regimes and vegetation attributes have been significantly altered from their historical range of variability. These areas will require moderate to high levels of mechanical restoration treatments before fire can be reintroduced to restore the historical fire regime. In contrast, fire regimes in spruce-fir forests have not been drastically altered from their historical range of variability and the risk of losing key ecological components is low to moderate. Therefore, hazardous fuel reduction and restoration treatments in the spruce-fir forests should be of low priority.

In ponderosa pine forests, treatments should focus on converting to uneven-aged management, reducing or removing shade tolerant conifers and oak, and reintroducing frequent prescribed surface fires. Mixed conifer forest treatments should focus on reducing the amount of shade tolerant species and leaving more fire-resistant tree species such as ponderosa pine and Douglas-fir. Mixed conifer forests should maintain a diversity of vegetation successional stages and age classes juxtaposed to each other. Reintroduction of prescribed fire should not be limited to under-burning since historical fire regimes included some crown fire in mixed conifer ecosystems. Spruce-fir forest treatments should focus on maintaining a landscape of different age structures, successional stages, and fuel breaks to lessen the risk of entire landscapes burning in one event. Fire suppression in spruce-fir forests should be tempered to allow some forests to burn.

To establish and continue a realistic restoration strategy, we must maintain the fire disturbance cycle, whether with prescribed burning or wildfire. The end result of any treatment is to create a heterogeneous landscape structure consisting of a diversity of successional stages and vegetative types that if subjected to wildfire, would retain key ecosystem components.

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