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Fire Ecology and Management of the Major Ecosystems of Southern Utah

Sharon M. Hood and Melanie Miller, Editors

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Abstract

This document provides managers with a literature synthesis of the historical conditions, current conditions, fire regime condition classes (FRCC), and recommended treatments for the major ecosystems in southern Utah. Sections are by ecosystems and include: 1) coniferous forests (ponderosa pine, mixed conifer, and Engelmann spruce-subalpine fir), 2) aspen, 3) pinyon-juniper, 4) big and black sagebrush, and 5) desert shrubs (creosotebush, blackbrush, and interior chaparral). Southern Utah is at the ecological crossroads for much of the western United States. It contains steep environmental gradients and a broad range of fuels and fire regimes associated with vegetation types representative of the Rocky Mountains, the Great Basin, Northern Arizona and New Mexico, and the Mohave Desert. The Southern Utah Demonstration Area consists of contiguous state and federal lands within the administrative boundaries of the Bureau of Land Management (BLM), Fishlake and Dixie National Forests, National Park Sevice, and State of Utah, roughly encompassing the southern 15 percent of Utah (3.24 million ha). The vegetation types described are similar in species composition, stand structure, and ecologic function, including fire regime to vegetation types found on hundreds of millions of hectares in the 11 western states.

Key Words: Fire regime condition class, disturbance, fire ecology, fuel treatment

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Editors

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Contents

Chapter 1

Introduction

Sharon M. Hood, Donald Long, Melanie Miller, and Kevin C. Ryan

Background

Many areas throughout the United States are facing the triple threat of increasing fire severity, residential growth in areas prone to wildland fire, and suppression costs and losses. In addition, substantial changes are occurring in the way we plan and implement management on federal lands relative to use of wildland fire, prescribed fire, and mechanical fuel management. Past emphasis in fire management has been on wildfire suppression and prescribed fire in support of other resources such as hazard reduction and site preparation in harvested areas and wildlife habitat improvement. Federal financial support has only recently supported the large-scale use of prescribed burning and mechanical fuels treatments to reduce unnatural fuel accumulations in non-wilderness areas.

The Southern Utah Fuel Management Demonstration Project was an effort to develop, evaluate, and compare methods to incorporate wildland fuels management into landscape scale land use planning processes for approximately 5.3 million ha (13 million acres) of southern Utah and 0.8 million ha (2 million acres) of northern Arizona (Ryan and Long 2004). This area was chosen because it is at an ecological crossroads for much of the western United States. It contains steep environmental gradients and a broad range of fuels and fire regimes associated with vegetation types representative of the Rocky Mountains, the Great Basin, and the Mohave Desert. The project developed GIS data layers for fuels, vegetation, and terrain that provided the inputs necessary to conduct fire behavior, fire effects, and succession modeling analyses. Several fuel treatments were also implemented at a variety of scales in the project area to demonstrate the use of fire behavior models for fuel treatment planning (Long and others 2003; Mathews 2003; Stratton 2004).

Project Objectives

This literature synthesis provides an ecological context for the larger Southern Utah Fuel Management Demonstration Project. The synthesis reviews the pertinent literature to address 1) historical conditions, 2) current conditions, 3) fire regime condition classes, and 4) recommended treatments for each of the major ecosystems found in the Southern Utah study area. Sections are by ecosystems and include: 1) coniferous forests (ponderosa pine, mixed conifer, and Engelmann spruce-subalpine fir), 2) aspen, 3) pinyon-juniper, 4) big and black sagebrush, and 5) desert shrubs (creosotebush, blackbrush, and interior chaparral).

Historical conditions are described in terms of characteristic species composition, structure, size class, age distribution, and fuel complexes that existed in presettlement times. In addition, disturbance frequency, size, and severity of historical disturbance regimes are discussed. Authors also address the ways disturbance historically affected community characteristics and position on landscape.

Current conditions of each focus ecosystem are then compared to historical. Current fuel condition descriptions may include additional information on amount of downed wood, amount of live and dead shrub and herbaceous fuels, annual production, range of annual variation caused by weather, and production relative to dominant life form. Where appropriate, authors also address changes in hydrologic function, such as streamflow, water yield, sediment production, and the potential effects of fire. Current disturbances are also described, including fire and fire suppression, insects, disease, ungulates (domestic and wildlife), exotic plants, patterns and frequencies of disturbances, as well as the resilience of each ecosystem to disturbance.

Each chapter includes a Fire Regime Condition Class (FRCC) (Schmidt and others 2002) assessment of the ecosystem with descriptions of each condition class's characteristics within that particular ecosystem and an estimate of the area currently categorized as FRCC 1, 2, or 3. Authors address the key components that are at risk of being lost if a wildfire were to occur under current conditions. Moreover, authors recommend treatments for each ecosystem by condition class and describe treatments in terms of timing and season, the scale of the treatment, and pre- and post-treatment management considerations.

Study Area

The Southern Utah Demonstration Area roughly encompasses the southern 15 percent of Utah and consists of contiguous state and federal lands within the administrative boundaries of the Bureau of Land Management (BLM), Fishlake and Dixie National Forests, National Park Service, and state of Utah (fig. 1). Dominant vegetation types found in the demonstration area include various associations of pinyon-juniper, ponderosa pine, sagebrush-grass, aspen, spruce-fir, interior chaparral, and desert shrubs (fig. 2). These vegetation types are similar in species composition, stand structure, and ecologic function to vegetation types found on hundreds of millions of hectares in western United States.

Fire Regime Conditions Classes

A fire regime condition class (FRCC) is a classification of the amount of departure from the historical natural fire regime (Hann and Bunnell 2001). Coarse-scale FRCC classes were first defined and mapped by Hardy and others (2001) and Schmidt and others (2002). They are a metric for reporting the number of hectares in need of hazardous fuel reduction and evaluating the efficacy of wildland fuel treatment projects (Rollins and others 2006). This departure results in changes to one or more of the following ecological components: vegetation characteristics (species composition, structural stages, stand age, canopy closure, and mosaic pattern); fuel composition; fire frequency, severity, and pattern; and other associated disturbances (for example, insect and diseased mortality, grazing, and drought) (Hann and others 2004).

FRCC stratifies three condition classes by five natural historical fire regimes. A natural fire regime is a general classification of the role fire would play across a

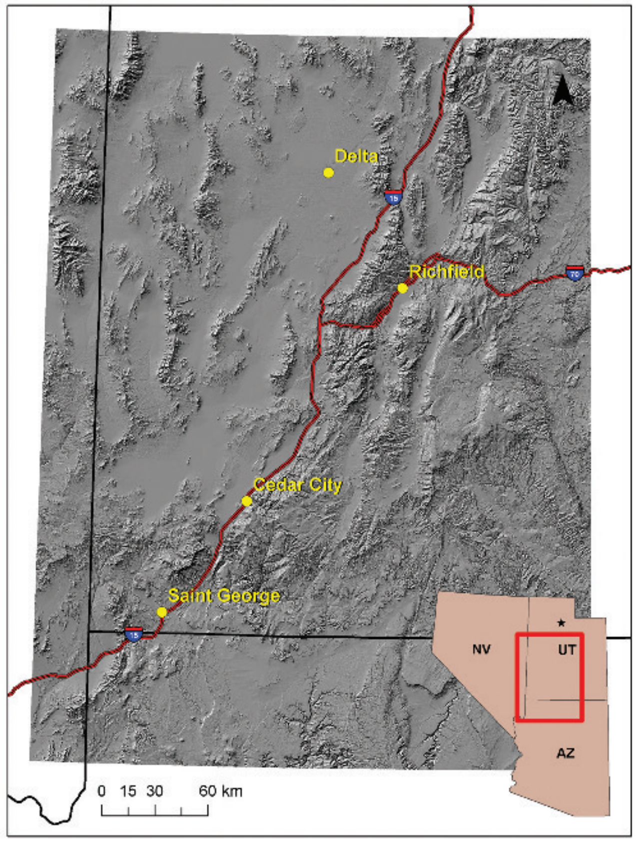

Figure 1—Southern Utah Fuel Management Demonstration Area location.

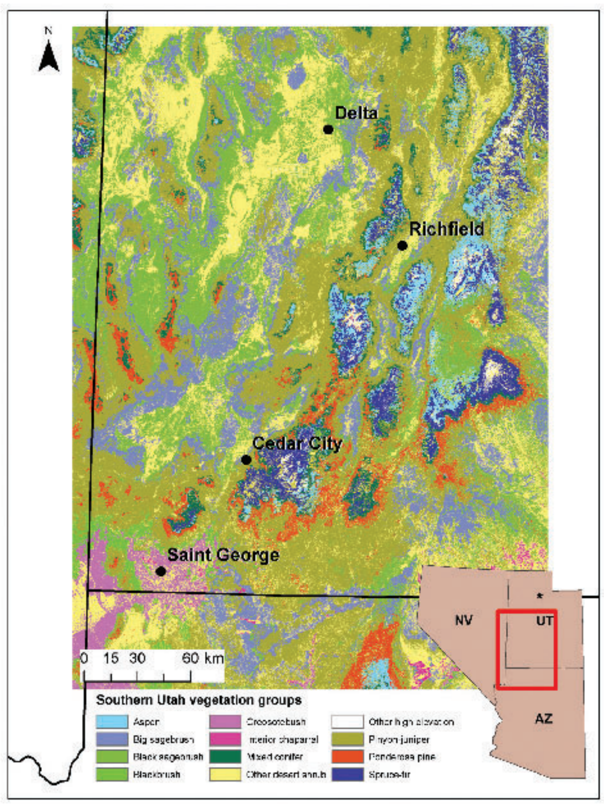

Figure 2—Major vegetation groups in southern Utah project area.

landscape in the absence of modern human mechanical intervention, but includes the influence of aboriginal burning. The five natural historical fire regimes are classified based on average number of years between fires (fire frequency) combined with the severity (amount of replacement) of the fire on the dominant overstory vegetation (table 1) (Hann and others 2004).

The three conditions classes are based on low (FRCC 1), moderate (FRCC 2), and high (FRCC 3) departure from the central tendency of the natural historical fire regime (table 2) (Hann and Bunnell 2001; Hardy and others 2001; Schmidt and others 2002). Low departure (FRCC 1) describes fire regimes and successional status operating within the historical range of variability. FRCC 2 and FRCC 3 characterize conditions outside the historical range (Rollins and others 2006). Characteristic vegetation and fuel conditions are those that occurred within the natural fire regime. Uncharacteristic conditions are those that did not occur within the natural fire regime, such as invasive species, "high graded" forest composition and structure, or repeated annual grazing that maintains grassy fuels across relatively large areas at levels that will not carry a surface fire. Determination of amount of departure is based on comparing a composite measure of fire regime attributes (vegetation characteristics; fuel composition; fire frequency, severity, and pattern) to the central tendency of the natural (historical) fire regime. The amount of departure is then classified to determine the fire regime condition class (table 3) (Hann and others 2004). Additional FRCC information can be found at http://www.frcc.gov/ (Hann and others 2003).

Management Implications

This literature synthesis provides land managers and planners in southern Utah and surrounding areas with the current state of knowledge of the dominant ecosystems in

Table 1—Natural (historical) fire regime classes from Hann and Bunnell (2001) for modeling landscape dynamics in the lower 48 States. Historical Range of Variability (HRV)–the variability of regional or landscape composition, structure, and disturbances during a period of time of several cycles of the common disturbance intervals and similar environmental gradients prior to extensive agricultural or industrial development.

Fire regime class	Frequency (Fire return interval, in years)	Severity	Modeling assumptions
I	Frequent (0 to 35)	Low	Open forest or savannah structures maintained by frequent fire; also includes frequent mixed severity fires that create a mosaic of different age post-fire open forest, early to mid-seral forest structural stages, and shrub or herb dominated patches (generally < 40 ha [100 acres]).
II	Frequent (0 to 35)	Stand replacement	Shrub or grasslands maintained or cycled by frequent fire; fires kill non-sprouting shrubs such as sagebrush, which typically regenerate and become dominant within 10 to 15 years; fires remove tops of sprouting shrubs such as mesquite and chaparral, which typically resprout and dominate within 5 years; fires typically kill most tree regeneration such as juniper, pinyon pine, ponderosa pine, Douglas-fir, or lodgepole pine.
III	Less frequent (35 to 100)	Mixed	Mosaic of different age post-fire open forest, early to mid-seral forest structural stages, and shrub or herb dominated patches (generally < 40 ha [100 acres]) maintained or cycled by infrequent fire.
IV	Less frequent (35 to 100)	Stand replacement	Large patches (generally > 40 ha [100 acres]) of similar age post-fire shrub or herb dominated structures, or early to mid-seral forest cycled by infrequent fire.
V	Infrequent (> 100)	Stand replacement	Large patches (generally > 40 ha [100 acres]) of similar age post-fire shrub or herb dominated structures, or early to mid to late seral forest cycled by infrequent fire.

Table 2—Condition classes from Hann and Bunnell (2001) for modeling landscape dynamics and departure from historical natural range of variability for the lower 48 States. Historical Range of Variability (HRV)–the variability of regional or landscape composition, structure, and disturbances, during a period of time of several cycles of the common disturbance intervals and similar environmental gradients prior to extensive agricultural or industrial development.

Condition class	Departure from HRV	Description
Class 1	None, minimal, low	Vegetation composition, structure, and fuels are similar to those of the historic regime and do not pre-dispose the system to risk of loss of key ecosystem components. Wildland fires are characteristic of the historical fire regime behavior, severity, and patterns. Disturbance agents, native species habitats, and hydrologic functions are within the historical range of variability. Smoke production potential is low in volume.
Class 2	Moderate	Vegetation composition, structure, and fuels have moderate departure from the historic regime and predispose the system to risk of loss of key ecosystem components. Wildland fires are moderately uncharacteristic compared to the historical fire regime behaviors, severity, and patterns. Disturbance agents, native species habitats, and hydrologic functions are outside the historical range of variability. Smoke production potential has increased moderately in volume and duration.
Class 3	High	Vegetation composition, structure, and fuels have high departure from the historic regime and predispose the system to high risk of loss of key ecosystem components. Wildland fires are highly uncharacteristic compared to the historical fire regime behaviors, severity, and patterns. Disturbance agents, native species habitats, and hydrologic functions are substantially outside the historical range of variability. Smoke production potential has increased with risks of high volume production of long duration.

Table 3—A simplified description of the fire regime condition classes and associated potential risks (Hann and others 2003).

Fire regime condition class	Description	Potential risks
Condition Class 1	Within the natural (historical) range of variability of vegetation characteristics; fuel composition; fire frequency, severity and pattern; and other associated disturbances.	Fire behavior, effects, and other associated disturbances are similar to those that occurred prior to fire exclusion (suppression) and other types of management that do not mimic the natural fire regime and associated vegetation and fuel characteristics. Composition and structure of vegetation and fuels are similar to the natural (historical) regime. Risk of loss of key ecosystem components (for example, native species, large trees, and soil) is low.
Condition Class 2	Moderate departure from the natural (historical) regime of vegetation characteristics; fuel composition; fire frequency, severity and pattern; and other associated disturbances.	Fire behavior, effects, and other associated disturbances are moderately departed (more or less severe). Composition and structure of vegetation and fuel are moderately altered. Uncharacteristic conditions range from low to moderate. Risk of loss of key ecosystem components is moderate.
Condition Class 3	High departure from the natural (historical) regime of vegetation characteristics; fuel composition; fire frequency, severity and pattern; and other associated disturbances.	Fire behavior, effects, and other associated disturbances are highly departed (more or less severe). Composition and structure of vegetation and fuel are highly altered. Uncharacteristic conditions range from moderate to high. Risk of loss of key ecosystem components is high.

the area. The review of historical and current conditions highlights how post-European settlement has changed the southern Utah landscape and the problems facing land managers today. The FRCC sections are intended as a general overview of ecosystems conditions. They should not be used to determine actual fire regime conditions in a given area. The Interagency Fire Regime Condition Class Guidebook and other tools are designed for assigning specific fire regime condition classes (Hann and others 2004, www.frcc.gov).

The recommended treatment sections include the commonly used methods to treat areas for fuel accumulation, exotic weed control, and other objectives. New methods, such as mastication, are always emerging to provide managers with more treatment alternatives. Research of these treatments will be necessary to improve our understanding of their effects on the ecosystems. With an understanding of the ecosystems and their responses to treatments and accurate spatial data on fuels, vegetation, fire regimes, and values, we can develop collaborative strategies for managing fuels in southern Utah on a landscape basis.

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Chapter 2

Ponderosa Pine, Mixed Conifer, and Spruce-fir Forests

Michael A. Battaglia and Wayne D. Shepperd

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 20^{th} 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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Chapter 3

Aspen

Dale L. Bartos

Introduction

Quaking aspen (*Populus tremuloides* Michx.) is the most widely distributed broadleaf tree in North American (Little 1971; Sargent 1890). Aspen forests occur from Labrador on the east coast to Alaska in the north to Mexico in the south. In its eastern range, aspen is relatively continuously distributed. In the western United States, however, it occurs on the more suitable sites on mountains and high plateaus (Jones 1985). On xeric sites, it is found primarily in riparian zones. Most western aspen occur on public lands and exist as pure clones, successional to conifer, or as small-scattered decadent groves. At least 75 percent of aspen in the western United States occurs in two states, Utah and Colorado. Almost 25 percent of Utah's forests are aspen (Mueggler 1988).

Aspen is portrayed as an excellent indicator of ecological integrity as well as landscape health (Kay 1991a,b; Woodley and Theberge 1992; Woodley 1993; Woodley and others 1993). Aspen ecosystems in the western United States yield numerous products and benefits, including forage for livestock, habitat for wildlife, water for downstream users, esthetics, sites for recreational opportunities, wood fiber, and landscape diversity (Hessl 2002). Loss and potential loss of aspen from these western landscapes is attributed primarily to the reduction of wildfire, long-term overuse by ungulates (both wildlife and domestic livestock), and successional processes. During 2006, considerable attention was given to aspen die-off. This die-off is described as the rapid death of entire clones, including the root system, with no potential for aspen survival (http://www.fs.fed.us/rmrs/research/highlights/aspen-restoration.php).

Aspen is an early successional, shade intolerant species with a short life span—usually less than 200 years in the western United States (Perala 1991). Aspen exist primarily as one of three types: 1) stable, 2) successional to conifers or seral, and 3) decadent and falling apart (Bartos 2001). Unlike other western forest trees, aspen are unique because they reproduce almost exclusively by suckering from the parent root system. Generally, a disturbance or die-back is necessary to stimulate regeneration of the aspen stands. It is speculated that these self-regenerating stands have existed for thousands of years. Unlike other tree species, if aspen stands are lost from the landscape they usually will not return through the natural process of seed dispersal.

Most aspen stands will eventually be replaced by conifers, sagebrush (*Artemisia* spp.), or possibly other shrub communities if the current conditions that have prevailed for the past 100 to 150 years (for example, lack of fire, excessive wildlife, and livestock browsing) continue. Current unpublished Forest Inventory and Analysis (FIA) data for the state of Utah show that there has been approximately a 60 percent decrease in aspen-dominated public lands since European settlement (Bartos and Campbell 1998a). This decrease is generally uniform across the entire state. When aspen lands convert to conifer dominated landscapes, substantially less water is available for streamflow, undergrowth biomass production is greatly reduced, and there is a marked decline in the diversity of plants and animals.

Development of useful management recommendations concerning the aspen, aspen/conifer, and aspen/sagebrush ecosystems requires a good understanding of the structure and function of these systems. Aspen literature for the western United States has been summarized through the early 1980s (DeByle and Winokur 1985). The published proceedings of a more recent aspen symposium held in June 2000 was a supplement to this earlier work (Shepperd and others 2001). Shepperd and others (2006) recently synthesized the existing information on the ecology and management of aspen in the Sierra Nevada of California.

The purpose of this paper is to discuss the historical and current conditions of aspen, with an emphasis on aspen in Utah. This paper also identifies clones at risk, provides a method of treatment prioritization, and offers management recommendations of aspen stands. Landscapes that need priority management are mixed-conifer/aspen, particularly where subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) dominates, and aspen/sagebrush transition zones. Recommendations to regenerate aspen include the need for immediate, large-scale (200 to 400+ ha [500 to 1000+ acres]) treatments that mimic natural mosaic patterns.

An aspen stand typically contains numerous genetically identical stems (ramets) that propagated vegetatively from seedlings at some time in the past when conditions were favorable. Aspen stems originate from root suckers, some of which still may be interconnected via a shared root system. Shepperd and Smith (1993) reported that aspen stems establish independent root systems by approximately 25 years of age, with few stems still connected to the original root system. Barnes (1975) speculated that these self-regenerating clones have existed for thousands of years, being perpetuated over time by disturbance – primarily fire. Some clones occupy large areas. One such clone, referred to as Pando (Latin for *I spread*), occurs on the Fishlake National Forest in central Utah (Mitton and Grant 1996; Grant 1993). The authors claim that this 43 ha (106 acres) clone is the world's largest living organism. Kemperman and Barnes (1976) originally identified the Pando clone and speculated that clones as large as 81 ha (200 acres) may exist.

Successful sexual reproduction of western aspen is extremely rare (Mitton and Grant 1996). Jelinski and Cheliak (1992) describe a "window of opportunity" that may allow seedling establishment at intervals of 200 to 400 years. Abnormally cool, moist years may allow sexual reproduction of aspen seedlings (Elliott and Baker 2004). There are some documented cases of successful aspen seedling establishment in the west. Romme and others (2005) documented the establishment and persistence of sexually produced aspen seedlings after the 1988 fires in Yellowstone National Park. Elliott and Baker (2004) hypothesized that expanding aspen clones at tree line in the San Juan Mountains of Colorado were established by seed. However, unlike other western tree species, aspen seedling establishment is rare and undependable (Barnes 1966). Therefore, once aspen is lost from the landscape, it will seldom establish from seed in the Intermountain West. Vegetative reproduction by suckers generally requires a disturbance (fig. 1) or die-back that alters the hormonal balance of a clone (Bancroft 1989; Schier and others 1985). Frey and others (2003) recently reviewed the process of aspen sucker regeneration. Bancroft (1989) and Bartos (2001) discuss what happens when there is a disruption in the flow of two hormones (cytokinin and auxin) within an aspen tree. Basically, when the tree is killed or stressed, the flow of sucker-suppressing auxin from the crown down to the root system is disrupted, which allows a second hormone, cytokinin, to stimulate suckering.

Figure 1—One-year old aspen (*Populus tremuloides*) sprouts after the Sanford Fire, UT.

To abide by the words of Aldo Leopold: "…if we are serious about restoring or maintaining ecosystem health and ecological integrity, then we must first know what the land was like to begin with" (quote in Covington and Moore 1994). For millennia, wildfire dominated the natural functioning of the aspen ecosystem (Baker 1925; Parfit 1996; Pyne 1995). Baker (1918) and Schier (1976) describe western aspen as a fire dependent species. Jones (1985) stated "…fire is responsible for the abundance of aspen in the west and for the even-aged structure of so many stands." Conversely, reduction of fire in the Interior West is probably one of the main causes of the decline of aspen from so much of our landscapes because it permits the encroachment of shade tolerant conifers that reproduce from seed (Bartos 2001; Schier 1975).

Historical Conditions

Fire has been a major component of the vegetation landscape prior to humans arriving on the scene, and later became a critical tool for man. Pyne (1995) suggests that fire and humans co-evolved. The native inhabitants of the North American continent have long been considered natural conservationists living in harmony with the environment. However, it is now becoming clear that the American Indians had a profound influence on the landscape, particularly with respect to the use of fire. Ongoing fire history studies in southern Utah indicate that during the approximately 400 years prior to settlement, fire-free intervals varied from 20 to 60 years. These fire free intervals were longer with increases in elevation (Chappell 1997). When European man arrived in the western United States around 1850, they found a mosaic of vegetation profoundly influenced by Native Americans over a period of 12,000+ years (Kay 1997). These "pristine" landscapes were in a continual state of flux as a result of both natural and human caused burning.

Early settlers further impacted the land through removal of fine fuels by grazing domestic livestock, thus limiting the spread of fire. In the early part of the 20th century, land managers instigated a vigorous campaign of fire control. These two practices (grazing and fire control) have had a profound effect on the vegetation we now see on western landscapes. As a result of this lack of fire, aspen stands on the landscape have been greatly reduced. Because western aspen reproduces primarily by suckering, the elimination of fire combined with excessive browsing of the asexual reproduction by ungulates has caused many aspen dominated sites to convert to other vegetation types. Barnes (1975) speculates that many of the existing clones have been present since the last ice age, while Knight (1994) states some clones may have persisted since Pleistocene times. Therefore, clones that have persisted for thousands, if not hundreds of thousands, of years are in danger of being eliminated in as little as a couple of centuries.

Brown and Simmerman (1986) detailed the difficulty of burning pure aspen, while DeByle and others (1987) describe the aspen type as "asbestos" in nature and discuss its use as natural firebreaks. Earlier, Fechner and Barrows (1976) discussed the use of pure aspen as firebreaks. They reported that fire ignition rates in quaking aspen stands are less than half those for all other cover types in Colorado. Crown fires often drop to the ground when they reach pure aspen stands and therefore, fire spreads only a short distance into the stands. As early as 1925, Baker (1925) cited grazing as the primary reason many aspen stands no longer burn. He stated that intense grazing pressure reduces the undergrowth in aspen clones such that fire spread is inhibited. Fires before grazing became heavy in the mountains of Utah were as frequent as 7 to 10 years, but stopped about the time grazing began (Baker 1925). DeByle and others (1987) gave four possible reasons why fires are less frequent in aspen today than historically: 1) effective direct control of wildfires, 2) grazing, 3) removal of American Indians from historic range, and 4) succession. This phenomenon of aspen no longer burning needs to be further evaluated, especially now that there is so much concern with respect to the wildland/urban interface.

Current Conditions

Aspen is a disturbance-dependent species that flourished when these western lands burned periodically. With suppression of this natural force, many of these lands have converted, or are in the process of converting, to other vegetation types, such as conifer (for example, subalpine fir) or sagebrush. Three of the most critical products being lost from the aspen system as a result of this conversion are water, undergrowth vegetation, and biodiversity.

DeByle and Winokur (1985) reported that there were more than 2.8 million ha (7 million acres) of aspen in the Interior West and, Mueggler and Campbell (1986) reported over 648,000 ha (1.6 million acres) of aspen-dominated forests in Utah. The Rocky Mountain Research Station's FIA Project has compiled data representing current and historical acreage of aspen in the Interior West. The historical data result from summing all acres that currently contain at least one aspen, either living or dead. This assumes that this acreage is, or once was, occupied by aspen. For the state of Utah, FIA data show that there has been almost a 60 percent decrease in aspen dominated lands since the arrival of European man (table 1). This loss is fairly uniform across the entire state as seen from the data for individual forests. Similar trends are seen elsewhere throughout the western United States. Analysis of FIA plots in Idaho, Colorado, and Wyoming show regional aspen cover loss (Rogers 2002). Rogers and others (1998) report that 31 percent of the FIA plots in Colorado aspen forest types are "unstable" and are possibly in transition to other types. However, Manier and Laven (2002) and Kaye and others (2003) found no evidence of aspen decline in some areas of Colorado. Lachowski and others (1996) and Wirth and others (1996) used remote sensing and geographic information systems (GIS) to evaluate the loss of aspen in the Gravelly Mountains in southwestern Montana. They found that aspen decreased by approximately 47 percent over a 45-year period (1947 to 1992) and attributed this decrease to succession to conifer species. Brown (1995), in a review article, states that the deterioration of aspen in Oregon and Washington appears similar to the decline of aspen in other parts of the west. In the South Warner mountains in northern California, Di Orio and others (2005) found a 24 percent decline in aspen cover between 1946 and 1994 and a decrease in the mean size of aspen stands.

Ungulates

Ungulates have a major impact on aspen regeneration (Bartos and Mueggler 1981; Bartos and others 1994; Kay 1985, 1990, 1995; Kay and others 1994; McCain and others 2003; U.S. Department of Agriculture 1994). Excessive browsing on aspen suckers by wildlife (elk [*Cervis canadensis*], deer [*Odocoileus hemionus*], or moose [*Alces alces*]) or by livestock (cattle or sheep) can suppress or eliminate regeneration in aspen stands. Excessive livestock use of young aspen suckers can threaten stand maintenance in climax aspen communities, especially if the use is in August or September (McCartney 1993). Early in the past century, Sampson (1919) noted that cattle were not a detriment to the establishment of aspen suckers "unless the range is stocked with cattle beyond its natural carrying capacity." He further noted that sheep would eliminate all aspen reproduction where sites were grazed 3 consecutive years. Smith and others (1972) concluded that proper livestock management is essential to regeneration of aspen following removal of older stands. A recent model-based assessment of aspen and elk herbivory in Rocky Mountain National Park suggests that a reduction of elk populations up to 80 percent is necessary to allow regeneration of 90 percent of the park's declining aspen stands (Weisberg and Coughenour 2003).

The Pando clone (Kemperman and Barnes 1976) mentioned earlier is a good example of the problems associated with the decline of aspen. Managers observed that this single-story clone was not regenerating successfully. A cutting treatment was imposed on a portion of the

Table 1—Current and historical acres of aspen found in Utah (Unpublished data provided by the USDA Forest Service, Rocky Mountain Research Station's Forest Inventory and Analysis Project.).

	Current aspen		Historical aspen		Decline
National Forest	**hectares**	**(acres)**	**hectares**	**(acres)**	**(percent)**
Ashley	41,036	(101,358)	130580	(322,532)	69
Uinta	70,645	(174,492)	115527	(285,351)	39
Wasatch-Cache	52,071	(128,615)	151351	(373,837)	66
Dixie	61,965	(153,053)	177213	(437,715)	65
Fishlake	57,469	(141,948)	127014	(313,724)	55
Manti-LaSal	64,318	(158,866)	136845	(338,008)	53
Southern Utah	183,752	(453,867)	441072	(1,089,447)	58
Northern Utah	163,751	(404,465)	397457	(981,720)	59
Total	347,503	(858,332)	838529	(2,071,167)	59

clone in the early 1990s. The initial treated area produced suckers; however, none of them survived. Subsequently, an additional portion was cut and fenced with a 2-m (7-ft) high woven wire fence to exclude large animals. This resulted in successful regeneration in the cut and fenced area that should be stand-replacing. Apparently, deer and cattle browsing effectively suppressed sucker reproduction in the unprotected portion of the stand. The Coconino National Forest in Arizona installed two large elk exclosures after a wildfire to facilitate aspen regeneration. Elk browsed 85 percent of aspen shoots in high severity burned areas outside the exclosures and 34 percent of the shoots on the intermediate burned areas (Bailey and Whitman 2002). This suggests elk pressure is greatest in areas where recent high severity burns produce large numbers of aspen shoots.

This potentially severe ungulate influence on successful aspen regeneration adds a critical component to management considerations in certain parts of the west (Shepperd and Fairweather 1994). Burned clones subjected to repeated browsing only hastens their demise (Bartos and others 1994; Kay 1990). Therefore, treatments to induce suckering must not be initiated before relief from excessive browsing is obtained (U.S. Department of Agriculture 1994). Kay (1997) speculates that the historic abundance of aspen in the Intermountain area suggests that these ecosystems developed with relatively low levels of ungulate pressure. Larsen and Ripple (2003, 2005) found that aspen recruitment into the overstory was significantly higher on the National Forests bordering the northern portion of Yellowstone National Park than inside the park where elk pressure is greater. They also found that overstory aspen recruitment inside the park was only occurring in areas protected from ungulate browsing, such as fenced exclosures and scree slopes. This further supports the need to control or modify browsing intensity, where necessary, if these western aspen communities are to survive by vegetative reproduction.

Insects and Disease

Quaking aspen plays host to a wide array of both insects and diseases. Hinds (1985) states that many diseases attack aspen, however, very few kill or seriously injure living trees. For example, leaf spot (*Marssonina populi*) is a common leaf disease of aspen in the west. If this disease prevails for 2 or more years, it can cause twig and branch mortality, but rarely kills mature trees (Harniss and Nelson 1984). Many of the prevalent diseases in aspen affect both regeneration and mature trees (Hinds 1985). In both cases, trees damaged and weakened by animals are easy targets for introduction of disease. Disease plays a part in thinning excessive regeneration of aspen in areas where stand replacing episodes have occurred. Hart and Hart (2001) working in western Wyoming found that larger aspen stems (>15 cm [5.9 inches]) had less mortality due to disease/animal wounds than smaller stems. This observation substantiates the fact that disease tends to reduce aspen regeneration and smaller trees. Aspen die-off, mentioned earlier, is occurring from northern Arizona to Canada and currently, is most prevalent in southern Utah and western Colorado. Some speculate that this die-off phenomenon is a result of prolonged drought conditions coupled with either opportunistic diseases and/or insects. More detailed studies are needed to define this problem in order to make management recommendations that address the issue (Aspen Summit 2006).

More than 30 insect species are listed by Furniss and Carolin (1977) as utilizing aspen. Some of these species cause minimal damage, some just attack and impact weakened or dying trees, and some may severely damage or kill otherwise healthy trees. Insect impacts coupled with other factors, such as ungulate use, can have major impacts on regeneration. Jones (1985) gives a thorough discussion of insects and their impact on western aspen.

Fire Exclusion

The 7- to 10-year fire return interval suggested by Baker (1925) indicates these lands were burning often prior to European settlement. Because of the disruption in the fire regime and natural succession, we are finding aspen that are considerably older than the normal pathological rotation age of 70 to 80 years (Personal communication, John Guyon, Pathologist, Forest Health Protection, Ogden, UT). This aging of aspen on the landscape gives rise to the potential for more catastrophic disease and insect events. Additionally, this excessive mortality will add to the fuel loads.

Ecosystem Impacts Caused by Aspen Loss

Loss of aspen from western landscapes translates to a loss of water, forage, and biodiversity, as well as other benefits (Bartos and Campbell 1998b). Harper and others (1981) suggest that water yields can be expected to decrease 5 percent as conifers replace aspen. Gifford and others (1984) concluded from modeling studies that a net water loss of 18.6 cm (7.3 inches) occurs when spruce (*Picea* spp.) replaces aspen, and a loss of 7.2 cm

(2.8 inches) of water when subalpine fir forests replace aspen. Therefore, aspen-to-conifer succession appears to have the potential to markedly reduce water yields in the western United States (Bartos and Campbell 1998b). This loss of water means that it is not available to produce undergrowth vegetation, recharge soil profiles, or increase streamflow. In dry climates, such as the Great Basin, this loss of water is substantial and should be of great concern to the public.

Aspen forests traditionally have been considered prime grazing lands in the Intermountain West. Forty to 70 percent of the undergrowth production found in association with major aspen community types in the Intermountain Region consists of palatable forage (fig. 2) (Mueggler 1988). Mueggler found that the most productive aspen communities produced as much as 3,200 kg/ha (2,900 lbs/acre) air-dry undergrowth material and averaged over 1,350 kg/ha (1,200 lbs/acre). Mueggler (1985) found when conifers comprised as little as 15 percent of the total tree basal area on the Wasatch Plateau of Utah, undergrowth production was reduced approximately 50 percent (fig. 3a). Mueggler (1988) reported an even greater reduction on the Fishlake and Dixie National Forests of southern Utah, with undergrowth production reduced two-thirds when conifers comprised as little as 15 percent of the total tree basal area (fig. 3b). Similar reductions in undergrowth forage production occur in other forest types as conifer basal area increases (Jameson 1967). Once conifer invasion approaches 50 percent of the total tree basal area in aspen stands, undergrowth production is likely to be only a small fraction of what it once was on these formerly prime grazing lands.

When we consider that more than half of our aspen dominated stands in the Interior west have converted to conifer during the past 150 years, we obviously can conclude we have experienced a large loss of forage production. If this dramatic decrease in vegetation is not considered when determining stocking rates for grazing allotments, an overgrazing problem is compounded. For example, if a 50 percent decline of aspen stands on an allotment occurred without a corresponding adjustment of allowable stocking, the remaining forage producing areas will be subjected to much heavier utilization than previously. This excessive pressure will adversely impact aspen regeneration as well as the other forage resources.

Figure 2—Aspen (*Populus tremuloides*) with productive herbaceous understory, including cowparsnip (*Heracleum lanatum*), snowberry (*Symphoricarpos oreophilus*), and numerous other forbs and grasses. In addition, subalpine fire (*Abies lasiocarpa*) and other conifers are establishing in the understory.

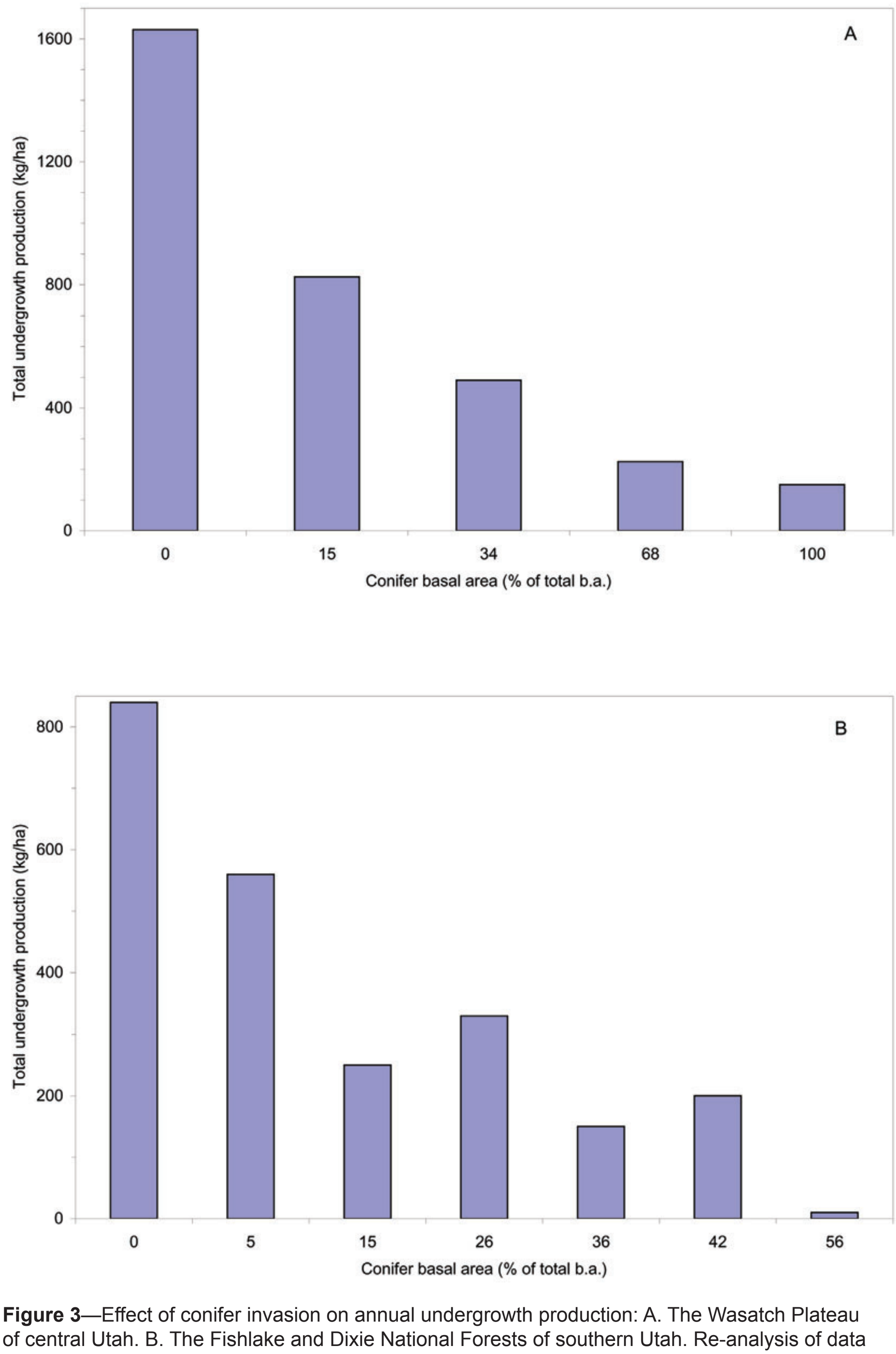

Figure 3—Effect of conifer invasion on annual undergrowth production: A. The Wasatch Plateau of central Utah. B. The Fishlake and Dixie National Forests of southern Utah. Re-analysis of data reported in Mueggler (1985).

Aspen communities have exceedingly high biodiversity–second only to riparian areas on western ranges (Kay 1997). When aspen lands become dominated by either conifer or sagebrush, marked changes in both flora and fauna occur and biodiversity is compromised. Not only is there a loss of forage production, but there is also a substantial decrease in plant species richness associated with the loss of aspen.

Winternitz (1980) reported that the density and diversity of birds was greater in aspen than conifer stands, and McGraw/Bergstrom (1986) found that mature stands of aspen had the most bird species compared to younger stands or those being invaded by conifers. Bird species diversity also increases with the size of aspen stands (Johns 1993). Linder (1995) found that bird community structure is positively correlated with the size of aspen patches.

Other aspects of biodiversity loss are not so readily apparent. For example lichens, bats, and snails are also affected by the decline of aspen. McCune (1997) indicates that a loss of aspen would adversely affect the epiphytic lichen communities that depend on mature to old aspen. Vonhof (1996) found in Southern British Columbia that both big brown bats (*Eptesicus fuscus*) and silver-haired bats (*Lasionycteris noctivagans*) prefer trembling aspen for roost sites. In the Cypress Hills, Saskatchewan, all little brown bats (*Myotis lucifugus)* roost in aspen trees (Kalcounis and Hecker 1996). Crampton and Barclay (1996) monitored two species of bats (little brown bats and silver-haired bats) in aspen stands in Lac La Biche, Alberta, and found that the bats preferred newly dead aspen with heart rot and trees with low leaf cover. Beetle (1997) determined that 21 species of land snails occurred in aspen stands in Yellowstone National Park and that these stands had a greater abundance and variety of snails than coniferous forests on more acidic soils.

Fire Regime Condition Classes

Fire Regime Condition Class 1 (FRCC 1)

Fire Regime Condition Class 1 (FRCC 1) indicates that the fire regimes are within a historical range and there is little risk of losing aspen from the ecosystem. This class includes the stable and seral stands that are reproducing and have little conifer encroachment. Mueggler (1989) speculated that as much as a third of the aspen in the west could be classified as climax or stable. Smith and Smith (2005) reported 16 percent of the aspen stands on the Uncompahgre Plateau, CO, were stable. Therefore, assuming a 60 percent decline in the historical range of aspen, only a portion of aspen dominated land (10 to 15 percent) would be considered functioning normally, in other words, regenerating naturally without major interventions. Stable clones are not subject to succession, primarily because there is a lack of conifer seed source and most natural regeneration is not consumed by ungulates. This leaves only a few aspen in Utah that are able to survive and perpetuate without naturally occurring wildfire or other disturbance. These are approximate figures for Utah and may be more or less depending on various situations. Caution should be used when extrapolating to aspen outside of Utah.

FRCC 1 aspen stands contain no conifers, are multistoried with numerous age classes, and usually have a fairy ring or skirt of regeneration around the outside of the clones, which makes them hard to see through or into (fig. 4) (Bartos 2001). These stands may also have present a large shrub component of species such as chokecherry (*Prunus virginiana* L.) or snowberry (*Symphoricarpos albus* (L.) Blake). Aspen stands in FRCC 1 regenerate without many external forces, such as disease or abiotic factors, acting upon them. The fire return interval in these stands is characterized by either frequent, very low intensity fires that cause some overstory mortality and subsequent regeneration, or more infrequent higher intensity fires every 100 to 200 years that result in new even-aged stands (Brown and Simmerman 1986). These high intensity fires in pure stands of aspen result when there are a lot of fine fuels, and/or a considerable shrub component coupled with extremely dry conditions. The low intensity fire return interval could be as frequent as 7 to 10 years (Baker 1925; DeByle and others 1987). In such cases, FRCC 1 seral stands should be able to regenerate enough sprouts to successfully restock the stand after fire. Baker (1925) states that 6,000 aspen sprouts per hectare (2,500 sprouts/acre) by the end of year 3 are successful post-fire stocking levels. However, ungulate pressure must also be low enough to prevent excessive browsing of the new aspen shoots.

Fire Regime Condition Class 2 (FRCC 2)

Fire Regime Condition Class 2 (FRCC 2) indicates that the fire regimes have been altered and there is a moderate risk of losing key ecosystem components. In Utah, approximately 25 to 30 percent of the aspen are in FRCC 2. Seral FRCC 2 stands have a large conifer component that over time will convert the stand to a mixed conifer type if no disturbance occurs (fig. 5).

Figure 4—Healthy aspen (*Populus tremuloides*) stand with a younger age class at the outer edge.

Figure 5—Aspen (*Populus tremuloides*) stand with a co-dominant overstory of subalpine fir (*Abies lasiocarpa*) and Engelmann spruce (*Picea engelmannii*). Dominant grass and shrub species are slender wheatgrass (*Elymus trachycaulus*) and gooseberry currant (*Ribes montigenum*).

Stable FRCC 2 stands have limited sprouting and some of the mature stems are dead and dying. Sprouting may be limited because of low light levels in the case of conifer encroachment, or if ungulate browsing pressure is heavy, the clone may be sprouting but the sprouts are summarily eaten.

Fire Regime Condition Class 3 (FRCC 3)

Fire Regime Condition Class 3 (FRCC 3) contains all aspen that are not included in the previous two classes. Fire return intervals have been drastically altered in FRCC 3 stands and there is a high risk of losing components of the ecosystem. Approximately 60 percent of the aspen component in Utah is classified as FRCC 3. Because of the lack of burning or other disturbance, these once aspen dominated lands are being replaced primarily by conifers. In FRCC 3 stands, conifer canopy cover exceeds aspen canopy cover and they are now considered mixed-conifer rather than aspen cover types. Without treatment or disturbance, these decadent aspen stands will soon be completely replaced by conifers and lost from the landscape (fig. 6). As these aspen stands age and are replaced by conifers, the aspen begin to die (around 120 years) and fall to the ground and increase fuel loadings. Higher fuel loadings can cause fires to burn more severely, creating significant soil heating. Most aspen suckers regenerate within 6 cm (2.4 inches) of the surface and virtually all regenerate within the upper 12 cm (4.7 inches) of soil (Perala 1991). Therefore, high severity fires could reduce suckering potential by killing the roots, and subsequent stocking levels may not be high enough to successfully regenerate the stand. Aging aspen stands also may not be able to regenerate successfully due to a reduced capacity of the root system to sprout after disturbance. Browsing by both wild and domestic ungulates further complicates the situation. In areas where ungulate pressure is intense, disturbance events are not successful because all suckers are browsed and some aspen clones are ultimately lost. Excessive utilization needs to be addressed before fire or other treatments are returned to the system (U.S. Department of Agriculture 1994). If one goal of management is to restore aspen, then animal use needs to be monitored, evaluated, and adjusted. Otherwise, animals utilizing the aspen regeneration can slow or defeat restoration efforts.

Recommended Treatments

Restoring aspen to a level near its natural range of variability requires that land managers take an aggressive management approach in the very near future. Land managers in the western United States are currently very interested in actively treating their deteriorating aspen stands. Five major risk factors have been identified

Figure 6—Former aspen (*Populus tremuloides*) stand now dominated by ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*), and white fir (*Abies concolor*).

to assist managers in evaluating their aspen resource (Bartos and Campbell 1998a). These risk factors are: (1) conifer cover >25 percent, including reproduction in the understory; (2) sagebrush cover >10 percent; (3) aspen canopy cover <40 percent; (4) dominant aspen trees >100 years of age; (5) aspen regeneration 1.5 to 4.5 m (5 to 15 ft) tall and <1,200 stems per ha (<500 stems/acre).

A prioritized key (table 2) was developed by Campbell and Bartos (2001) that incorporates the above risk factors and is applicable to most aspen stands found in southern Utah. In part, this key was developed with knowledge and expertise of aspen systems that occur in this region. The stands at greatest risk and highest priority are those having conifer canopy cover that exceeds aspen canopy cover. These are currently mixed-conifer rather than aspen cover types. However, with proper treatments, the aspen cover type can usually be restored and sustained. Mueggler (1989) developed a general management decision model for maintaining aspen stands in the west (fig. 7). From work in Ontario, Peterson and Peterson (1992) suggested that stands targeted for treatment need at least 40 parent aspen stems per hectare (16 stems/acre) to produce the minimal acceptable stocking and about 125 parent aspen stems per hectare (50 stems/acre) to fully stock a stand. However, Shepperd (2004) reported successful regeneration of an aspen clone in Arizona with only two remaining parent stems when fencing was used to protect suckers.

Numerous techniques are available for restoring these decadent aspen clones or late successional clones dominated by conifers. These include prescribed burning, clear cutting, cutting and burning, fencing, spraying, ripping, and chaining (Bartos and Harniss 1990; Bartos and Lester 1984). Such treatments should give rise to abundant aspen suckers if the three essential elements to the aspen regeneration triangle are met: 1) hormonal stimulation, 2) proper environment, and 3) protection of new suckers (Shepperd 2001, 2004). Shepperd (2001) lists seven aspen regeneration alternatives: 1) doing nothing, 2) commercial harvest, 3) prescribed fire, 4) mechanical root stimulation, 5) removal of vegetative competition, 6) protection of regeneration from herbivory, and 7) regenerating from seed.

Table 2—Key to the risk factors used to prioritize areas with aspen for restoration and conservation actions in the Intermountain West. Assumption: Aspen are present with a density of at least 50 mature trees per hectare (20 mature trees per acre). Note: Couplet 1 refers to relative cover; couplets 2 to 5 use absolute cover (Campbell and Bartos 2001).

Risk Factor			Priority
1.	a.	Conifer species comprise at least half of the canopy cover.	Highest priority
	b.	Aspen comprises more than half of the total canopy cover.	2
2.	a.	Aspen canopy cover is less than 40 percent; *and* sagebrush usually a dominant understory species exceeds 15 percent cover.	High priority
	b.	Not as above.	3
3.	a.	Conifer cover (including overstory and understory) exceeds 25 percent.	Moderate to high priority
	b.	Conifer cover is less than 25 percent.	4
4.	a.	Aspen regeneration (1.5 to 4.5 m [5 to 15 feet tall]) is less than 1,200 stems/ha (500 stems/acre).	Moderate priority
	b.	Aspen regeneration exceeds 1,200 stems/ha (500 stems/acre).	5
5.	a.	Any two of the following three risk factors are represented: 1 – Aspen canopy cover is less than 40 percent. 2 – Dominant aspen trees are greater than 100 years old. 3 – Sagebrush cover exceeds 10 percent.	Low to moderate priority
	b.	Two of the three risk factors in 5a are not represented.	6
6.	a.	One of the three risk factors in 5a is represented.	Low priority
	b.	None of the risk factors above are represented.	Candidate for properly functioning condition

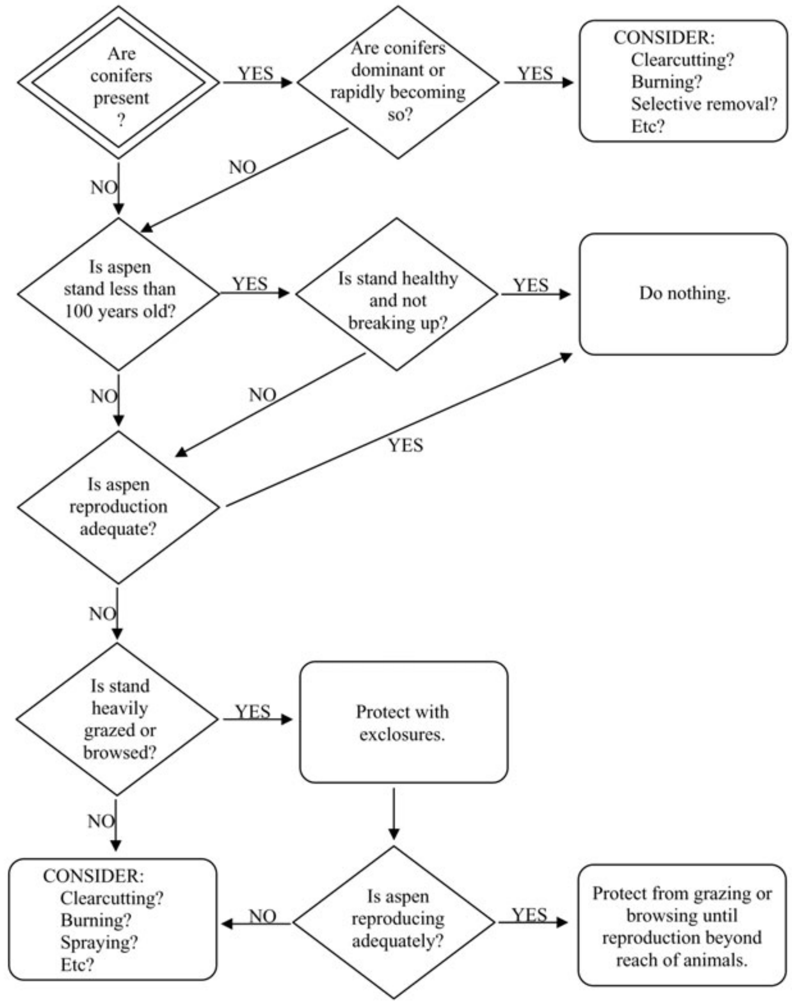

Figure 7—General management decision model for maintaining aspen (*Populus tremuloides*) stands in the western United States. Adapted from Mueggler (1989).

Doing Nothing

This action is suitable for stands in FRCC 1. Management intervention is likely unnecessary if the stand is "showing little sign of decline, disease, or distress from competition, contains multiple age classes, or is successfully suckering" (Shepperd 2001).

Commercial Harvest

Where aspen markets exist, clearfelling is the preferred harvest method to regenerate a declining stand. Partial harvests result in fewer suckers and expose the residual mature stems to breakage, windthrow, and sunscald (Shepperd 2001). Dormant season logging promotes more suckering than spring or summer harvests (Perala 1991; Frey and others 2003). Ohms (2003) found clearfelling aspen clones produced significantly more suckers and increased sucker height compared to the control clones. Treatment response was even more successful in the areas of the clearcuts where browsing was excluded. The successful regeneration of the aspen stand after clearfelling is dependent upon the size of the area treated, unless browsing is controlled (Mueggler and Bartos 1977). If ungulate use is not controlled, treated areas should be greater than 5 ha (12 acres).

Fire

In climax aspen stands, suckering following the death of decadent stems is usually sufficient to maintain the stands. However, in many areas where aspen is successional to conifers, fire can promote sufficient suckering to maintain aspen stands and remove encroaching conifers. Working in central Utah, Baker (1925) stated that a 50-year fire return interval would keep most aspen stands free of conifers. A comprehensive fire guide for treating aspen in the western United States was developed by Brown and Simmerman (1986). In this guide, they evaluated fuels and flammability for five specific types: aspen/shrub, aspen/tall forb, aspen/low forb, mixed/shrub, and mixed/forb. Various factors were evaluated and a probability for a successful burn was given. This information should be useful to the resource manager for developing prescribed fire burn plans.

Bradley and others (1992) prepared a detailed report that discusses fire ecology of forests and woodlands in Utah. This paper identifies various fire groups, some of which include an aspen component. Habitat types developed by Mauk and Henderson (1984), Youngblood and Mauk (1985), and some of the community types discussed by Padgett and others (1989) and Mueggler (1989) are assigned to 13 "Fire Groups" (Bradley and others 1992). Five of these fire groups contain aspen and would be relevant in planning any prescribed burn. These five specific fire groups are: (a) 7-community types where aspen appears to be climax or a long-term seral dominant; (b) 3-Ponderosa Pine Habitat Type (*Pinus ponderosa* P. & C. Lawson); (c) 5-Douglas-Fir Habitat Type (*Pseudotsuga menziesii* (Mirbel) Franco); (d) 6-White Fir and Blue Spruce Habitat Types (*Abies concolor* (Gord. & Glend.) Lindl. Ex Hildebr. and *Picea pungens* Engelm.); and (e) 10-Subalpine Fir Habitat Types. Items b through e contain aspen that are successional to various coniferous species indicated in the habitat type.

Fire that kills most of the aspen overstory produces more shoots than less intense fires that result in lower overstory mortality. Keyser and others (2005) reported high severity fires (>75 percent aspen mortality) in aspen stands produced more shoots, killed encroaching conifers, and had greater shoot height growth than low severity fires (25 to 75 percent aspen mortality) or unburned aspen stands. Aspen sprout aboveground biomass was 10 times greater on high severity sites (100 percent aspen mortality) than intermediate severity burned sites (~50 percent aspen mortality) after a wildfire in Arizona (Bailey and Whitman 2002).

Mechanical Root Stimulation

Severing aspen roots from parent stems can stimulate suckering. Shepperd (1996) found that tipping over mature aspen stems resulted in significantly more sprouts than stems that had been chainsaw felled. Ripping around the perimeter of a decadent aspen clone can also stimulate suckering. Variations of ripping around aspen clones have proved successful on the Coconino National Forest in Arizona (Shepperd 2004). This method can expand the size of the clone and introduce new age classes without cutting existing mature stems.

Removal of Vegetative Competition

Removing competing vegetation increases light to the understory (Shepperd 2001). Vegetation removal can be done alone or in conjunction with other treatments to increase aspen regeneration. In northern California, conifers up to 66 cm (26 inches) diameter at breast height (DBH) were removed around existing aspen stems to encourage suckering (Jones and others 2005). Aspen stem density was significantly higher in the treated stands 4 years post-treatment when compared to controls. In a separate study to re-invigorate the clone, all conifers were removed and an elk-proof fence was installed around a declining aspen clone consisting of only two remaining live aspen stems (Shepperd 2004). Four years post-treatment, the stand had produced numerous sprouts and the treatment was considered successful.

Protection of Regeneration from Herbivory

In some areas, confounding factors exist that will challenge the manager to be successful. Treatment to induce aspen suckering is not enough in areas where there is extensive ungulate pressure (both domestic and wildlife) because uncontrolled ungulate use could cause some aspen treatments to be unsuccessful. Such actions to induce suckering must not be initiated before relief from excessive browsing is obtained (U.S. Department of Agriculture 1994; Brown 1995). Treatments to stimulate aspen regeneration that are less than 2 ha (5 acres) may concentrate deer use and cause excessive browsing (Mueggler and Bartos 1977). In Colorado, harvesting several adjacent large (6 to 8 ha [15 to 20 acres]) units at one time resulted in successful aspen regeneration, even with large ungulate populations (Crouch 1983). Additional protection, such as temporary fencing, may be required to permit the growth of the aspen suckers beyond the reach of browsing animals if ungulate populations are high. Excessive browsing of sucker reproduction is perhaps the most detrimental influence on the successful regeneration of stands burned or otherwise treated to simulate suckering, as well as in stable aspen communities which usually produce ample suckers for regeneration as the aspen overstory gradually thins with age. Obtaining ample suckers to replace aspen stands is usually not a problem. Allowing these suckers to grow beyond the reach of ungulates is the issue (Shepperd and Fairweather 1994; Rolf 2001).

Various barriers to protect aspen regeneration under these conditions were evaluated by Kota (2005) in the Black Hills of South Dakota. He found that a barrier of hinged trees was useful in protecting aspen regeneration. Hinging is accomplished by felling the tree (live or dead) and keeping the bole on the stump, thus creating a barrier around patches of aspen regeneration. Standard fencing practices can also be used but are sometimes cost prohibitive, especially if large areas need to be protected. Kay and Bartos (2000) surveyed all known aspen exclosures on the Dixie and Fishlake National Forests in Utah in 1995 and 1996. The exclosures were originally established between 1930 and 1970 and were designed to exclude either all ungulates (livestock and wildlife) or just livestock. Aspen in the total exclusion exclosures successfully regenerated without other treatments and developed multi-sized, multi-aged stems. Aspen in the livestock-exclusion exclosures either failed to produce new stems greater than 2 m (6.5 ft) tall, or regenerated at lower densities than the total-exclusion plots. Outside the exclosures, aspen either failed to regenerate successfully or regenerated at lower densities than the livestock-exclusion plots. Aspen regenerated successfully on the livestock excluded plots and outside the exclosures only when mule deer populations were low.

Regenerating from Seed

The feasibility of seeding aspen in areas of abundant soil moisture is currently being tested (Shepperd 2001). If successful, this technique could be used to restore aspen where it has been lost from the system.

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Chapter 4

Pinyon/Juniper Woodlands

Robin J. Tausch and Sharon Hood

Introduction

Pinyon-juniper woodlands occur in 10 states and cover large areas in many of them. These woodlands can be dominated by several species of pinyon pine (*Pinus spp*. L.) and juniper (*Juniperus spp*. L.) (Lanner 1975; Mitchell and Roberts 1999; West 1999a). A considerable amount of information is available on the expansion of the woodlands that has occurred over large parts of the geographic ranges of the tree species involved (Miller and Tausch 2001). In southern Utah, the woodlands contain Utah juniper (*Juniperus osteosperma*), singleleaf pinyon (*Pinus monophylla*), and two-needled pinyon (*Pinus edulis*). Singleleaf pinyon is present in the western, and two-needled, or Colorado pinyon, in the eastern portions of the region. Each species can occur alone, or in a mix of one of the pinyon species with Utah juniper. Overall, information on woodland ecology for the southern Utah study area is limited. For this reason, the available literature for the woodlands in general, but particularly for the Great Basin, will be summarized and possible implications for southern Utah indicated.

Wherever they are found, pinyon and juniper woodlands are a landscape scale phenomenon (fig. 1). These trees have large ecological amplitudes and are capable of invading into, and dominating, a wide range of communities (Miller and Tausch 2001; West and others 1978a,b; West and others 1998). Their range can extend

Figure 1—Pinyon-juniper landscape with an old chaining treatment.

from the upper edge of salt desert shrub communities at the lowest elevations to the lower fringes of subalpine communities at the higher elevations (West and others 1998). The dynamics of these woodlands and their associated vegetation processes need to be understood at landscape scales. In the Great Basin, most of the communities where the trees have established were dominated by sagebrush (*Artemisia* spp.) (West and others 1978b). The woodland tree species are found associated with a range of sagebrush species and subspecies, and each taxon can represent a range of plant communities. Sagebrush presence usually extends in elevation both below and above the range of potential tree distribution, and their associated ecosystems are the matrix within which the trees occur. As a result, the dynamics of the woodlands are linked with, and in many ways dependent on, the dynamics of the shrub-dominated ecosystems involved.

Where they co-occur, sagebrush and woodland communities can have different states or levels of co-dominance within the overall successional dynamics of the sagebrush/woodland ecosystem complex of a particular landscape area. It is not possible to really understand or manage pinyon and juniper at these landscape scales without understanding the entire topography, soils, and vegetation complex for each landscape area of interest. Because this ecosystem complex is dynamic and highly variable across the landscape, identification of community type is determined from the species composition of the associated, usually sagebrush-dominated, communities (West and others 1978b; 1998). At any woodland location on the landscape, its successional status and associated ecosystems are the result of complex interactions of topography, soils, environmental conditions, past patterns of disturbance, and how successional processes have operated through time. In some locations in southern Utah, Gambel oak (*Quercus gambellii*) can be a part of the community (Thompson 1999; West 1999b) and can influence community dynamics.

Historical Conditions

How the patterns of disturbance were spatially distributed across the landscape and the subsequent successional changes through time following those disturbances were much different prior to Euro-American settlement than afterward. Prior to settlement, the primary disturbance was fire (Gruell 1999; Miller and Tausch 2001). The pattern and behavior of fire was closely related to the unique interactions of topography, soils, environmental conditions, and vegetation composition present at that time on each landscape area of interest. These complex community types contained an equally complex mix of fuel types and levels that determined fire pattern and behavior. Across the Great Basin region, trees were present on less than one-third of the area they currently occupy. For areas where trees were present, their pre-Euro-American settlement densities were on average about one-fourth to one-tenth or less of the density present at the beginning of the twenty-first century (Bauer 2006; Miller and others, in press). Similar densities appear to have been present on the eastern Colorado Plateau portion (Floyd and others 2000; Romme and others 2003) of the southern Utah study area.

Vegetation cover prior to pre-European settlement varied widely, depending on local conditions, from less than 20 percent to over 80 percent on the most productive sites (West and others 1998). The majority of sites were below 50 percent total cover (Miller and Tausch 2001). Similar averages and variability in cover appear to have been present in the southern Utah area. Total vegetation cover appears to have always been relatively the same for similar sites, whether they were sagebrush-dominated or tree-dominated. Total biomass, however, varied from about seven to nearly 20 times greater when a site was tree-dominated versus sagebrush-dominated (Tausch and Tueller 1990).

The mix of sagebrush- and tree-dominated sites over the pre-Euro-American settlement landscape and the distribution of size and age classes within tree-dominated sites depended on the interactions between disturbance patterns and post-disturbance successional development. The primary control on these differences appears to be landscape variation in the pattern and frequency of fire. The heterogeneity of the landscape, combined with variation in successional processes, associated heterogeneous mix of community types, and associated fuel types and fire regimes, resulted in the maintenance of vegetation that varied widely across the landscape.

Pre-settlement old growth woodlands were commonly found on relatively fire safe sites with limited surface fuel loads (fig. 2) (Burkhardt and Tisdale 1969; Burwell 1998, 1999; Holmes and others 1986; Miller and Rose 1995, 1999; West and others 1998). The high level of landscape and associated vegetation heterogeneity present prior to European settlement resulted in a high degree of edge between sagebrush and tree-dominated communities (Tausch and Nowak 1999). These heterogeneous conditions often represented optimum habitat for many species of wildlife (Miller and Tausch 2001).

Figure 2—The presence of very old Utah juniper (*Juniperus osteosperma*) suggests that this rocky site would rarely, if ever, develop a grassy understory capable of carrying a surface fire.

Then, as now, larger fires tended to occur during periods of drought (Betancourt and others 1993; Swetnam and Betancourt 1998). Insects, diseases, and native ungulates appear to have played a widespread but relatively minor role. Information is more limited for the Colorado Plateau than the Great Basin, but it indicates fire may have been less frequent in many areas compared to the Great Basin (Floyd and others 2000; Romme and others 2003). Overall, there was a dynamic balance between disturbance and succession resulting in a complex shifting distribution of the woodland and sagebrush dominance throughout the landscape.

It is the interaction between topography, vegetation, and fire that influenced both the patterns of disturbance and the kinds of communities that were found on a particular position on the landscape at a particular point in time. The deeper soils in the canyon bottoms and swales are generally more productive, particularly for the herbaceous species. These locations appear to have had the highest fire frequencies (Bauer 2006; Burwell 1998; Gruell 1999; Swetnam and Basian 1996). As soils become shallower, generally as the topography becomes steeper, the abundance of perennial herbaceous vegetation is limited to years with above average precipitation. On these locations, fires appear to have been less frequent, increasing the probability of dominance by trees. The most fire-safe sites, generally on the steepest slopes or shallowest soils, were generally the locations of woodlands that were often several centuries old (Miller and others 1999; Waichler and others 2001). These sites also have generally low levels of productivity of perennial herbaceous vegetation. A few pre-settlement aged woodlands appear to be present from nothing more than the off-chance of not having burned for over 200 years (Miller and others, in press).

Current Conditions

Euro-American settlement activities have caused major changes to the composition of vegetation within the Great Basin (Miller and Tausch 2001; Rowland and Wisdom 2005). The rapid woodland expansion observed during the late 1800s and early 1900s resulted from a combination of conditions (Miller and Tausch 2001; Miller and others, in press): (1) heavy livestock grazing that removed the herbaceous vegetation (fine fuels), (2) the associated reduction in the presence of fire (Heyerdahl and others 2001; Savage and Swetnam 1990; Swetnam and Betancourt 1998), and (3) wet conditions that created an ideal situation for tree establishment (Antevs 1938). The resulting expansion and increasing dominance of the trees in the Great Basin has continued to the present (Burkhardt and Tisdale 1976; Cottam and Stewart 1940; Miller and Rose 1995, 1999; Miller and others, in press; Tausch and others 1981).

Livestock grazing, particularly in the late 1800s and early 1900s (Young and Evans 1989), generally had the largest impact on the vegetation composition. Grazing reduced the herbaceous vegetation cover, which resulted in a reduction in fire frequency (Burkhardt and Tisdale 1969, 1976; Campbell 1954; Ellison 1960; Miller and Rose 1999; Miller and Tausch 2001). The reduction of herbaceous species by grazing also promoted an increase in shrub cover. The shrubs acted a nurse crop and promoted tree seedling establishment (Burwell 1998, 1999; Chambers and Vander Wall 1999; Chambers and others 1999; Chambers 2001; Cottam and Stewart 1940; Eddleman 1987; Madany and West 1983; Miller and Rose 1995, 1999). With the reduction in fire frequency, the new tree seedlings were able to survive and the areas of woodlands expanded. As with pre-settlement woodlands, total vegetation cover of expansion woodlands remains relatively similar to the shrub cover that preceded tree dominance (Tausch and Tueller 1990; Tausch and West 1995). Therefore, when the shrub layer was absent, the establishment of the trees was more limited (Erdman 1970; Everett and Ward 1984).

Much of the woodland expansion has been into the more productive sites (for example, canyon bottoms and swales). In the absence of fire, the trees are well adapted and competitive in these more productive locations. Prior to tree expansion these areas represented some of the more diverse and productive sagebrush ecosystems in the region and currently support, or will support, some of the highest levels of tree dominance and fuel loads. Pre-Euro-American settlement woodlands have had up to a 10-fold increase in tree densities during this period (Bauer 2006; Miller and others, in press). Density increases may be less on the eastern Colorado Plateau portion (Floyd and others 2000; Romme and others 2003) of the southern Utah study area. As the area of tree dominance continues to increase in the Great Basin, the heterogeneous sagebrush-dominated ecosystems are being replaced by homogenous woodlands (fig. 3) (Miller and Tausch 2001; Milne and others 1996; Tausch 1999a).

Before about 1870, woodlands occurred on less than 10 percent of their currently occupied area in the northwest Great Basin (Miller and others 1999) and on less than 30 percent in the central and southern Great Basin (Creque and others 1999; Miller and others, in press; Miller and Tausch 2001; O'Brien and Woodenberg 1999; Rogers 1982; Tausch and others 1981). Little information is available on the pre-settlement woodlands of the Colorado Plateau. Expansion woodlands now cover an average of three to four times the pre-Euro-American settlement area (Chambers and others 2000a, b; Miller and others, in press). This woodland expansion has proceeded at a nearly continuous rate across the Great Basin over the last 100 years (Chambers and others 2000a, b; Miller and others, in press) and possibly equals or exceeds previous woodland expansions of the Holocene (Miller and Wigand 1994). Consequently, sagebrush communities will continue to decline as this tree dominance continues to increase (Despain and Mosley

Figure 3—Initial establishment of Utah juniper (*Juniperus osteosperma*) in a stand of big sagebrush (*Artemisia tridentata*).

1990; Miller and others 1994; Miller and Tausch 2001; Suring and others 2005; Tress and Klopatek 1987; West 1984).

Because of the generally slow growth of the trees, it has taken all of the approximately 100 years since Euro-American settlement for a doubling of the fuel loads to take place. Trees in the extensive areas of woodlands that have established over the last century are now rapidly maturing, and as they do, the fuel loads are increasing at an accelerated rate on these sites. On the majority of these areas, the density needed for trees to dominate is now in place (Miller and others, in press). While it took fuel loads in the expansion woodlands the past 100 years to double (Chambers and others 2000a, b), they will double again in the next 40 to 50 years (Miller and Tausch 2001). The expansion of tree distribution into new sagebrush areas is continuing (Betancourt 1987; Knapp and Soule 1998; Miller and others 2000; Miller and Tausch 2001; Suring and others 2005; West and Van Pelt 1986), and with it continues the increase in the level and continuity of tree-dominated fuel loads. Similar patterns appear to exist in southern Utah.

The rate of the transition from sagebrush ecosystem to tree-dominated woodland is variable and depends on the site potential, sagebrush species and subspecies present, and rate of tree establishment (Miller and Tausch 2001; Miller and others, in press). In general, a minimum of 60 to 90 years is required for trees to dominate a site (Barney and Frischknecht 1974; Huber and others 1999; Miller and others 1999; Miller and Rose 1995; Miller and Tausch 2001; Tausch and Tueller 1990; Tausch and West 1995). The decline in sagebrush biomass is not proportional to the increase in tree biomass (fig. 4). When the trees have reached about one-half their potential biomass, sagebrush biomass has declined to about one-third, sometimes one-fourth, of its former level (fig. 5) (Miller and Tausch 2001; Tausch and West 1995). The pattern of the decline is relatively consistent across the sagebrush species and subspecies, although the rate involved is not (Miller and others, in press; Miller and Tausch 2001). This expansion may be facilitated by the increasing CO_2 in the atmosphere (Johnson and others 1993). Similar changes in the landscape level patterns are present in the woodland changes of southern Utah.

As the dominance of the trees continues to increase beyond Phase I or Phase II (fig. 5), not only will fuel loads double from current levels over the next few decades, but the continuity of those fuels across the landscape will rapidly increase. Because of the young age of the trees, the ongoing increases in fuel loads are primarily in the highly flammable fine fuels. Once the trees dominate a site, these fine fuels can reach 9,000 kg/ha (10 tons/acre) on more productive sites (Chambers and others 2000a, b). Overall, woodland sites in the Great Basin vary widely, but probably have average fuel levels of about two-thirds of those sampled on the more

Figure 4—Increasing dominance and water use by Utah juniper (*Juniperus osteosperma*) are the likely cause of the die-back of big sagebrush (*Artemisia tridentata*) on this site.

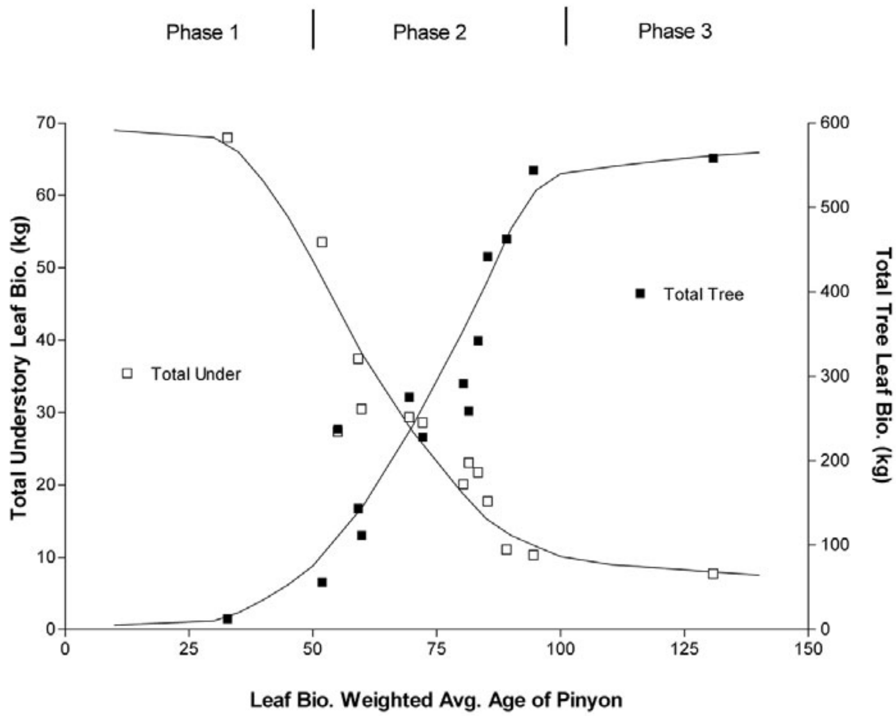

Figure 5—Comparison of both the total tree leaf biomass (closed boxes) and total understory leaf biomass (open boxes) over time as indexed by the range in leaf biomass weighted average age of pinyon for 14 plots in southwestern Utah (Tausch and West 1995; Miller and Tausch 2001). X-axis = (sum of [tree age * leaf biomass] over all trees) / total stand leaf biomass. Phase I is the early tree establishment phase, Phase II is the period of tree growth and increasing tree dominance, and Phase III is tree dominance in expansion woodland sites.

productive central Nevada sites. The increasing crown closure of post-settlement woodlands is increasing the occurrence of crown fire (Miller and Tausch 2001; Tausch 1999a,b; West 1999a,b). Similar patterns appear to be taking place in southern Utah.

Post-fire vegetation response depends on the composition of the shrub-dominated community and the level of tree dominance (Barney and Frischknecht 1974; Dhaemers 2006; Erdman 1970; Everett and Ward 1984; Pickett 1976; West and VanPelt 1986). As the trees dominate a site, there is a decrease in the herbaceous species (Dhaemers 2006; Erdman 1970; Koniak and Everett 1982; West and others 1978a; West and others 1998), an increase in soil erosion (Wilcox and Breshers 1994), changes in soil fertility (Rau 2005), losses in forage production, and changes in wildlife habitat (Miller and Tausch 2001). The more dominant the trees are at the time of disturbance, the more the plant species composition of the communities that follow the disturbance can change. The intense crown fires more frequently occurring on tree-dominated sites further reduces understory plant species survival (Tausch 1999a).

Exotic species are changing the outcome of post-fire response (D'Antonio and Chambers 2006). The higher the level of tree dominance, the higher the probability that a crown fire will leave an open site. These open sites are increasingly being dominated by exotic plant species, many of which are annuals, such as cheatgrass (*Bromus tectorum*) (Chambers and others 2007; Roundy and others, in press). The presence of cheatgrass can cause increases in fire size and frequency (D'Antonio and Chambers 2006; Swetnam and others 1999; Tausch 1999b; Whisenant 1990; Young and Evans 1973) and homogeneity of those communities across the landscape (Young 1991; Young and Evans 1973). More recently, exotic perennials have begun establishing in these areas (D'Antonio and Chambers 2006). Once this conversion occurs, any return to the original sagebrush ecosystem, or even eventually to woodland, is often no longer possible without extreme restoration efforts (Miller and Tausch 2001). Our ability to assess the susceptibility of the communities associated with different sagebrush species and subspecies to conversion to cheatgrass is improving (Chambers and others 2007; Roundy and others, in press).

The problems associated with the exotic grasses are increasing through time in proportion to the level of CO_2 in the atmosphere. Three ecotypes of cheatgrass from three elevations in the Great Basin have been investigated for the effects of increasing levels of atmospheric CO_2. Four levels of CO_2 were used, ranging from pre-settlement to an estimate of 2020 levels. From pre-industrial levels to the estimated 2020 level, the productivity of the upper elevation ecotype doubles, the mid-elevation increases 2.5 times, and the low-elevation triples (Ziska and others 2005). Flammability also increases (Blank and others 2006). Cheatgrass will have an increasingly negative impact over time on any woodland site where it becomes established following fire.

Prior to European settlement, ecosystems in the Great Basin and southern Utah were, on a landscape basis, more resilient to disturbance by fire. Fire regimes differed between the different sagebrush species and subspecies-dominated ecosystems and their associated communities, but were relatively consistent within each community. With tree expansion and increasing dominance there has been an increase in homogeneity and a loss of resiliency. Even old growth woodland areas that were relatively protected from fire prior to settlement are increasingly susceptible to fire damage from the adjacent tree-dominated areas (Miller and Tausch 2001) that increasingly dominate fire behavior (Hann and others 2004). Developing effective restoration procedures requires more study on the ecology, structure, and long-term dynamics of the woodlands and their interaction with the associated sagebrush ecosystems (Chambers 2005; Dhaemers 2006; Miller and others, in press).

Fire Regimes

Five fire regime classes have been defined for assessing landscape dynamics for historic or past fire patterns and frequency (table 1) (Hann and others 2004; Hann and Strohm 2003; Romme and others 2003; Schmidt and others 2002; Waichler and others 2001). Class I was very rare in the southern Utah area prior to Euro-American settlement and is not covered here. Prior to settlement, a heterogeneous mix of fire regime classes II through V often existed within relatively

Table 1—Natural (historical) fire regime classes from Hann and others (2004) as interpreted by the authors for this analysis.

Fire regime class	Frequency (mean fire return interval, in years)	Severity	Community structure description
I	0 to 35 frequent	Surface mixed	Open woodland or savannah structures maintained by frequent fire; also includes frequent mixed severity fires that create a mosaic of different age post-fire open woodland, early to mid seral woodland structural stages, and shrub or perennial grass dominated patches.
II	0 to 35 frequent	Replacement	Shrub or shrub/perennial grass maintained or cycled by frequent fire: fires kill non-sprouting shrubs, such as sagebrush, which typically regenerate and become dominant within 10 to 20 years; fires remove tops of sprouting shrubs such as rabbitbrush, which typically resprout and can dominate after several years; fires typically kill most tree regeneration.
III	35 to 200 infrequent	Mixed surface	Mosaic of different age post-fire woodland, early to mid-seral (Phase I and Phase II, fig. 6) woodland structural stages, and shrub or shrub/perennial grass dominated patches maintained or cycled by infrequent fire.
IV	35 to 200 less frequent	Replacement	Large patches of post-fire shrub or shrub/perennial grass dominated structures, or early to sometimes late seral (Phase I to Phase III, fig. 6) woodland cycled by infrequent replacement fire.
V	> 200 rare	All types	May have large patches of similar post-fire shrub or shrub/perennial grass dominated structures, or early to usually late seral (Phase I to usually Phase III, fig. 6) woodland cycled by rare replacement fire. In systems with little fire or only localized torching effects of lightning strikes effects the composition and structure may be complex.

small areas of the landscape. Because of the vegetation heterogeneity that existed prior to settlement, the abundance and distribution of the various types were first controlled by topographic heterogeneity and secondly, were both determined and controlled by the vegetation heterogeneity. Some separation of, or differences in, fire regime probably existed between the sagebrush species and subspecies present prior to settlement reflecting differences in their specific site conditions. Even with the abundance and widespread distribution of areas with the fire regimes classes II, III, and IV, large areas representing fire regime V, often old growth woodlands, were still present and widely distributed within, and often surrounded by the other, fire regimes.

Since Euro-American settlement, increasing homogeneity of the vegetation has resulted in increased fuel loads and continuity. Areas that were in fire regime V are now in fire regime IV, or sometimes even III. This trend will continue as the surrounding vegetation changes. The vegetation heterogeneity that resulted from differences between sagebrush species and subspecies is generally disappearing (Miller and Tausch 2001). Many areas are increasingly at risk of fire and increased fire size and frequency (Hann and Strohm 2003; Swetnam and others 1999). Despite the increasing appearance of homogeneity with woodland expansion, the vegetation response following a crown fire can still be driven as much by the differences between the sites identified by the original sagebrush community as by the level of tree dominance.

Fire Regime Condition Classes

Three Fire Regime Condition Classes (FRCC) have been defined for assessing the departure of current vegetation communities from historical vegetation structure and fire patterns (chapter 1, table 2, this volume) (Hann and Strohm 2003). Overall, the appearance of a particular woodland site and its associated area of the landscape determine the effective FRCC. This is because the context of the surrounding landscape, particularly where it is represented by expansion woodlands, can drive fire behavior and severity independent of the conditions of a particular old growth woodland site (Hann and others 2004). Effectively restoring a mix of sagebrush and woodland dominance at the landscape level also requires the restoration of the former landscape heterogeneity that makes a dynamic stability with fire possible.

Prior to settlement, most of the sagebrush/woodland areas of the Great Basin, and apparently southern Utah as well, were in FRCC 1. Woodlands and sagebrush ecosystems were in a dynamic balance from areas that burned two to three times a century, to areas that burned about once a century, to areas where fire occurred at intervals greater than 200 years. Areas that burned more frequently were sagebrush and bunchgrass dominated at the time of the fire, although some trees may have established after a previous fire. Because of the ongoing changes, most of the Great Basin is at least in FRCC 2. Many sites may already be in FRCC 3 or are rapidly approaching that condition.

For many woodlands, FRCC sometimes does not depend so much on what an individual pinyon-juniper stand looks like, but on the probable pre-settlement community composition and the current landscape context within which it is located. For example, an area that was old growth woodland prior to settlement could be in FRCC 1 when surrounded by sagebrush-dominated communities. The surrounding areas remained in sagebrush because those communities supported more frequent fire of lower intensity. Under these conditions fires, usually did not crown into the adjacent old growth woodlands and they appear to have remained relatively fire safe (fig. 6). Most of the expansion woodlands, however, are occurring in areas that were usually sagebrush-dominated prior to settlement, changing the pattern and behavior of fire compared to what occurred prior to tree dominance. These tree-dominated expansion woodlands often have continuous canopy cover, which can support high intensity crown fires under high wind conditions. These adjacent, expansion woodland sites can now drive fire behavior (Hann and others 2004). As a result, the old growth woodland stand becomes FRCC 3, even where little change to the woodlands has occurred. However, many of these pre-Euro-American settlement woodlands have also experienced increases in tree density, sometimes up to 10 times or more over the last century (Bauer 2006; Miller and others, in press), which is also directly changing the Fire Regime Condition Class of these sites. Combined, these vegetation changes are resulting in changes in the size, intensity, and frequency of fires for all parts of the landscape. This includes the increasing risk for conversion to cheatgrass.

For the Great Basin as a whole, little of the current woodland area is in FRCC 1. These are the pre-settlement woodlands that have not seen a significant increase in tree density and may represent less than 10 percent of today's total (Bauer 2006; Miller and Tausch 2001). Future loss of some amount of current sagebrush-dominated ecosystems to tree encroachment is still possible (Suring and others 2005). The situation in southern Utah appears

Figure 6—An old growth Utah juniper (*Juniperus osteosperma*) growing with pinyon pine (*Pinus* spp.). The tree likely reached this age because of inadequate surface fuels to carry high intensity fire and stand density was too low to support crown fires. The increased tree density is increasing the risk of lethal fire.

to be similar. Depending on location and community, current vegetation attributes put one-half to two-thirds of the current woodland/sagebrush ecosystem complex in the Great Basin in FRCC 2. Most of these woodlands that are currently in FRCC 2 will be rapidly transitioning to FRCC 3 over the next 40 to 50 years. This has serious implications for habitat for multiple species (Wisdom and others 2005). Similar changes appear to be occurring in southern Utah with similar consequences.

Recommended Treatments

The goals of treating woodlands include fuel load reductions, restoration of sagebrush communities, increasing the heterogeneity of the landscape and associated disturbance processes, improving watershed protection, enhancing wildlife habitat, and increasing forage production (Miller and Tausch 2001). The locations of the treatment sites or patches should be based on topographic features and areas that tended to have a higher fire frequency and thus, historically were more likely dominated by sagebrush communities. These are areas with deeper soils and higher herbaceous vegetation productivity that can carry fire. Retaining pre-settlement woodland sites requires as much or more effort to restore the surrounding communities as it does to restore the pre-settlement site.

Many treatment procedures have been attempted, but they have often been unsuccessful over the long term because of the lack of information about treatments (Chambers 2005; Tausch and Tueller 1977, 1995). A focus on landscape scales, rather than on just individual project scales, can improve treatment effectiveness (Hann and Bunnell 2001). Central to this has been the general lack of recognition of the variability of the communities that the trees are capable of dominating, and the range of disturbance histories represented by the previous communities. Because there is even less direct information available for the southern Utah woodlands, the distribution and extent of similar conditions and patterns of change in the area need to be determined on a site-specific basis.

Tree Removal

Tree removal is the primary management option for restoring areas affected by the ongoing woodland expansion. However, additional treatments have been proposed, many of them using new techniques. First and second order effects, and the success and longevity of the outcomes of any treatment, are highly specific to the site and the method used, how the treatment is used and its timing. For a detailed description of common treatments in pinyon-juniper communities, refer to the restoration chapter in Monsen and Stevens (1999) and Monsen and others (2004). However, some general guidelines are becoming apparent as the results from past and current studies improve our understanding of how these treatments interact with the vegetation dynamics in the woodland zone (Chambers 2005; Miller and Tausch 2001; Monsen and others 2004). Applied in the right way, at the right place, and at the right time with the proper follow up, if needed, any of the existing or proposed treatments can have positive outcomes.

Prescribed Fire

Prescribed fire may be used to remove trees and restore sagebrush communities before tree dominance is so high it reduces surface fuels to a low enough level that they cannot carry fire. Once tree dominance is at the high levels of late Phase II or Phase III (fig. 5) for an extended period of time, susceptibility to the establishment of exotics such as cheatgrass increases. Once these levels of dominance are reached, some form of mechanical treatment followed by seeding is necessary to reduce the level of tree dominance (Chambers 2005). This allows recovery of sagebrush and herbaceous vegetation before the use of prescribed fire can more fully restore a sagebrush ecosystem.

Prescribed fire in pinyon-juniper has been used to control the establishment of trees, increase forb productivity, increase habitat diversity, control invasion of other conifers, alter herbivore distribution, enhance forage palatability and nutritive quality, and prepare sites for reseeding (Bunting 1990). While prescribed fire can be beneficial, many limitations exist. Vegetation response following fire depends on the composition of the shrub community on a site and the level of tree dominance (Barney and Frischknecht 1974; Dhaemers 2006; Erdman 1970; Everett and Ward 1984; Monsen and others 2004; Pickett 1976; West and VanPelt 1986). As trees increasingly dominate a site, the associated sagebrush ecosystems are greatly reduced (Chambers 2005; Erdman 1970; Koniak and Everett 1982; West and others 1998). This reduction in fine fuels often makes it difficult for a fire to carry through a mid-successional stand. If fire does occur, increasing tree dominance increases the recovery time of herbaceous plants and increases the potential for invasion of exotic plants and erosion (Bunting 1990; D'Antonio and Chambers 2006). Bruner and Klebenow (1979) developed an index to predict when fire will carry through mid-successional pinyon and juniper based on wind speed, shrub and tree cover, and air temperature. Dangerous burning conditions exist when the index is greater than 130. Optimal prescribed burning conditions are an index between 125 and 130. This can be modified by fuel moisture levels. Tree-dominated woodlands can be easier to burn than the mid-successional woodlands and are increasingly carrying large crown fires (Miller and Tausch 2001).

Mechanical Thinning

Chaining and thinning are the most commonly used mechanical methods to reduce tree cover. This may be necessary prior to prescribed burning in order to reduce crown fuels and stimulate understory vegetation. In Spanish Fork Canyon, UT, chaining increased total ground cover from 47 to 80 percent and forage production from < 22 kg/ha (<20 lbs/acres) to 1,120 kg/ha (1,000 lbs/acres) 7 years after treatment (Chadwick and others 1999). Similar increases were seen between 4 and 7 years after chaining in eastern Nevada (Tausch and Tueller 1977, 1995) This initial increase in ground cover resulted in significantly less runoff and soil erosion than the control area (Farmer and others 1999). The size, type, and arrangement of the chain can be varied to accomplish different objectives and control the size and amount of trees removed. Stevens and Monsen (2004) provide basic guidelines for chaining in pinyon-juniper. Double chaining in opposite directions removes additional trees missed in the first pass and covers the seed after the area has been broadcast seeded prior to the second pass (Stevens 1999a). A once over chaining is appropriate if sufficient understory remains, trees are sparse and mature, and seeding is not required (Stevens 1999b).

Chaining for tree control increases herbaceous biomass, but can be short-lived. Often after the 4- to 7-year increase there can be a rapid decline to pre-chaining levels in 25 years as a result of accelerated growth of surviving trees (Tausch and Tueller 1977, 1995). Although usually a stand alone procedure, chaining should generally be used only as an effective first treatment followed by a second treatment, such as prescribed fire, which would remove the surviving trees.

Thinning overstory trees with handsaws reduces tree cover and causes less soil disturbance than chaining (Loftin 1999). In a case study in New Mexico, Loftin (1999) reported 2.5 times greater herbaceous cover two growing seasons after hand felling juniper trees without seeding. This method can also be marketed as a fuelwood sale to offset costs. In an economic comparison of chaining versus thinning using chainsaws, Chadwick and others (1999) found thinning cost 44 percent more than chaining. In the same study, thinning did not create an effective seedbed, and subsequent forage production was low compared to the chaining treatment. The different responses between the two studies are most likely due to differences in pre-treatment site conditions. This underscores the importance of choosing appropriate site-specific treatments. Hand thinning can be as equally short-lived as chaining and should also be considered as either a pre-treatment procedure before prescribed fire, or a regularly repeated treatment.

Mastication is another increasingly popular mechanical thinning method. This method grinds and chips trees to reduce tree cover and compact fuel beds. Over 13,360 ha (33,000 acres) have been masticated in Utah alone (Bruce Roundy, personal communication). Because mastication is such a new treatment, the ecological effects are largely unknown and warrant future research.

Seeding

Seeding may be required to prevent the establishment of exotic weeds if the understory is depauperate (Thompson and others 2006). After a tree removal treatment, seeding should occur prior to the next growing season to restrict the establishment of exotics (Stevens and Monsen 2004). Fall seeding is the most ideal time to seed in the Intermountain West, although in southern Utah, seeding just prior to mid-July monsoons has also been successful (Stevens 1999b). Fixed wing aircraft, helicopters, or rangeland drills are normally used for seeding. Aerial seeding treats large areas on steeper slopes or where tree densities are high. Drill seeding is used in open areas (Thompson and others 2006). Aerial seeding followed by chaining after fire significantly increased seeded grass cover and decreased cheatgrass cover compared to seeding alone (Ott and others 2003).

Historically, introduced species seed mixes were used to control soil erosion and forage production. In recent years, there has been more interest in using native seed mixes to increase species diversity and restore ecosystems (Richards and others 1998). Successful establishment of native grasses and forbs from different seed mixes has been demonstrated in several recent studies (Ott and others 2003; Thompson 2006; Waldron and others 2005).

Herbicides

Herbicides to control encroaching pinyon and juniper trees are another alternative to reduce tree cover. Basal spraying of herbicides allows for highly selective application with little effect on non-target species. Tebuthiuron (Spike® 80W) and picloram (Tordon® 22K) are commonly used herbicides in these systems. Parker and others (1995) tested the two chemicals' efficacies in controlling pinyon and juniper trees using basal application under different concentrations. Control was best for trees less than 1.8 m (6 ft) in height, with picloram killing over 90 percent of the sprouts and seedlings. Tebuthiuron, a slower acting herbicide, killed 30 to 60 percent of the sprouts and seedlings after 9 months, but results were expected to improve over time. Mortality of trees taller than 1.8 m (6 ft) was between 10 and 30 percent for picloram and 5 and 10 percent for tebuthiuron (Parker and others 1995). Johnsen (1986) states that individual tree application is best suited for newly invaded sites with fewer than 500 trees/ha (200 trees/acre) under 1.8 m (6 ft) tall. The longevity of these treatments will depend on the number and age class of the trees removed. Concentrating only on the older trees and leaving many of the younger trees will reduce the longevity of the treatment.

Broadcast application of tebuthiuron and picloram produce more variable results. One-seed juniper (*Juniperus monosperma*) and Rocky Mountain juniper (*J. scopulorum*) are often the most difficult to control (McDaniel and WhiteTrifaro 1986). Johnsen (1986) reported that herbicides readily killed trees on the ridges, but not on areas of deep soils or bottom land. Trees along the ridges are often old growth pockets of pinyon and juniper that generally should not be a target for removal. Areas invaded by pinyon and juniper, where herbicides are not as effective, are also places where fire would have historically limited their establishment. These are the areas often needing treatment. Concern over killing non-target species, with potentially limited mortality of pinyon and juniper, makes this treatment less desirable than individual tree application.

Summary

Management goals that deal with woodland expansion need to account for the landscape variability in community composition and disturbance regimes (Miller and Tausch 2001). Vegetation treatments should also focus on the source of woodland changes. In other words, they should focus on areas of the woodlands that represent expansion beyond the pre-settlement distributions. Within these areas, the focus should then be on woodland sites that have only recently transitioned away from FRCC 1. There are areas of recent tree expansion with vegetation attributes that indicate they may still be returned relatively easily to FRCC 1. These communities include the remaining sagebrush-dominated areas, sagebrush areas still in early Phase I (fig. 5), and the areas of old growth woodlands that are present within the landscape matrix. The sagebrush ecosystems and their associated communities, successional classes, and distributions need to be determined on a landscape basis for each management area.

There is, however, another management reality. With at least two-thirds of the woodland area in the Great Basin (and probably in southern Utah as well) representing expansion woodlands, millions of acres are now involved. Even under the best of conditions, only a minority of such a large area will be successfully treated before a wildfire occurs. For the remaining areas that are being increasingly dominated by trees that will burn before treatment is possible, we need to determine restoration/rehabilitation needs and possibilities following wildfire, particularly when cheatgrass or other exotics are present. Additional research is needed to help with the development of these procedures.

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Chapter 5

Big and Black Sagebrush Landscapes

Stanley G. Kitchen and E. Durant McArthur

Introduction

Perhaps no plant evokes a common vision of the semi-arid landscapes of western North America as do the sagebrushes. A collective term, sagebrush is applied to shrubby members of the mostly herbaceous genus, *Artemisia* L. More precisely, the moniker is usually restricted to members of subgenus *Tridentatae*, a collection of some 20 woody taxa endemic to North America (Beetle 1960; McArthur 1979; McArthur and Plummer 1978). As a group, the *Tridentatae* are distinguished from other members of the genus by a combination of traits including their woody habit, floral morphology, stem anatomy, plant chemistry, and chromosomal karyotype (McArthur 1979).

The genus *Artemisia* originated on the Eurasian landmass during the mid-Tertiary as the late-evolving Asteraceae rapidly diversified in response to global expansion of drier and cooler habitats (Beetle 1979; Raven and Axelrod 1974). Toward the end of this period, herbaceous, mesic-adapted progenitors to contemporary *Tridentatae* migrated across Beringia and spread across western North America eventually developing a woody habit (Beetle 1979; McArthur 1999; McArthur and Plummer 1978; Stebbins 1972). Opportunities for continued diversification were plentiful in the spatially and temporally diverse environment provided by the interaction of a complex geography with the increasingly variable climate of the Quaternary (Beetle 1979; McArthur and Plummer 1978). Over time, one widely adapted species, big sagebrush (*Artemisia tridentata*), emerged as the "most widespread and common shrub of western North America" (fig. 1) (McArthur and Stevens 2004).

Figure 1—A big sagebrush (*Artemisia tridentata*) landscape.

Scientific and management interest given to sagebrush in general, and big sagebrush in particular, can be measured by the considerable volume of literature generated primarily during the last half-century. McArthur and others (1979), Blaisdell and others (1982), and McArthur and Stevens (2004) provide useful reviews of the ecology and management of sagebrush species and ecosystems. Various papers presented in thematic symposia (McArthur and Welch 1986; Utah State University 1979) were effective in synthesizing available knowledge, and although dated, the published proceedings remain valuable reference materials. Literature summaries are available online by species or subspecies from the USDA Forest Service, Fire Effects Information System data base (http://www.fs.fed.us/database/feis/) (Howard 1999; Johnson 2000; McMurray 1986; Tirmenstein 1999a). Welch's (2005) big sagebrush synthesis is the latest and most comprehensive review published for this species complex.

Our purpose is not to provide yet another general sagebrush review, but to explore relevant published work and current thought regarding structure and successional processes, particularly as they relate to fire in ecosystems dominated by big sagebrush, and to a lesser extent, black sagebrush (*A. nova*). Although we briefly consider paleo distribution to provide context, our temporal focus will be primarily the last two centuries with emphasis on systematic changes that resulted from Euro-American settlement in the mid- to late-1800s. The time periods before and after this major cultural/ecological shift will be referred to as pre- and post-settlement. The geographical area of interest is represented by a broad zone in central and southern Utah where the Great Basin and Colorado Plateau meet, referred to here as the Southern Utah study area, or simply, the study area. Black sagebrush is included in the discussion because of its broad ecological overlap with big sagebrush and because of its widespread distribution and importance in the study area. In certain cases, such as the assessment of paleo and historic distribution, information is lacking for clear distinctions among sagebrush taxa. However, we believe that inferences made for the group as a whole will largely hold true for its dominant member, big sagebrush, and the closely allied black sagebrush.

Sagebrush Biology and Reproduction

Big sagebrush is a medium- to long-lived (20 to 200+ years; McArthur and Stevens 2004; Perryman and others 2001) aromatic evergreen shrub with one to several main stems (McArthur and Stevens 2004). The gray to black bark on older branches is shredded and shaggy (Beetle 1960; McArthur and others 1979). Typical persistent leaves are small, pale green to blue-green, narrowly wedge-shaped, with three blunt teeth on the broadened end (McArthur and Stevens 2004). Spring ephemeral leaves are larger and generally more variable in shape and size than persistent leaves (Miller and Shultz 1987). Leaves and young stems are covered by a mat of fine hairs that provide a silvery cast. Inconspicuous, wind-pollinated flowers are held above foliage on fine, more or less erect stems. The wind-dispersed seeds (achenes) are small (4,000 to 6,000 seeds per g) and lack specialized appendages (Meyer and others 1988a; Welch 2005). Dispersal distance varies with topography and local conditions. Maximum dispersal distance for sagebrush seeds has been estimated at 30 m (98 ft) (Johnson and Payne 1968; Walton and others 1986), although the majority of seeds generally disperse less than 3 m (10 ft) from the mother plant (Walton and others 1986; Wambolt and others 1989). Seeds are short-lived and do not form a persistent seed bank (McDonough and Harniss 1974; Meyer 1990; Young and Evans 1989).

Three widely recognized subspecies of big sagebrush differ in a number of morphological and physiological traits (McArthur and Stevens 2004). Basin big sagebrush (ssp. *tridentata*) is the tallest, typically 1 to 2.5 m (3.3 to 8.2 ft), and has an uneven-shaped crown. The crown shape of Wyoming big sagebrush (ssp. *wyomingensis*) is similar to that of basin big sagebrush, however, plants are generally less than 1 m (3.3 ft) tall. Mountain big sagebrush (ssp. *vaseyana*) is intermediate in height at 0.6 to 1.5 m (2 to 5 ft). Seed stalks and foliage are of even height above the crown and give it a more flattened appearance than the other subspecies. Seed maturation and dispersal vary among subspecies and stand elevations (McArthur and Stevens 2004; Welch 2005) and are generally latest for basin big sagebrush (November to early December) and earliest for higher elevation populations of mountain big sagebrush (late September to October). Seed production is greatest for basin big sagebrush and least for Wyoming big sagebrush. Significant variation in palatability and nutritive content to wildlife and livestock has been documented and related to differences in the chemistry of secondary compounds (Welch and McArthur 1981, 1986; Welch and others 1981, 1987; Sheehy and Winward 1981). Dormancy in recently dispersed seed correlates with climate (elevation) of the collection site (Meyer and Monsen 1991, 1992). Although similar in many ways, black sagebrush differs from big sagebrush in a number of traits including shorter stature (generally less than 60 cm [24 inches]), darker appearance of leaves and reproductive stems, larger seed size (2,200 seeds per g), and fewer seeds produced (McArthur and Stevens 2004; Meyer and others 1988b).

Geographic Distribution

Vegetation reconstructions based upon sediment core pollen records and woodrat midden macrofossil assemblages reveal that sagebrush-dominated ecosystems have been widespread during both the warm and cold phases characteristic of the Quaternary (last 2 million years). Studies suggest that during the late Pleistocene and early Holocene (40,000 to 10,000 years b.p.), sagebrush was widespread throughout most of its modern range (Mensing 2001; Nowak and others 1994; Rhode and Madsen 1995; Thompson 1990) and sagebrush-steppe ecosystems likely formed an ecotone with montane and continental tundra (Anderson and others 1999; Betancourt 1990; Fall and others 1995; Vierling 1998). Although the specific sagebrush taxa are unknown, a dominant sagebrush steppe association extended eastward into what is now the Central and Southern High Plains (Fredlund 1995; Hall and Valastro 1995) and into the desert valleys and plains of the southwest (Spaulding 1990; Van Devender 1990) during this same period. Sagebrush largely retreated from these eastward and southern range extensions during the hot dry conditions of the mid-Holocene (8,000 to 5,000 years b.p.) to an area similar to its modern distribution.

Today, big sagebrush is found throughout western North America from southern Canada to Baja California (McArthur 1999; McArthur and Plummer 1978). Beetle (1960) estimated that big sagebrush-dominated communities occupy approximately 586,306 km^2 (226,374 mi^2) in 11 western states. Although this is considered an overestimate (McArthur and Stevens 2004; Wright and others 1979), its widespread ecological dominance remains impressive. With an estimated area of dominance of 112,150 km^2 (43,301 mi^2) (Beetle 1960), black sagebrush communities occupy a greater total area than any other member of the *Tridentatae* except for big sagebrush and silver sagebrush (*A. cana*; McArthur and Stevens 2004).

Historical Conditions

A long standing controversy persists regarding the relationship between modern and pre-settlement distribution and condition of big sagebrush communities (Johnson 1986; Peterson 1995; Young and others 1979). One view holds that in response to livestock grazing practices and altered fire regimes, big sagebrush invaded large landscapes that were predominantly grasslands (Arno and Gruell 1983; Christensen and Johnson 1964; Cottam 1961; Cottam and Stewart 1940; Hull and Hull 1974, Stewart 1941). In this context, big sagebrush is considered an indicator, or even as an agent, of grassland degradation justifying eradication in the name of restoration (Blaisdell and others 1982; Britton and Ralphs 1979). This view became entrenched in early range ecology dogma (Cottam and Stewart 1940; Stewart 1941; Stoddard 1941) and has retained popularity to the present. The opposing view claims that, with the exception of lands converted to other uses, the distribution of big sagebrush landscapes is essentially unchanged from historic times (Hironaka 1979; Johnson 1986; Vale 1975; Welch 2005). This view admits to changes in shrub dominance in response to disturbance (for example, livestock grazing; Austin 2000), but denies the supposition that significant change in vegetation type has occurred. This view is supported by arguments that expansion rates for sagebrush are too slow to account for significant range advances in the suggested time frame of approximately 100 years (Welch 2005).

Early written accounts produced by trappers, explorers, immigrants, and settlers have been interpreted to support both mindsets. Young and others (1979) found support for grass-dominated systems in Stewart's (1941) review of historical records of Utah's rangelands. In his treatment of the same document, Welch (2005) argues that grassy areas were not characteristic. Vale (1975) found consistent references to expansive sagebrush-dominated landscapes from central Wyoming to western Nevada and central Oregon in early journal/diary descriptions of vegetation along major migration routes across the western United States. He concluded that grasslands were restricted to canyon and valley locations with favorable soil moisture conditions. His interpretations are strengthened by the decision to only use accounts of observations made before heavy use by immigrants and their livestock. Analysis of published series of photographs taken as early as the 1870s and recent retakes provide additional insight but fail to resolve the issue. Arno and Gruell (1983) and Kay (2003) provide photographic evidence and arguments in support of sagebrush invasion of grasslands in southwestern Montana and central Utah. In contrast, Johnson (1986), after examination of 1870s photos from Wyoming, northern Utah, and southeast Idaho, concluded that grasslands and shrublands have been quite stable for 115+ years in spite of a wide range of disturbances imposed during the interim. Interpretation of frozen-in-time descriptions or photos is confounded by a lack of knowledge regarding disturbance history and the relative proportions of the various seral stages on historic landscapes (Young and others 1979). Thus, a definitive answer to the debate may never be found due to lack of reliable information (Johnson 1986; Young and others 1979) and because of the likelihood that no single answer is correct across the full geography of big sagebrush. However, after a review of the available evidence and arguments, it is our opinion that, allowing for defensible exceptions, changes in the distribution of sagebrush-dominated landscapes over the last 150 years came primarily in the form of reductions rather than expansions. The principal causes for these reductions are land use conversion, woodland expansion, and more recently, increased fire frequencies associated with invasive annual grasses.

Ecological Distribution and Associated Species

Big and black sagebrush are largely restricted to semi-arid climate regimes where winter temperatures are cool to cold and winter-spring precipitation is sufficiently reliable to support spring growth (Beetle 1960; McArthur and Stevens 2004). Summer precipitation varies regionally, but soils at rooting depth are typically dry for much of the growing season. Soils on big sagebrush-dominated landscapes are moderately shallow to deep, well drained, and low in salt content (West 1979). Soil pH may vary from slightly acidic to moderately alkaline. Soils associated with black sagebrush tend to be drier and are generally more shallow or of higher percent rock than soils supporting big sagebrush (McArthur and Stevens 2004; Welch 2005).

Big and black sagebrush ecosystems form a wide, mostly continuous band across gradients in elevation in the southern Utah study area with lower limits defined by an ecotone with salt-desert shrublands and an upper boundary somewhat restricted by dense stands of montane or subalpine forest (fig. 2). Imbedded at mid-elevations within this sagebrush-grass matrix is a broad zone prone to recurrent invasion by species of pinyon pine and juniper (fig. 3) (Miller and others 1999; Tausch 1999; Tausch and Hood, this volume). This invasion belt is somewhat centered on the Wyoming-mountain big sagebrush transition zone (Goodrich and others 1999). These woodland species have been more or less permanent occupants on fire-protected topographical

Figure 2—Upper end of the sagebrush zone, with mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana*), shrubs, and grass intermixed with conifers and aspen (*Populus tremuloides*).

Figure 3—Utah juniper (*Juniperus osteosperma*) moving into a stand of big sagebrush (*Artemisia tridentata*).

units since their arrival in the Holocene (Tausch 1999; West and others 1998). For millennia, the rate and extent of expansion into the sagebrush matrix and frequency of retreat have been regulated by climate, topography, and fire regime (Tausch 1999; Tausch and Nowak 2000). The close association of sagebrush and woodland cover types in this zone led Tausch and Hood (this volume) to suggest that they are best considered as different phases of a single system (West and others 1978, 1998). They characterize a system of pinyon-juniper woodlands superimposed over a sagebrush-grass matrix with variation in the relative importance of the two types as an expression of variation in successional status across complex topographic and disturbance landscapes. This is clearly a refreshing and useful way to consider these dominant shrubland-woodland mosaics. For a detailed consideration of sagebrush shrubland-pinyon-juniper woodland successional patterns and management implications, see Tausch and Hood (this volume). Our focus here will primarily be sagebrush-grassland ecological processes that function independent of a woodland component.

Before settlement, Wyoming big sagebrush was the most abundant of the three subspecies of big sagebrush across its geographic range (West 1979). Within the southern Utah study area, it is a dominant on deep to moderately deep soils at elevations between 1,500 and 2,000 m (4,920 and 6,560 ft). Typical landforms include broad alluvial fans, low foothills, plateaus, and valleys receiving 170 to 350 mm (6.7 to 13.8 inches) annual precipitation (Goodrich and others 1999). At lower and upper ends of its range, it is bounded by, and intermixed with, elements of salt-desert shrub, black sagebrush, and mountain big sagebrush-pinyon-juniper communities (Howard 1999). The kind and abundance of sub-dominant shrubs and perennial grasses varies with soil attributes and disturbance history. Common shrub associates include species of ephedra (*Ephedra* spp.), rabbitbrush (*Chrysothamnus* spp.), snakeweed (*Gutierrezia* spp.), horsebrush (*Tetradymia* spp.), saltbush (*Atriplex* spp.), antelope bitterbrush (*Purshia tridentata*), and winterfat (*Ceratoides lanata*). Historically, perennial grasses dominated herbaceous understory. Common native species in the study area are Sandberg bluegrass (*Poa secunda*), bluebunch wheatgrass (*Pseudoroegneria spicata*), western wheatgrass (*Pascopyron smithii*), bottlebrush squirreltail (*Elymus elymoides*), Indian ricegrass (*Stipa hymenoides*), needle-and-thread (S*tipa comata*), and galleta grass (*Hilaria jamesii*). Perennial forb diversity and cover is relatively low on Wyoming big sagebrush sites (Bunting 1985).

Mountain big sagebrush in the study area is found on foothills and dry mountain slopes and ridges at elevations of 1,900 to 3,000 m (6,230 to 9,840 ft) in moderately deep loamy soils (fig. 4) (McArthur and Stevens 2004). Annual precipitation varies from 300 to 700 mm (11.8 to 27.6 inches) (Goodrich and others 1999). At higher elevations it occurs in forest openings of various sizes in association with quaking aspen (*Populus tremuloides*), Douglas-fir (*Pseudotsuga menziesii*), Englemann spruce (*Picea englemannii*), white fir (*Abies concolor*), and ponderosa pine (*Pinus ponderosa*) (Johnson 2000). At lower elevations, mountain big sagebrush dominates many treeless landscapes and co-exists in shrub — woodland mosaics with single-leaf pinyon (*Pinus monophyla*; Great Basin), two-needle pinyon (*P. edulis*; Colorado Plateau), Rocky Mountain juniper (*Juniperus scopulorum*), Utah Juniper (*J. osteosperma*), and Gambel oak (*Quercus gambelii*). The importance of co- and sub-dominant shrubs varies with topography, soils, and disturbance history. Common associates include mountain snowberry (*Symphoricarpus oreophilus*), common juniper (*J. communis*), currants (*Ribes* spp.), Oregon grape (*Mahonia repens*), mountain mahogany (*Cercocarpus* spp.), serviceberry (*Amelanchier* spp.), antelope bitterbrush, rabbitbrush, and green ephedra (*E. viridis*). Numerous species of perennial grasses and forbs combine to make a productive understory. Species composition and annual biomass production vary with site productivity potential and disturbance history.

Basin big sagebrush grows in deep, well-drained soils of plains, valleys, low foothills, and canyon bottoms at elevations of 1,500 to 2,100 m (4,920 to 6,890 ft) in the southern Utah study area (McArthur and Stevens 2004). Mean annual precipitation is approximately 300 mm (11.8 inches). It is most often found in association with species common to upper Wyoming big sagebrush and lower mountain big sagebrush although populations intermixed with salt tolerant black greasewood (*Sarcobatus vermiculatus*), shadscale (*Atriplex confertifolia*), and saltgrass (*Distichlis spicata*) also exist (McArthur and Stevens 2004).

Within the study area, black sagebrush is most abundant at elevations of 1,500 to 2,400 m (4,920 to 7,870 ft) (fig. 5) (McArthur and Stevens 2004) with an extended elevational range of 1,400 to 2,780 m (4,590 to 9,120 ft), thus the elevational range of black sagebrush is nearly equal to the combined range of the three sub species of big sagebrush (McMurray 1986). Black sagebrush segregation from big sagebrush is due to its ability to grow in shallow, rocky soils and mixing of the two species is generally limited to narrow ecotones. On lower elevations, it is common to find black sagebrush in nearly pure stands with only sparse herbaceous understory and associated shrubs. As with big sagebrush, the diversity and abundance of associated shrubs and understory species increases with elevation.

Figure 4—Dense stand of mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana*).

Figure 5—Black sage (*Artemisia nova*) with needle-and-thread grass (*Stipa comata*).

Fire Effects

Although extreme weather (Anderson and Inouye 2001; Nelson and others 1989; Nelson and Tiernan 1983), insects (Haws and others 1990; Nelson and others 1989), and disease (Nelson and others 1989, 1990; Sturges and Nelson 1986) apparently play a significant role in big sagebrush population dynamics, and do so at multiple spatial scales, fire is believed to be the dominant disturbance force in natural populations (Wright and Bailey 1982). Curiously, big sagebrush lacks morphological or physiological adaptations to survive fire or facilitate rapid recolonization (Welch and Criddle 2003). Plant stature is low to the ground and wood, bark, and foliage are highly flammable resulting in complete shoot mortality of burned plants (McArthur and Stevens 2004). Top-killed plants do not re-sprout from roots or crown (Blaisdell and others 1982; Britton and Ralphs 1979; Peterson 1995; Wright and others 1979). Seeds mature and disperse after the risk of fire has all but passed (Beetle 1960; Young and Evans 1989). Seeds have no mechanism for long-distance dispersal (Chambers 2000; Johnson and Payne 1968; Walton and others 1986; Wambolt and others 1999; Young and Evans 1989). The soil seed bank is ephemeral or absent (Beetle 1960; McDonough and Harniss 1974; Meyer 1990, 1994; Meyer and Monsen 1992; Young and Evans 1989). It appears paradoxical that the widespread landscape dominant, big sagebrush, is so poorly adapted to flames when fire is considered the "keystone disturbance" of western North American landscapes (Frost 1998; Keane and others 2002). Black sagebrush is no better adapted to fire than is big sagebrush, but dominates on sites less prone to burn. In contrast, many co-occurring shrubs have at least some ability to tolerate burning or to rapidly recolonize after fire (table 1). Fire adaptation by herbaceous species associated with big and black sagebrush varies, but is generally superior to that of these dominant shrubs (Britton and Ralphs 1979; Wright and Bailey 1982). Hence, a resolution to the apparent fire-big sagebrush paradox is not to be found solely in a species by species description of fire effects and adaptations, but in an examination of the manner in which fire is manifest on the landscape through time and space, also known as the fire regime.

Table 1—Fire adaptations for big sagebrush and co-occurring shrub species. Each species is rated on a scale of 0 to 4 for each area of adaptation where a 0 indicates no adaptation and a 4 indicates the species is highly adapted. See discussion and references in (Aleksoff 1999; Anderson 2004; Howard 1997, 1999, 2003; Johnson 2000; Marshall 1995; Tirmenstein 1999a, b; Welch and Criddle 2003; Zlatnik 1999).

Scientific name	Common name	Resprouting capability	Seed maturation timing	Seed dispersal distance	Seed bank
Artemisia tridentata	Big sagebrush	0	0	1	1
Chrysothamnus nauseosus	Rubber rabbitbrush	4	0	4	0
Purshia tridentata	Antelope bitterbrush	2	2	2	2
Amelanchier alnifolia	Saskatoon serviceberry	4	1	3	1
Cercocarpus montanus	True mountain mahogany	3	1	3	1
Atriplex canescens	Fourwing saltbush	2	0	1	3
Ephedra nevadensis	Nevada ephedra	3	3	1	2
Symphoricarpus oreophilus	Mountain snowberry	3	0	1	4

Big and Black Sagebrush Fire Regimes

Fire regime is quantified using various temporal and spatial parameters including frequency, seasonality, predictability, extent, and pattern (Morgan and others 2001). Fire regime is also expressed in terms of intensity, a measure of heat production per unit of time, and severity, a measure of fire-induced ecosystem change (Romme and others 2003; Ryan and Noste 1985). Fire regimes vary through time and across the landscape. Temporal variation is generally climate driven (Brown and others 2001; Grissino-Mayer and others 2004; Grissino-Mayer and Swetnam 2000; Heyerdahl and others 2002; Swetnam and Betancourt 1998), while spatial variation is primarily a product of topographic variation through its effects on species composition, productivity, desiccation rates, fuel continuity, and wind speed (Brown and others 2001; Heyerdahl and others 2001; Swetnam and Baisan 1996; Taylor and Skinner 1998). Fire characteristics vary locally in response to recent fire history and adjacency to fire prone landscape units (temporal and spatial autocorrelation; Morgan and others 2001).

Fire frequency is the most common measure of fire regime. It is an expression of the mean number of years between fire events, or mean fire interval (MFI), for a defined geographic unit. Estimates and interpretations of MFI are dependent upon spatial scale (Baker and Ehle 2001); the larger the area the shorter the interval in which no fire occurred. Hence, the most interpretable estimates of fire frequency are those associated with relatively small geographic units (Xiaojun and Baker 2006). Because populations of fire intervals frequently are not normally distributed, other measures of central tendency may be more appropriate for predicting fire free intervals (Grissino-Mayer 1999). However, differences among candidate statistics are generally not ecologically significant. Conversely, interval variability can be important and is often overlooked. For species that must regenerate from seed, such as big sagebrush, the length and frequency of short intervals is most important in determining the compatibility of the fire regime with species persistence (Crawford and others 2004). Conversely, the length and frequency of long intervals are also important for ecosystems prone to invasion by fire sensitive species. The susceptibility of many big sagebrush landscapes to invasion by pinyon pine, juniper, or other conifer species in the absence of fire illustrates this point (Heyerdahl and others 2006; Miller and others 1999; Tausch and Hood, this volume). Thus, big sagebrush-dominated ecosystems provide clear examples of how a fire-free window, defined by both short and long interval statistics, can be more useful in determining ecosystem structure than are estimates of central tendency alone.

Estimates of MFI for forested ecosystems are most often generated from two types of dendrochronological evidence. Years of low severity or surface fires are determined from tree ring series with datable fire scars (Arno and Sneck 1977). Once injured, fire scarred trees become more susceptible to injury from subsequent fires. Consequently, individual fire-recording trees may provide evidence of a large percentage of low severity fires that burned at one location for extended time periods. More complete fire chronologies are obtained by combining fire records from annually cross-dated trees growing in close proximity (Dieterich 1980). Fire dates from severe, stand-replacing fires can be estimated based upon synchronous patterns in stand establishment dates (Heyerdahl and others 2001).

These methods are difficult to apply to sagebrush ecosystems except where fire-recording trees grow in isolated pockets or forest-shrubland ecotones. Such conditions are limited primarily to the more mesic mountain big sagebrush sites. Houston (1973) used fire scarred Douglas-fir and lodgepole pine trees growing at the ecotone between forest and mountain big sagebrush-grass steppe in northern Yellowstone National Park to estimate MFI in sagebrush steppe for that area. After adjusting data to reflect only pre-1900 conditions, he calculated mean single-tree MFI values of 32 to 70 years for all study units and 44, 56, and 50 years for trees growing in the sample units most representative of the whole study area. He considered this an overestimate of true MFI so, using composite chronologies of questionable accuracy (not cross-dated), he estimated MFI values of 20 to 25 years. Using a similar approach, Arno and Gruell (1983) examined fire frequency at forest-mountain big sagebrush-grass steppe ecotones of southwest Montana. They calculated pre-1900 MFI values of 41, 45, and 74 years for moist, dry, and hot-dry habitat types. These investigators suggested that these estimates were likely overestimates of MFI due to possible missing fire evidence and adjusted their estimate of forest "grassland" ecotone MFI to 35 to 40 years. They concluded that sagebrush distribution and density has increased considerably in this region due to a reduction of fire frequency during the last 100 years. In a more recent study from the same region, Heyerdahl and others (2006) sampled fire scarred Douglas-fir trees from a 1,030 ha (2,544 acre) site topographically characterized as a mosaic of forest islands and mountain big sagebrush-grass elements. They estimated an average fire return interval of 37 years for the study period (1700 to 1860) with a range of fire-free intervals of 2 to 84 years. This is similar to the results observed by Arno and Gruell (1983). They also quantified the increase in tree distribution and density that occurred after 1860. Miller and Rose (1999) estimated MFI for mountain big sagebrush steppe in a south central Oregon study area using fire scars from four isolated clusters of ponderosa pine trees located in the mountain big sagebrush-grass matrix. Composite, pre-1900 MFI ranged from 12 to 15 years for three of the four clusters. Seven major fires (three or four clusters affected) occurred between 1650 and 1880, resulting in an approximate MFI for major fires of 38 years. Miller and Rose concluded that, "In the mountain big sagebrush community, mean fire intervals, prior to 1871, ranged from 12 to 15 years…" The estimates generated by these studies are cited extensively in the literature and provide the basis for a well-developed core conventional wisdom regarding sagebrush and fire.

Developing estimates of big sagebrush fire frequency directly from fire chronologies found on proxy species (trees) has the advantage of temporal precision (when properly dated) over long time periods. Spatial ambiguity is lessened by sampling from multiple locations on the landscape (Brown and others 2001; Heyerdahl and others 2001, 2006; Morgan and others 2001; Taylor and Skinner 1998). There are, however, disadvantages to this approach. Scarred trees are often scarce and distributed disproportionately across the landscape leading to spatial gaps in the record. Also, difficult to test assumptions must be made regarding historic fire regime continuity across the shrubland-forest ecotone. Perhaps the greatest problem with this approach rests in the fact that no suitable proxy species exist for the great majority of big sagebrush habitat types. Even if fire frequency estimates derived from the above cited studies prove to be accurate, there is considerable risk of inappropriate extrapolation of values to other localities.

Using an alternative approach, historic fire frequency for big or black sagebrush-dominated communities can be estimated indirectly based upon post-fire succession rates (Welch 2005; Welch and Criddle 2003). We suggest that the recovery pattern of big sagebrush to pre-burn conditions serves as an adequate index of post-fire succession for these plant communities. Several studies have attempted to quantify big sagebrush recovery time following both wild and prescribed fires. Harniss and Murray (1973) determined that big sagebrush on an upper Snake River Plain site in Idaho required at least 30 years to recover to pre-burn conditions and shorter fire-free intervals would lead to shrub dominance by species of horsebrush or rabbitbrush. Humphrey (1984) examined community composition in eight areas of southeastern Idaho where time-since-burn ranged between 2 and 36 years. His data indicated that big sagebrush (probably mountain big sagebrush) was still in a recovery phase 30+ years after burning. In Wyoming big sagebrush steppe in southwestern Montana, Watts and Wambolt (1996) observed that big sagebrush canopy cover reached 10 percent 30 years after burn treatment compared to the 13.5 percent for the unburned control. Wambolt and others (1999) observed similar delays in recovery for all three major subspecies after a wildfire burned sagebrush steppe communities north of Yellowstone National Park. Stand density for Wyoming, mountain, and basin big sagebrush was 2, 12, and 16 percent respectively, of unburned reference areas 19 years after the fire. In a southwestern Montana study of 13 spring and fall prescribed burn sites (2 to 32 years post-burn), mountain and Wyoming big sagebrush canopy cover or stand density was significantly less for burned areas than for unburned areas in 36 of 40 comparisons (Wambolt and others 2001). The authors suggested that 30 years might be inadequate for full recovery in many cases. We further anticipate that recovery periods will often be longer after the more intense, mid-summer wildfires. Twenty years after a wildfire burned a central Utah site, West and Yorks (2002) found that Wyoming big sagebrush recovery had barely started based on the low density of sagebrush plants. They concluded that recovery rates for "sagebrush semi-desert" communities are much slower than they are for sagebrush steppe communities. These studies suggest that big sagebrush requires from 20 to 35+ years for post-fire stand recovery under favorable conditions and much longer intervals when conditions dictate a slower pace of recovery. Correspondingly, longer intervals are expected on xeric sites where fine fuel production under average weather conditions is inadequate to carry fire except under severe conditions.

There is little information regarding historic fire regimes for black sagebrush-dominated landscapes. It is generally believed that fire was rare on these landscapes due to insufficient fine fuels to carry fire (Wright and others 1979). Miller and Rose (1999) found evidence of just two fires in 300 years of record for a low sagebrush (*Artemisia arbuscula*)-western juniper (*Juniperus occidentalis*) community. Low sagebrush is similar to black sagebrush and like black sagebrush, occupies low productivity sites (McArthur and Stevens 2004). Although this information is insufficient to estimate MFI, it is sufficient to support the notion of long fire-free intervals for sagebrush communities with low fine fuel production.

In cases where a significant number of big sagebrush seedlings establish from surviving residual seeds, stand recovery is relatively rapid (Bunting 1985), and the size, pattern, and continuity of the burn has little impact on recovery time. This phenomenon is most common when fires burn mature stands of mountain big sagebrush, but has also been observed with basin and Wyoming big sagebrush on mesic sites (Wambolt and others 1999, 2001). Frequently, big sagebrush seedlings do not establish within 1 year post-burn, either because of a lack of viable seeds on the landscape, or because of the failure of seeds to produce viable plants (Welch and Criddle 2003). When this occurs, big sagebrush recovery is dependent upon seed dispersal from unburned source areas and favorable weather patterns. Fire size, pattern, and continuity directly impact the distance that seeds must disperse, and thus have considerable impact on recovery time (Crawford and others 2004). Welch and Criddle (2003) provide an example that illustrates this effect. They measured the down wind (west to east) advance of mountain big sagebrush 14 years post-burn on a southern Idaho site and determined a mean annual spread rate of 13 m (43 ft). Northward spread was half that of the eastward spread and big sagebrush encroachment from the north and east burn margins was negligible. Based on these data, they estimated that it would take 71 years for big sagebrush to reoccupy this 146+ ha (360+ acre) burn. In effect, large continuous fires without unburned islands result in long seed dispersal distances that translate into long recovery periods while recovery from small discontinuous fires with short seed dispersal distances is more rapid.

Although big sagebrush post-fire recovery time varies situationally, a general relationship between recovery time and minimum (significantly shorter than the mean) fire-free intervals should be expected. Therefore, estimates of MFI (or any other measure of central tendency) must be substantially longer than estimates of mean recovery time in order to capture the full variability in fire interval duration. Conversely, we recognize that on landscapes prone to conifer invasion, lengthy intervals would result in shrubland displacement by woodlands (Tausch and Hood, this volume). Taken together, these assumptions provide the theoretical basis for our estimates of MFI on sagebrush-dominated landscapes. We suggest that historic MFI values ranged from 40 to 80 years for mountain big sagebrush and some productive basin and Wyoming big sagebrush communities and were as long as 100 to 200 years or longer for big and black sagebrush sites with low productivity. We offer broad estimates here in order to capture the range in MFI length we believe existed across the full ecological and geographical distribution of big sagebrush. A range of intervals lengths must be expected in conjunction with a single MFI value. For example, natural variability corresponding to a MFI of 50 years might produce intervals as short as 10 to 15 years and as long as 100 to 120 years; however, most intervals would likely fall between 25 and 75 years. Our estimates are similar to pre-1900 MFI values calculated from tree-ring records at forest-shrubland ecotones by Heyerdahl and others (37 years; 2006), Houston (32 to 70 years; 1973), and Arno and Gruell (41 to 74 years; 1983) before data adjustment. Although the estimate for historic MFI of 12 to 15 years proffered by Miller and Rose (1999) is unrealistic, the approximate MFI of 38 years derived from their data for widespread landscape fires only approaches our estimate and is likely a more realistic application of their data to the mountain big sagebrush matrix of their study area. In their North American synthesis of fire ecology, Wright and Bailey (1982) estimate MFI for mountain and Wyoming big sagebrush at 50 and 100 years – numbers that are in general agreement with what we suggest here.

So, what of the big sagebrush-fire paradox? It seems that in contrast to strategies employed by co-occurring shrubs (table 1), big sagebrush solved the fire problem by producing highly competitive, yet disposable plants. It does not invest resources in morphological or physiological adaptations to fire, as it never had to in its short evolutionary past. This was particularly true for the 2+ million years of the Pleistocene, during which time cooler climatic conditions would have rarely favored fire to the extent they do today. Sagebrush thrives on suitable landscapes as long as the fire-free intervals are sufficiently long to permit re-establishment of mature stands, and short enough to prevent displacement by forest or woodland (Miller and Tausch 2001). This leads us to the conclusion that past variation in big sagebrush distribution and dominance was to some degree influenced by climatically-driven changes in fire regime parameters. Where changes were large and persistent, fire regime-driven shifts among shrub-steppe, grassland, and woodland or forest ecosystems must have occurred. Consequently, caution is warranted when making comparisons of big sagebrush distribution or dominance from time periods experiencing distinct climatic patterns. Judgments made when comparing pre-settlement big sagebrush conditions corresponding to the end of the "Little Ice Age" with contemporary environmental conditions should be tempered by the context of the corresponding change in climate.

Variation in historic fire regimes might also be attributed to variation in human-caused ignitions. Fire was the most important tool available to aboriginal inhabitants for manipulating the natural environment (Griffin 2002; Williams 2004). It was used to promote the growth of desirable resource plants, enhance habitat for important animals, and drive game and insects (Griffin 2002). It might also have been important for warfare, clearing travel corridors, and providing fire-safe camp sites (Williams 2004). Anecdotal and ethnographic accounts describing the use of fire within the big sagebrush domain are reviewed by Baker (2002), Griffin (2002), and Whitlock and Knox (2002). These authors conclude that, in spite of known patterns of fire use, it is difficult to find available evidence sufficient to attribute landscape-scale variation in pre-settlement fire regimes to human fire practices. Thus, we are left to consider the ecological consequences of historic fire regime variability with little ability to ascertain the human role in that variation.

Finally, we pose the question, "What does big sagebrush-grass climax look like in the absence of woodland invasion and fire?" Current theory suggests that, with time, competition from big sagebrush will eventually reduce perennial grasses and forbs to scattered remnants (Miller and Tausch 2001). Indeed, one does not need to look far when in sagebrush country to find dense stands of sagebrush with depleted herbaceous understories. However, the question can not be properly addressed without also considering the impact of domestic livestock on the competitive relationship between shrubs and perennial grasses (Austin 2000). Unfortunately, there are few reference areas not impacted, either historically or currently, by livestock grazing. The question is thus compounded by the additional variable and becomes, "What does big sagebrush-grass climax look like in the absence of fire, woodland invasion, and livestock grazing?" Although somewhat short term in nature, results from a 45-year study (Anderson and Inouye 2001) found that after livestock removal, a dynamic equilibrium was reached between shrubs and perennial grasses where both were well represented in the plant community. Other studies show significant, sustained increases in percent grass cover concurrent with increases in big sagebrush cover following livestock reduction or removal (Branson and Miller 1981; McLean and Tisdale 1972; Pearson 1965). We suggest that such a dynamic balance should be expected between big sagebrush and grasses in a variety of settings, although the actual nature of the balance is likely to differ substantially from place to place. This is not to suggest that fire is not a fundamental ecological process in sagebrush grass communities, nor that fire should not be used as a management tool. Rather, we suggest that losses of perennial grasses in big sagebrush-dominated communities may have more to do with the effects of intense, selective grazing than with the periodicity of fire.

Euro-American Settlement

Ecological processes associated with sagebrush/grass ecosystems in the southern Utah study area began to be altered soon after Euro-American settlement in the mid-1800s (Young and others 1979). First settlers were primarily small groups of Mormons sent by their leader, Brigham Young, to establish organized communities wherever water, timber, and forage conditions permitted. Numerous agricultural-based communities were started where mountain streams met valley floors. Early settlers soon learned that the deep, loamy soils associated with tall (basin) big sagebrush were well suited for irrigated crops and significant acreages of this type were converted to cropland (Kearney and others 1914).

Originally, settlers pastured livestock anywhere good grass could be found close to their communities. Although few settlers owned many animals individually, livestock were commonly united in community herds (Young and others 1979). Cool-season grasses growing in association with big sagebrush at valley and foothill locations were used heavily and sometimes year-round when located at close proximity. The area impacted by grazing livestock gradually expanded as the local forage base depleted, herd numbers increased, and concern for potential losses to opportunistic native peoples decreased.

Livestock numbers, especially sheep, grew rapidly in the late 1870s and 1880s (Keck 1972; Murdock and Welsh 1971) as livestock production shifted from a subsistence to a market economy. In addition to expanding local herds, large numbers of sheep were trailed and eventually freighted by train to and from the area by parties with little or no interest in the grazing needs of local communities. Sheep herds often spent several months trailing between mountains in fall and summer and foothills and deserts in winter and spring. By 1890, livestock numbers far exceeded realistic estimates of carrying capacity and the degradation of plant/soil environments was widespread. Over-grazing persisted for several decades resulting in widespread degradation of big and black sagebrush landscapes similar to that imposed on other plant community types.

Current Conditions

Ungulate Impacts

To some degree, pasturing of domestic livestock on big and black sagebrush-dominated landscapes affected, and continues to affect, community structure and ecological processes at all grazing intensities. Minimum effects include the removal of fine fuels and subsequent reduction in capacity to carry fire. At higher grazing intensities, palatable herbaceous species were weakened or eliminated. The ecological changes caused by these alterations occurred rapidly on productive landscapes characterized historically by relatively short fire-free intervals and a propensity for invasion by woodland or forest conifers. In response to a weakened, and often depleted herbaceous understory, big sagebrush density and canopy cover increased and the pace of woodland invasion was accelerated (Miller and Tausch 2001; Tausch 1999). The combination of reduced fire and accelerated woodland invasion resulted in loss of landscape level heterogeneity and a major shift in the sagebrush-woodland complex to increasingly more widespread and dense woodland dominance (Tausch and Hood, this volume). Similar degradation has occurred on drier landscapes, although woodland expansion is either absent or occurring at a slower pace.

Exotic Weed Impacts

Immigrating Europeans intentionally and accidentally brought the seeds of numerous plant species to North America. Some of these species are invasive in big sagebrush habitats. The winter annual, cheatgrass (*Bromus tectorum*), is particularly well adapted for big sagebrush habitats and has had considerable success invading the weakened and often depleted understories of basin and Wyoming big sagebrush, mid elevation black sagebrush, and drier mountain big sagebrush communities (Young 1994). During years with good spring moisture, cheatgrass produces large quantities of continuous fine fuels that cure earlier in the season than do native perennials and effectively lengthen the fire season. Where established, cheatgrass has resulted in shorter fire-free intervals, earlier fires, and larger, more continuous fires than existed historically (Peters and Bunting 1994; Whisenant 1990). Once established, cheatgrass out-competes seedlings of native perennials disrupting natural regeneration processes (Billings 1994; Young 1994; Young and Evans 1978). Where the perennial herbaceous understory is depleted, the cheatgrass-fire cycle eventually reduces what is left of sagebrush-dominated communities to cheatgrass-dominated annual grasslands (Pellant 1990; Young and Evans 1978).

Fire Regime Condition Classes

The classification of existing vegetative communities based upon the degree of departure from historic conditions and the risk of loss of one or more defining components provides a framework to guide restoration and management efforts. Three broad classes reflect increasingly greater departure from historic conditions implicating parallel increases in intervention needs.

Fire Regime Condition Class 1 (FRCC 1)

Herbaceous species diversity and relative cover for healthy (FRCC 1) big or black sagebrush-grass communities in the southern Utah study area vary across gradients in site productivity. In general, diversity and relative cover decrease with increasing aridity. The extreme of this condition can be observed in some low elevation (xeric) black sagebrush communities that essentially function as stable shrublands with extensive bare interspaces and few scattered herbs (McArthur and Stevens 2004). Conversely, mesic stands of mountain big sagebrush have a diverse assemblage of perennial grasses and forbs that resist weed invasion (Anderson and Inouye 2001; West and Yorks 2002). Depending on disturbance history, percent cover of mountain big sagebrush varies from <5 percent, 1 to 20 years post burning, to 20 to 40 percent at shrub/grass equilibrium 30 to 70 years post burn (Harniss and Murray 1973; Humphrey 1984; Wambolt and others 2001; Welch 2005; Welch and Criddle 2003). Mature sagebrush stands are generally multi-aged (Perryman and others 2001). On the landscape scale, multiple seral stages are represented in an ever-shifting mosaic reflecting periodic reoccurrence of fire or other disturbances (Crawford and others 2004). Although landscape-level complexity may appear to decrease with the longer fire-free intervals expected for basin and Wyoming big sagebrush sites, microscale compositional variation may if fact increase as the plant community has time to fine tune responses to small variations in the physical environment (Anderson and Inouye 2001). Typically, sagebrush cover for basin and Wyoming big sagebrush-dominated communities ranges from 15 to 35 percent (Welch and Criddle 2003). If present in FRCC 1, cheatgrass is scattered and forms an insignificant portion of the herbaceous biomass. At mid to lower elevations, cryptobiotic crusts may be present on bare inter-shrub openings. Woodland species are widely scattered, if present. Fires are generally patchy. If grazed by livestock, light to moderate stocking rates and periodic rest during the growing season are needed to maintain the herbaceous component of this condition class.

Fire Regime Condition Class 2 (FRCC 2)

FRCC 2 for sagebrush-grass communities occurs with and without elements of woodland invasion. The herbaceous perennial component in this condition class is moderately depleted in abundance or diversity relative to FRCC 1. A shift in species composition reflecting palatability is often noticeable. The primary causes are chronic overstocking of livestock and periodic abusive grazing practices. Sagebrush cover often exceeds that of FRCC 1 due to competitive release by a weakened herbaceous understory. On drier sites, cheatgrass is generally present (some soil types excepted), but does not dominate except in scattered patches. Current fire-free intervals are often much longer than estimated historic intervals. However, the risks of weed invasion for this condition class are greater than the risks of woodland conversion. The probability of conversion (transition) from a shrub-grass community to weed-dominated grassland is moderate.

Tausch and Hood (this volume) give descriptions of big and black sagebrush landscapes susceptible to invasion by woodland trees. In FRCC 2, trees have reached 25 to 50 percent of their potential cover for the site and shrub, and herbaceous cover has been reduced by up to 75 percent. Total conversion to tree dominance (FRCC 3) may occur in 40 to 50 years on moderately productive sites (Tausch and Hood, this volume). Because cheatgrass invasion frequently occurs synchronously with woodland expansion, fire compounds risks. The risk of cheatgrass expansion with fire counters that of conversion to woodland (loss of shrubs and herbs) without fire. Alternative non-fire treatments or combinations of treatments are needed for sagebrush-grass restoration.

Fire Regime Condition Class 3 (FRCC 3)

Characteristically, perennial herbs are severely depleted in FRCC 3 big sagebrush communities of the study area. Weedy annuals, especially cheatgrass, dominate the herbaceous understory. Big sagebrush plants tend to be old-aged with little new recruitment. Shrub cover may be variable. There is a high probability that a single fire event will result in conversion to an annual-dominated community with corresponding short fire-free intervals and large fire size. Indeed, outside of the study area, extensive areas of big sagebrush plant communities have already experienced this type conversion (Billings 1994; Miller and others 1999; Whisenant 1990; Young and Evans 1978). The current fire-free interval may be much longer than estimated historic intervals; however, the greater risks for this condition class are associated with shortened fire-free intervals due to the loss of perennial herbs and probability of invasion by weeds such as cheatgrass. Where susceptible to woodland invasion, trees may have reached densities sufficient to fully occupy the site (Tausch and Hood, this volume). Shrubs and perennial herbs are weak and scattered in tree openings. Litter accumulates under trees and extensive bare soil may be exposed and eroding between trees. Long-term overgrazing of stable black sagebrush communities by domestic sheep can result in conversion to a FRCC 3 characterized by nearly solid stands of low rabbitbrush (*Chrysothamnus greenii*) or broom snakeweed (*Gutierrezia sarothrae*).

Fire Regime Condition Class Assessment

We assessed condition class for big and black sagebrush sites in the southern Utah study area using data and photographs obtained from 148 sites between 1997 and 1999 (Davis and others 2004). Sites with boundaries roughly corresponding to the study area were selected within Utah Division of Wildlife Resources Wildlife Management Units 16, 19 through 25, and 27 through 30. Study sites in which neither big nor black sagebrush were listed as descriptive, that appeared to have been recently treated (chaining or seeding), or were protected from grazing were excluded. A second assessment was made for 117 of the sites from photos and data collected 5 years after the first field visit. Data from four additional sites were added for this later period. The number of sites per sagebrush taxon was 59, 15, and 57 for mountain, basin, and Wyoming big sagebrush, respectively, and 21 for black sagebrush. These data are the most comprehensive available and provide a sound basis for estimating big and black sagebrush condition class for the study area.

Four photographs per site/assessment period were used to make a qualitative assessment of condition class. An assessment of woodland encroachment (primarily juniper) was also made from photographs. As a quantitative indicator of condition for each site, we used the Desirable Components Index (DCI) developed by Davis and others (2004). The DCI is computed on a scale of 0 to 100 and is based on cover percentages for shrubs, perennial grasses, perennial forbs, and annual grasses and the presence of noxious weeds. Shrub values are adjusted based upon size class (seedlings, young, and mature) and vigor (normal, decadent, and dead) distributions. Optimal values for the DCI are obtained by 20+ percent shrub cover, 15+ percent perennial grass cover, 5+ percent perennial forb cover, no annual grasses or noxious weeds, shrub decadence less than 20 percent, and percent young shrubs greater than 10 percent. For more details on DCI computation see Davis and others (2004). We derived condition class estimates from site DCI scores using two scales that correspond to mesic and xeric habitats (table 2).

Our qualitative and quantitative estimates of site condition class are in general agreement (table 3). Averaged across all sites, condition class ratings associated with the late 1990s assessment are somewhat higher than those for the 2002 to 2004 assessment. This difference reflects the severity of a regional drought from 1999 to 2004. Shrub and perennial grass mortality was particularly striking in 2003 at low elevations in the southern and eastern portions of the study area. These data suggest that from 10 to 20 percent of big sagebrush landscapes and 30 to 50 percent of black sagebrush landscapes in the study are currently in FRCC 1. Approximately 40 to 60 percent of the four sagebrush taxa are in FRCC 2. Relative area in FRCC 3 is highest for basin and Wyoming big sagebrush sites and lowest for mountain big sagebrush and black sagebrush. Moderate to advanced encroachment by juniper was not different for the three big sagebrush subspecies (40, 40, and 38 percent for mountain, basin, and Wyoming sites, respectively). We rated encroachment as moderate to advanced for 29 percent of black sagebrush sites. These estimates of woodland encroachment may be somewhat low due to a possible bias against woodland-dominated sites in the site selection criteria.

Table 2—Scales for deriving condition class from Desirable Components Index (DCI) scores (Davis and others 2004).

Scale	Condition class	DCI score
Scale 1 – mountain big sagebrush and upper elevation black sagebrush	1	70+
	2	45-69
	3	<45
Scale 2 – basin and Wyoming big sagebrush and lower elevation black sagebrush	1	55+
	2	30-54
	3	<30

Table 3—Big and black sagebrush condition class estimates for 152 sites in the southern Utah study area based upon Desirable Components Index (DCI) and qualitative assessment of photographs for each site (Davis and others 2004). Percentages are based upon 148 and 121 sites for the 1997 to 1999 and 2002 to 2004 assessments, respectively.

		Condition class estimate			
Sagebrush taxon	**Condition class**	**1997 to 1999 DCI**	**Assessment photos**	**2002 to 2004 DCI**	**Assessment photos**
		Percent of sites			
Mountain big sagebrush	1	26	31	13	9
	2	64	64	58	74
	3	10	5	29	17
Basin big sagebrush	1	20	20	13	19
	2	53	40	47	37
	3	27	40	40	44
Wyoming big sagebrush	1	39	28	15	5
	2	47	53	48	65
	3	14	19	37	30
Black sagebrush	1	52	43	50	33
	2	39	57	36	58
	3	9	0	14	8
All taxa combined	1	34	31	18	11
	2	53	56	50	64
	3	13	13	32	25

Recommended Treatments

A variety of treatments has been developed to modify and restore big sagebrush communities. The appropriateness of each is dependent upon site condition class, existing uses, available resources, and management goals (Monsen 2004). The underlying principle is to repair structure and ecological processes in existing vegetative communities (Whisenant 1999). Treatment objectives for FRCC 1 and many FRCC 2 sites are maintenance in nature and include: reduction of big sagebrush density or cover; removal of young pinyon or juniper trees; increase density, productivity, or diversity of perennial herbaceous understory; increase productivity of associated shrub species; and create spatial heterogeneity among seral stages across the landscape. Potential treatments that are effective in achieving those objectives include prescribed fire; selective herbicide application; and low-impact mechanical treatments such as anchor chaining and railing. Objectives for treating lower end FRCC 2 and FRCC 3 sites are clearly remediation focused and include soil stabilization, water capture and retention, and reconstruction of resilient shrub-perennial grass communities. Practices employed to achieve these objectives include removal of woodland trees, control of invasives (usually annuals but may include perennials), and restoration plantings of perennial herb and shrub elements. Successful restoration of big sagebrush-grass communities from FRCC 3 to FRCC 1 or 2 is generally an expensive multi-step process requiring combinations of treatments (Lancaster and others 1987) and fortuitous timing and quantities of precipitation after planting. Pre-emergent herbicides and tillage treatments have proven to be at least moderately effective in controlling invasive annuals. Bio-control methods for controlling cheatgrass have been investigated (Kennedy 1994; Meyer and others 2001); however, to date, are undeveloped or unproven. Prescribed fire and high impact mechanical treatments are employed to remove woodland trees. Restoration plantings require proper seedbed preparation and timely planting of appropriate seed mixes of adapted, compatible ecotypes (Monsen and Stevens 2004). Appropriate long-term management practices, including changes in livestock use, are essential after treatment. In the following, we discuss the advantages and limitations of these treatment options.

Prescribed Fire

Prescribed fire is an efficient, cost effective method for removing big sagebrush and woodland trees on portions of the landscape (Britton and Ralphs 1979; Bunting and others 1987; Wright and others 1979). Because it mimics natural fire, properly timed prescribed fire is supportive of natural ecological processes of nutrient cycling and plant succession. As previously discussed, time needed for big sagebrush recovery after burning can vary greatly and depends upon community composition before the burn, fire intensity (linked to season of burn), fire size and pattern, and weather conditions after the burn (Bunting and others 1987). Non-target species can be damaged, and increased herbaceous production is not always realized after burning (Britton and Ralphs 1979; Bunting and others 1987; Welch 2005; Wright and Bailey 1982; Wright and others 1979). Prescribed fire, or wildland fire use, should be limited to stands where perennial grasses and forbs are sufficiently abundant to preclude the risk of expansion by cheatgrass or other fire tolerant invasives. Typical sites are mountain big sagebrush communities in FRCC 1 and 2. Restoration fires should be small or patchy, facilitating the perpetuation of a mosaic of seral stages and minimizing seed dispersal distances for recovering big sagebrush. Large patches of mature sagebrush should be left unburned as critical wildlife habitat. Based on a 50 year MFI, mean area burned per year (natural and wildfires combined) for mountain big sagebrush should not exceed 2 percent. Prescribed fire should not be considered for dry basin and Wyoming big sagebrush and black sagebrush stands in the study area due to slow recovery time and the high risk of conversion to weeds. Deferral from livestock grazing for 1 or more years before burning may be necessary to allow for fine fuel accumulation and curing (Whisenant 2004; Wright and others 1979). A post-treatment rest from grazing of one to two growing seasons (Bunting and others 1987; Whisenant 2004; Wright and others 1979) should be considered a minimum requirement that is not always adequate. Burning restrictions often result in narrow windows of opportunity for treatment and may require rapid mobilization. Wright and Bailey (1982), Bunting and others (1987), and Whisenant (2004) outline guidelines for prescribed burning of big sagebrush communities.

Herbicides

Herbicide treatments are used effectively as substitutes for fire to reduce big sagebrush cover. The volume of literature dedicated to the development and testing of various compounds for this purpose reveals the level of interest that existed during the mid 1900s in finding novel ways to control or eradicate this species (Crawford and others 2004; Welch 2005 and references therein). Herbicide selectivity and effectiveness varies with concentration, season of use, soil characteristics, and community composition (Vallentine 2004; Welch 2005). Here we discuss the use of the two herbicides most frequently used to control big sagebrush. Early work focused on the use of 2,4-D [(2,4-D-dichlorophenoxy) acetic acid] a synthetic auxin, or plant growth regulator (Welch 2005). Although effective in controlling big sagebrush short-term, long-term effects on the plant community were difficult to predict (Watts and Wambolt 1996; Welch 2005). This may be due to its effects on non-target species, especially broadleaf forbs. Consequently, 2,4-D is no longer the herbicide of choice for big sagebrush reduction (Crawford and others 2004). Tebuthiuron ([N-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2-yl]-N,N'-dimethylurea]; Spike®), a photosynthesis inhibitor, is applied to the soil where it moves into the rooting zone with water and remains active for several years (McDaniel and others 2005; Wachocki and others 2001). It is absorbed through the roots and is functionally selective against big sagebrush at low application rates (Baxter 1998; Crawford and others 2004; McDaniel and others 2005; Wachocki and others 2001). Post-treatment increases in productivity for herbaceous species can be substantial (Baxter 1998; McDaniel and others 2005; Olson and Whitson 2002). Herbicide treatments, particularly the use of tebuthiuron, pose certain advantages over prescribed fire. Spatial precision of treatment application is greater with herbicide application than with prescribed fire. Longer windows of opportunity for treatment are available, especially for tebuthiuron (Baxter 1998; Marion and others 1986), than for prescribed fire. Damage to non-target species is often less with tebuthiuron than with prescribed fire (Baxter 1998; McDaniel and others 2005). The level of "thinning" and associated treatment longevity are effectively regulated by altering application rates (Crawford and others 2004; McDaniel and others 2005; Olson and Whitson 2002; Wachocki and others 2001), although these must be calibrated for soil texture and precipitation (Baxter 1998). There are disadvantages to using tebuthiuron relative to prescribed fire including greater per-acre cost and ineffectiveness in controlling woodland trees. Tebuthiuron has greatest application where there is a need to reduce sagebrush density or cover to allow existing herbaceous understory to respond to the competitive release. It is particularly valuable where prescribed fire is not practical or where complete removal of big sagebrush is undesirable. Tebuthiuron may have application with FRCC 2 basin and Wyoming big sagebrush communities where the ecological risks of prescribed fire would be excessive.

A second class of herbicides is used to control invasive annuals, such as cheatgrass, in big sagebrush communities. Broad spectrum contact herbicides are effective, but collateral damage to non-target species can complicate their use. Soil active, pre- or post-emergent herbicides have proven effective in killing annual weeds in early post-germination stages, thus effectively depleting the seed bank and releasing residual perennials from weedy competition. Applications of sulfometuron methyl (Oust®) have proven effective in providing a 1- to 2-year window of greatly reduced competition from cheatgrass (Pellant and others 1999). Questions remain regarding residual time in the soil and impacts on established perennials. In recent years, considerable interest has been generated for imazapic (Plateau®) as a soil-active herbicide. Early results suggest that this herbicide is quite selective in its effects and that it is particularly effective with annual bromes (Bekedam and Pyke 2004; Porath and others 2003; Smith and Anderson 2003; Whitson 2003). This kind of treatment has application on FRCC 2 sites to encourage release of weakened herbaceous plants and on FRCC 3 sites to deplete the weed seed bank in preparation for restoration plantings.

Mechanical Treatments

A wide variety of mechanical techniques has been devised to eliminate invading pinyon and juniper trees from sagebrush grass communities and to reduce cover and density of big sagebrush. These treatments are generally used as precursors or sometimes simultaneously with restoration/reclamation plantings (Monsen 2004; Stevens 1999). Treatment effectiveness and management considerations for the mechanical control of woodland trees are discussed in Stevens and Monsen (2004a) and Tausch and Hood (this volume). Parker (1979), Mattise and Scholten (1994), Welch (2005), and Wiedemann (2005) provide brief but adequate descriptions of major equipment developments and their applications. Various plow and disk type implements kill most big sagebrush plants, as well as associated, species necessitating follow-up plantings of adapted species. Mature woodland trees and a high percentage of non-sprouting shrubs can be removed by dragging a long section of anchor chain between two crawler tractors. Actual treatment outcome is affected by link weight and modifications, chain length, relative tractor positions, treatment passes, and tractor speed (Stevens 1999). Low-impact treatments leave a majority of herbaceous species intact. A second pass of the chain (two-way chaining) improves juniper kill and is reasonably effective in burying seeds broadcast after the first pass as part of restoration plantings (Stevens 1999). Chaining is a preferred technique on rough terrain up to 20 to 30 percent grade. The disk chain is an implement that combines design features of the anchor chain and disk implements. The railer and pipe harrow are implements that are dragged behind tractors. They are designed to remove mature sagebrush and leave some smaller plants intact. Damage to herbaceous species is minimal; however, the pipe harrow creates enough soil disturbance to facilitate seed burial and establishment of desired species (Welch 2005). Equipment and practices should be selected to minimize risks to soil erosion. Archeological surveys are required prior to mechanical treatment in order to avoid cultural site disturbance. These treatments are generally used for FRCC 2 and 3.

Restoration Plantings

The concept of restoration planting can be defined in either broad or narrow terms. Narrowly, a restoration planting is seen as an attempt to re-establish a native plant community that is indistinguishable, or nearly so, in composition, structure, and ecological process from what is perceived as the natural state. This view dictates a careful selection of source germplasms for plant propagules, usually seeds, which are consistent with the goal. Although a worthy target to aim for, actualization of this kind of restoration is generally difficult to achieve for big or black sagebrush communities. Alternatively, a broader view of restoration plantings includes all attempts to establish complimentary assemblages of plant species that structurally and functionally resemble pre-disturbance conditions in so far as the level of site degradation will allow. Developed cultivars of native and introduced species may be planted in various combinations deemed most likely to achieve goals of site stabilization and other management objectives. Intermediate approaches with varying restrictions on plant material origin are common for big sagebrush-grass plantings (Roundy and others 1997; Stevens and Monsen 2004b). Although restoration practices vary, most are designed to either facilitate natural repair processes or supplant them (Whisenant 1999). Examples of facilitative actions associated with the repair of big sagebrush-grass communities might include protection of residual big sagebrush islands to allow natural seed dispersal into surrounding treated or disturbed landscapes (Longland and Bateman 2002) and delaying livestock grazing until after perennial grass seed shatter to allow maximum seed production and dispersal on soil surfaces. Although facilitative restoration can be a slow process, costs are relatively low, allowing treatment of large areas.

Most efforts at developing restoration methodologies for big sagebrush-grass communities have taken a more direct, essentially agronomic approach. Over time, equipment and methods developed for the efficient establishment of crop monocultures on uniform, submissive environments were adapted and modified for planting a wide variety of seed types on highly variable and sometimes harsh environments (Keller 1979; Monsen and Stevens 2004; Young and Evans 1987). The greatest innovations have been associated with the collection, cleaning, and planting of native shrubs and forbs (Jorgensen and Stevens 2004). General principles and guidelines for big and black sagebrush restoration plantings that have stood the test of time and experience are discussed by Monsen and Stevens (2004) and Stevens and Monsen (2004b). First, competition from weedy species must be controlled. The seedbed should be firm, but not overly compacted. Effects of litter or mulching vary by species. Larger seeds should be sown at a depth of 1 to 2 cm (0.4 to 0.8 inches). Seeds of small seeded species, such as sagebrush, must be placed at or near the soil surface. It is generally best to plant seeds of slower growing forbs and shrubs separate from those of grasses. The optimal time for planting is from late fall to early winter, allowing for maximum use of winter and spring soil moisture and removal of seed dormancy. For seed collected from wild populations, climate and soils of the collection site should match those of the treatment site. Published seed transfer zones similar to those produced for trees species have been developed for a few key species (Mahalovich and McArthur 2004). More are needed. Although much progress has been made in recent decades, seeds of many species desirable for restoration plantings are either not available, or are available only in small quantities (McArthur and others 1987; McArthur and Young 1999; Roundy and others 1997). Too little is known of the biology of many of these species to plan for their efficient use. Additional research is needed to ascertain relationships between soil water and temperature and seed germination and seedling growth (Roundy 1994). The potential effects of present and future changes in biological and physical environments on community stability are not well understood and are in need of thoughtful attention. Finally, a commitment must be forged to manage preserved and restored big sagebrush landscapes for long-term sustainability or the degradation – restoration cycle will become a permanent feature of the big sagebrush landscape.

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Chapter 6

Creosotebush, Blackbrush, and Interior Chaparral Shrublands

Matthew L. Brooks, Todd C. Esque, and Tim Duck

Introduction

The vegetation, fire regime, and Fire Regime Condition Class descriptions in this paper apply broadly to the Mojave Desert, Colorado Plateau, and southern Great Basin of western North America. More detail on these topics, including estimated percentages within each condition class, is provided for the Mojave-Colorado Plateau ecotone spanning southern Nevada, the Arizona Strip northwest of the Grand Canyon, and southwestern Utah, including the area within the boundaries of the Southern Utah Fuel Management Demonstration Project (Hood and others, this volume; www.firelab.org/fep/research/sufm/home.htm). Fire Regime Condition Classes (FRCC) are an interagency, standardized tool for describing the natural range of variation in vegetation, fuels, and fire regime characteristics for a particular biogeographic region or vegetation type. They summarize how past land use and land management actions (or inactions) may have caused the FRCC to change.

The three shrubland vegetation types that we review are creosotebush scrub, blackbrush, and interior chaparral, which are typically situated along an increasing elevation gradient where they co-occur. The interior chaparral vegetation type is sometimes considered to include both Arizona chaparral, which occurs mostly in Arizona and New Mexico, but also in southern Utah and Nevada, and mountain brush (or Petran chaparral), which occurs primarily farther north in Utah and Colorado. We focus on Arizona chaparral. Two vegetation types dominated by big sagebrush (*Artemisia tridentata*) and pinyon-juniper (*Pinus edulis, Pinus monophylla*, and *Juniperus* spp.) often intergrade between blackbrush and interior chaparral in the alluvial soils of broad valleys and foothill toe-slopes. Kitchen and McArthur (this volume) reviewed big sagebrush and Tausch and Hood (this volume) reviewed pinyon-juniper.

Creosotebush Scrub

Introduction

Low cover (5 to 30 percent) of woody shrubs of various heights (0.5 to 1.5 m [1.6 to 4.9 ft] characterizes Creosotebush scrub (Vasek and Barbour 1995). It occurs across the warm desert regions of western North America and is the most common plant assemblage in the Mojave Desert (fig. 1) (MacMahon 2000). Creosotebush scrub is typically found below 1,500 m (4,920 ft) on well-drained alluvial flats and slopes below the blackbrush zone and above the saltbush zone that often occur within valley basins (Vasek and Barbour 1995). It phases into shrub-steppe in regions with high proportions of summer rainfall, typically encountered in the eastern Mojave Desert and Sonoran Desert.

Creosotebush scrub is dominated by the type-species creosotebush (*Larrea tridentata*), which has the highest cover and is the most wide-ranging plant species in the Mojave Desert (Rowlands and others 1982). It is most frequently associated with white bur-sage (*Ambrosia dumosa*), but a wide range of other plants can co-occur with creosotebush, including goldenhead (*Acamptopappus* spp.), saltbush (*Atriplex* spp.), Mormon tea (*Ephedra* spp.), goldenbush (*Ericameria* spp.), wild buckwheat (*Eriogonum fasciculatum*), ratany (*Krameria* spp.), winterfat (*Krascheninnikovia lanata*), boxthorn (*Lycium* spp.), indigo bush (*Psorothamnus* spp.), desert needlegrasses and Indian ricegrass (*Achnatherum* spp.), galleta grass (*Pleuraphis* spp.), cholla, beavertail, and other cacti (*Opuntia* spp.), and Joshua tree, Mojave yucca, and others (*Yucca* spp.). Dominant non-native species include red brome (*Bromus rubens*), Mediterranean split-grass (*Schismus* spp.), and red-stemmed filaree (*Erodium cicutarium*).

Figure 1—Creosotebush (*Larrea tridentata*) scrub in the Mojave Desert.

Within the Mojave-Colorado plateau ecotone, creosotebush scrub reaches its northeast limit in southwestern Utah near the entrance to Zion National Park and follows the Virgin River Valley to Lake Mead in Nevada. It occurs in most of this river drainage below 1,250 m (4,100 ft) elevation. Its distribution broadens below the Hurricane Fault in Utah, spreading northward as far as Browse along Interstate 15 and then following the base of the red cliffs throughout the Dixie Valley north of Washington, St. George, and Gunlock, Utah. There is a large stand south of St. George in the Blake Lambing Grounds and other stands are located throughout the Virgin River Gorge. Below the Gorge, the distribution spreads from the foot of the Beaver Dam Mountains and around Utah Hill, up the Beaver Dam Wash northward to Jackson Wash almost meeting with the creosotebush scrub community west of the town of Gunlock. From Littlefield, Arizona, there is continuous creosotebush scrub all the way to Lake Mead and Las Vegas, Nevada, with only minor interruptions in very rocky areas.

Creosotebush and associated plants provide much of the microhabitat diversity and vertical habitat structure in the Mojave Desert. Branches provide perches and nesting opportunities for songbirds, while desert tortoises (*Gopherus agassizii*) and a variety of rodents incorporate the root structure into their burrows. Shrubs also provide concealment and escape cover from predators and shade from the sun.

Many Mojave Desert shrubs are not fire-tolerant because their drought-adaptive features (thin bark, shallow root system, small leaves) and high dead-to-live woody material ratio make them vulnerable to fire. Some species, including creosotebush, can resprout after burning (fig. 2); however, survival rates decline significantly if more than 10 percent of their aboveground biomass is consumed by fire (Brooks and Minnich 2006). Native plants are generally slow to re-establish after fire and recurrent fire may prevent their re-establishment (Brooks and Minnich 2006; Brown and Minnich 1986; O'Leary and Minnich 1981). The loss of native plants can be followed by increased dominance of non-native annual grasses. The post-fire vegetation has typically lower species diversity and plant structural diversity than the native community, which can negatively affect the desert tortoise (Brooks and Esque 2002; Esque and others 2003) and other desert wildlife. Repeated burning in creosotebush scrub can lead to significant decreases in plant species richness with each subsequent fire (M. Brooks, unpublished data).

Figure 2—Creosotebush (*Larrea tridentata*) resprouting 1 year after a low intensity fire.

Historical Conditions

We do not know the pre-settlement fire conditions in warm desert plant communities such as creosotebush scrub because the typical analytical methods, such as dendrochronology and evaluating charcoal deposits in lakes, are not possible where the requisite trees and lakes are not present (Brooks and Minnich 2006; Esque and Schwalbe 2002). It is generally thought that fires in creosotebush scrub were an infrequent event in pre-settlement desert habitats because fine fuels from winter annual plants were probably sparse, only occurring in large amounts after exceptionally wet winters (Brooks and Esque 2002; Brooks and Minnich 2006; Brown and Minnich 1986; Esque and Schwalbe 2002; Humphrey 1974; O'Leary and Minnich 1981; Salo 2003). Fires were probably more frequent in adjacent shrub-steppe areas where perennial grasses provided additional fuel continuity to carry fire. It appears that wildfire was not historically a dominating influence in creosotebush scrub landscapes, except possibly where it intergraded with shrub-steppe.

Creosotebush scrub fuels are comprised primarily of woody shrubs, but it is the fine fuels from annuals and perennial grasses that facilitate the ignition and spread of fires. Native annual plants usually break down rapidly during the summer after they are produced and do not create a long-lived fuelbed (Brooks 1999a). As a result, the historical annual plant fuelbed, prior to the invasion by non-native species, was probably transient, only lasting for one summer fire season after winters of exceptional high rainfall. Perennial grasses may have been more prevalent in creosotebush scrub before the introduction of livestock grazing, providing another source of fine fuels that could additionally have helped carry fire in the past (Brooks and Minnich 2006). However, stands of perennial bunchgrasses were probably patchy, reflecting localized soil conditions, and the discrete clumps of fine fuels created by individual bunchgrasses did not likely create a continuous fuelbed.

Since settlement in the 1860s, creosotebush scrub has been used for livestock operations along the Mojave Desert-Colorado plateau ecotone and elsewhere throughout its range. Over the past two centuries, there has been widespread cattle and sheep use of this habitat. Widespread livestock grazing probably promoted the invasion of the annual grasses red brome, cheatgrass, and Mediterranean split-grass (Brooks and Pyke 2001), which create more continuous fine fuelbeds that can persist for years (Brooks 1999a). Although these non-native annual grasses have probably been present for over 100 years (Brooks 2000a, b; Esque and Schwalbe 2002; Salo 2003), wildfire in creosotebush scrub has only been prevalent in this region since the 1970s (Jerry Empey – BLM Color Country South Zone dispatch, personal communication). The recent increase in fire frequency was coincident with the end of the mid-century drought (Hereford and others 2006) and the beginning of a 20-year period of high rainfall in the late 1970s (Brooks and Esque 2002; Brooks and Minnich 2006).

Current Conditions

Fine fuels from non-native annual grasses currently represent the most important fuelbed component in creosotebush scrub (fig. 3) (Brooks and Minnich 2006). Non-native annual grasses typically comprise >50 percent of the total annual plant biomass in creosotebush scrub (Brooks 1999b; Brooks and Berry 2006). Annual plants were monitored as livestock/wildlife forage in unburned areas near St. George, Utah, from 1980 through 1995 and ranged from 0 to 700 kg/ha (624 lbs/acre) (BLM and T. Esque, unpublished data). In unburned areas near Littlefield, Arizona, the aboveground production of annual plants was over 1,000 kg/ha (892 lbs/acre) in 1993, and was comprised mostly of non-native annual grasses and forbs (Esque 1994). Above ground production of annual plants in a recurrently burned area of the Pakoon Basin of the Arizona Strip was measured at >2,000 kg/ha (1,784 lbs/ac) in 2001, and was mostly comprised of *Bromus spp*. (T. Esque, unpublished data).

During the 1980s and early 1990s, fire frequency increased substantially within the Mojave Desert (Brooks and Esque 2002; Brooks and Matchett 2006), and many of these fires occurred in creosotebush scrub. In the Opal Mountain and Stoddard Mountain regions of California, non-native annual grasses fueled recurrent fires, and some areas burned as many as three times in 10 years (M. Brooks, unpublished data). This increase in fire frequency is attributed to high rainfall in 1983 and 1992, which produced prodigious amounts of fine fuels, especially of red brome. Rainfall was positively correlated with fire size between 1980 and 2004 in the low-elevation ecological zone dominated by creosotebush scrub (Brooks and Matchett 2006). It appears that fine fuels produced during years of high rainfall are largely required for the development of large fires in this zone (Brooks and Matchett 2006; Brooks and Minnich 2006).

During the middle 1990s through 2004, the number of fires declined in lower and middle elevation zones where creosotebush predominates (Brooks and Matchett 2006). However, the amount of area burned at the upper elevation creosotebush ecotones increased slightly, reflecting a trend toward increasingly larger fires. This trend was punctuated during the summer of 2005 when hundreds of thousands of acres burned in areas dominated by creosotebush. The current fire return interval in this region is still unknown, but it seems to be decreasing due to the abundance of non-native invasive grasses after particularly wet years.

Figure 3—Highly flammable understory of red brome (*Bromus rubens*) in creosotebush (*Larrea tridentata*) scrub.

Fire Regime Condition Classes

Fire Regime Condition Class 1 (FRCC 1)

FRCC 1 is characterized by vegetation and fire regime attributes within the natural range of variation. The risk of losing key ecosystem components, such as habitat diversity and cover sites for the desert tortoise, is low. This is the natural condition where invasive annual grasses are largely absent, thus precluding the fine fuel buildup usually required to carry fire at lower and middle elevations in the Mojave Desert (Brooks and Matchett 2006; Brooks and Minnich 2006). Fire return intervals are extremely long, except for areas near the base of mountains that experience locally higher rainfall and fine fuel buildup from native annuals, or where creosotebush scrub intergrades with shrub-steppe. There is currently zero percent of the creosotebush scrub in FRCC 1 at the Mojave Desert-Colorado plateau ecotone in southern Utah, southern Nevada, and northwestern Arizona, because invasive, non-native annual grasses have infested the entire area. However, many of the more arid regions of the Mojave Desert further to the southwest in California may classify as Condition Class 1.

Fire Regime Condition Class 2 (FRCC 2)

FRCC 2 is characterized by vegetation and fire regime attributes that have been moderately altered from their natural range. The risk of losing key ecosystem components, such as habitat diversity and cover sites for the desert tortoise, is moderate. Patchy fires of limited extent are possible in more mesic regions during any year. Risk of larger, more continuous fires is high during 10 to 20 percent of all years when rainfall is particularly high. Fire risk can be exceptionally high when successive years of above average rainfall promote the accumulation of non-native annual grass fuels (in other words, production values in excess of 300 kg/ha (268 lb/acre) in 2 or more successive years. An example of this dynamic occurred in the Dixie Valley north of St. George, Utah, and on the Beaver Dam Slope north of Littlefield, Arizona (T. Esque, unpublished data). Approximately 80 percent of the creosotebush scrub at the Mojave Desert-Colorado Plateau ecotone is in this condition class.

Fire Regime Condition Class 3 (FRCC 3)

FRCC 3 is characterized by vegetation and fire regime attributes that have been significantly altered from their natural range. The risk of losing key ecosystem components, such as habitat diversity and cover sites for the desert tortoise, is high. Wildfires are frequent and widespread. Examples of FRCC 3 areas exist in the Pakoon Basin of northern Arizona, the Tule Desert of southern Nevada, and the Opal Mountain and Stoddard Valley areas of California. Return intervals in this condition class may be as short as 5 years. Fuel loads for areas in FRCC 3 may exceed 2,000 kg/ha (1,784 lbs/acre) during peak years (T. Esque, unpublished data). The invasive plant/fire regime cycle characterized by this condition class creates a feedback loop of decreasing habitat quality for wildlife and livestock. Native seedbanks may be depleted, lengthening the recovery times for native plants. Approximately 20 percent of the creosotebush scrub at the Mojave Desert-Colorado plateau ecotone is in this condition class.

Recommended Treatments

It is unlikely that areas can ever be restored to FRCC 1 due to the widespread distribution of non-native annual grasses, but the size of their impact may be mitigated through appropriate land management policies. We recommend suppression of all wildfires (because most native vegetation in this community vegetation type responds poorly to fires) and minimization of surface disturbances that promote the dominance of non-native annual grasses. Prescribed fires should not be conducted except for small research burns designed to evaluate fire behavior, fire effects, and fire management techniques and treatments. We also recommend a program of early detection, evaluation, and eradication for new invasive plants before they become established. Many new plant species are in the process of invasion, and some pose potential fire threats due to their ability to produce large amounts of continuous fine fuels. New invaders of particular concern are mustards (for example, *Brassica tournefortii* and *Hirschfeldia incana*) and perennial grasses (for example, *Pennisetum setaceum* and *Cenchrus ciliaris*) (Brooks and Esque 2002).

In FRCC 2 areas suppress all wildfires and avoid conducting prescribed fires except for small research burns. Livestock grazing may reduce fine fuel loads temporarily and may be effective for managing fuels in small defined areas, such as at the wildland urban interface. However, regular grazing is required to maintain these managed fuel zones, except during years of very low rainfall. Regular grazing may reduce dominance of late seral native plants and increase the dominance of non-native and early seral plants that are often more flammable.

We recommend studies designed for the restoration of FRCC 3 sites, which can include research on adapted plant materials, propagation techniques, and planting techniques. Suppress all wildfires and avoid conducting prescribed fires except for small research burns. Livestock grazing to reduce fuel loads may be counterproductive in the long run if it also hinders the re-establishment of late seral native plants.

Blackbrush

Introduction

The blackbrush vegetation type is characterized by relatively high cover (50 percent) of low statured (50cm [20 inches] tall), woody evergreen shrubs (fig. 4). It occurs at the bioregional transition between the Mojave and Great Basin deserts, from California through Nevada, Arizona, and Utah (Bowns 1973). Blackbrush is typically found in the elevational zone from 1,220 to 1,520 m (4,000 to 4,985 ft) above the creosotebush zone and below the interior chaparral or big sagebrush/pinyon-juniper zones (Beatley 1976; Bradley and Deacon 1967; Randall 1972). Within the Mojave-Colorado plateau ecotone, blackbrush is found on dry slopes and benches above the river canyons of southern Utah and northern Arizona (Turner 1994). It is also found mid-slope on mountain ranges throughout this ecotone.

Blackbrush is dominated by the type-species, blackbrush (*Coleogyne ramosissima*), which can comprise 90 to 95 percent of the total plant cover (Shreve 1942). Cover of blackbrush is highest in late seral stands on shallow, sandy soils with strong petrocalcic (caliche) horizons where it is the primary dominant plant species. Cover of blackbrush is lowest in deeper, silty soils, or at its upper or lower ecotones, where it is co-dominant with other native species such as creosotebush, juniper, desert almond (*Prunus fasciculata*), Anderson wolfberry (*Lycium andersonii*), Joshua tree (*Yucca brevifolia*), bladder sage (*Salazaria mexicana*), desert needle grasses, Indian ricegrass, and galleta grass. Dominant non-native species include the annuals red brome, cheatgrass, and red-stemmed filaree.

Blackbrush is used as winter forage by deer and bighorn sheep (Bowns and West 1976), and provides cover for nongame birds and small mammals (Brown and Smith 2000). It also protects soil from water and wind erosion and promotes soil fertility (Bowns 1973).

Figure 4—Blackbrush (*Coleogyne ramosissima*) dominated site with thread snakeweed (*Gutierrezia microcephala*), desert peach (*Prunus fasciculata*), and Joshua tree (*Yucca brevifolia*).

Blackbrush shrublands are one of the more flammable vegetation types in the desert bioregion due to the high proportion of fine fuels and optimal fuel bulk density. Blackbrush fires are usually stand-replacing, burning plants to the ground and killing most of them. Individual fires can kill upwards of 80 percent of the seeds of all species in the soil seedbank (Brooks and Draper 2006). It is commonly thought that blackbrush stands take centuries to recover (Bowns 1973; Webb and others 1987). However, when only a portion of a blackbrush plant is consumed, it may survive and resprout from the root crown (Bates 1984; M. Brooks, personal observation). Analyses of historical photographs from Joshua Tree National Park and southern Nevada also indicate that blackbrush stands can recover after as little as 50 to 75 years (Minnich 2003; M. Brooks, unpublished data). It seems probable that the ability of blackbrush to resprout after burning varies across its wide geographic range, which extends from the Colorado Plateau and southern Great Basin on through the Mojave Desert. In general, the frequency of blackbrush resprouting after partial consumption by fire seems to be highest in more mesic areas at the edges of its geographic range (M. Brooks, personal observation).

Historical Conditions

Blackbrush is considered a poor livestock forage species and ranchers noted that during the late 1930s and early 1940s, wildfires increased production of livestock forage in blackbrush rangeland of southern Nevada and northwestern Arizona (Anonymous 1945). In an attempt to further increase forage production, ranchers and the Bureau of Land Management (BLM) began a program of prescribed burning in the 1940s, during which time approximately 20 percent of the 161,875 ha (400,000 acres) of blackbrush were burned by prescribed fire or wildfire in southern Nevada (BLM, Las Vegas, Nevada, grazing district 5) (Croft 1950). Many blackbrush fires also occurred in northwestern Arizona during this time (BLM, Arizona Strip, Arizona, grazing district 2). Additional blackbrush burning likely occurred at least through the 1960s because a policy review during that time by the Range and Forestry Officer of the Bureau of Land Management in Nevada recommended that blackbrush burning be continued to increase livestock forage (Dimock 1960). Before 1940, fires in these regions were relatively uncommon (Croft 1950). The long-term effects of these mid-century range burns are currently being evaluated using repeat photography of historical photos originally taken 5 to 10 years post-fire and analyses of field reports, memos, and vegetation plot data collected during the late 1940s through the early 1960s by range conservationists and foresters from the BLM and the Forest Service, Intermountain Forest and Range Experiment Station (M. Brooks, unpublished data).

Prior to European contact (pre-settlement), late seral blackbrush stands were probably more extensive than they are today. The vast expanses of blackbrush rangeland that were burned to improve livestock production during the mid-1900s are still dominated by early seral species and have been re-colonized only sporadically by blackbrush (M. Brooks, unpublished data). Blackbrush within the Desert Wildlife Range and Nevada Test Site in southern Nevada that has not been managed for livestock production since the 1930s, and likely was not burned for rangeland improvement, does not currently contain evidence of widespread historical burning (M. Brooks and T. Esque, personal observations). It therefore appears that extensive burning to remove blackbrush probably created many of the vegetation stands where blackbrush is either absent or a sub-dominant species today.

The historical fuel complex in late seral blackbrush stands was probably similar to that observed in relatively undisturbed sites today, except for the current prevalence of *Bromus* spp. and *Erodium cicutarium* in many stands (Brooks and Matchett 2003). Vegetation characteristics of these stands were characteristic of blackbrush in FRCC 1 described below. Shrub cover was likely comprised primarily of blackbrush at 30 to 50 percent total cover, and interspaces were probably mostly bare, even during years of high rainfall, due to root competition from blackbrush. Other species, such as perennial grasses and early seral shrubs, probably occurred sporadically, as they do today, along wash stringers and on steep hillslopes where cover of blackbrush is typically low.

Low amounts of fine fuels in interspaces probably limited fire spread to only extreme fire weather conditions during which high winds, low relative humidity, and low fuel moisture led to high intensity stand-replacing crown fires. Natural fire return intervals appear to have been on the order of centuries (Webb and others 1987). The long intervals without fire allowed late seral blackbrush stands to re-establish.

Current Conditions

Blackbrush is considered to be one of the most flammable native plant assemblages in the Mojave Desert. Many large fires have occurred in this vegetation type since the 1980s in the Spring Mountains and Mormon Mountains in Nevada, the Beaver Dam Mountains in Utah, the Black Mountains and Virgin Mountains in Arizona, and at Joshua Tree National Park in California (Brooks and Esque 2002). Although fire is generally no longer advocated as a tool for range improvement, ignitions from lightning and accidental ignitions along roads have been sufficient to burn significant acreage of blackbrush during the past few decades.

At Joshua Tree National Park in California, blackbrush was burned during the early 1990s to reduce woody fuel loads at the wildland-urban interface between Joshua Tree and the town of Yucca Valley. During the first few post-fire years, the landscape was dominated by native annual wildflowers, but by the fourth post-fire year, the non-native annual grasses red brome and cheatgrass became the dominant annual plants and remained as such into the 2000s (Brooks and Matchett 2003; M. Brooks, personal observations). The appearance of this new flashy fuelbed resulted in a change in fire management at Joshua Tree. It put a stop to the use of fire as a management tool until prescriptions could be identified that would not create continuous fuelbeds of non-native annual grasses. At the current rate of burning during 1980s and 1990s, managers estimate that all the blackbrush at Joshua Tree National Park will burn by 2015 to 2020 (Hank McCutchen, Chief of Resources, personal communication). This is a significant concern because the blackbrush stands located there are disjunct from the rest of the blackbrush range, and if all the stands were to burn, it is very likely that blackbrush would not be able to re-establish.

The fuel complex in blackbrush appears to be more conducive to burning now than in the past. Non-native annual grasses currently occur in most blackbrush stands (M. Brooks, personal observations), although their dominance can vary significantly among sites (Brooks and Matchett 2003). Post-fire landscapes are even more dominated by these non-native grasses, which raises concerns that they will promote recurrent fire and prevent the re-establishment of *Coleogyne ramosissima*. This link between fine fuels and fire size is supported by recent analyses that demonstrate that years of high rainfall, which lead to high production of fine fuels, are correlated with larger fires in the elevations where blackbrush occurs in the Mojave Desert (Brooks and Matchett 2006). This is in contrast to the conclusions of Minnich (2003) who states that fine fuels that respond to short pulses of rainfall have less of an effect on fire regimes in blackbrush than woody fuels that accumulate slowly over time. Both conclusions probably have some validity, with fine-fuels taking precedence at lower elevations of the blackbrush zone and woody fuels taking precedence at higher elevations. At lower elevations, spacing between blackbrush plants is often relatively high (~1m [3.3 ft]). This requires fine fuels to carry fire under most circumstances, whereas at higher elevations spacing between blackbrush plants is often relatively small (<50cm [20 inches] (M. Brooks, personal observation).

Reports from the mid-1900s also acknowledge the role that non-native annual grasses, especially red brome, can play in facilitating the spread of fire in blackbrush (Dimock 1960; Holmgren 1960; Jenson and others 1960), although there was disagreement as to whether the burned landscapes were more or less susceptible to reburning than unburned landscapes. Jenson and others (1960) thought the chances of reburning were low because they observed low fine fuel levels in post-fire landscapes. However, their observations were made during the mid-century drought (Hereford and others 2006) when fine fuel loads were on the low end of their possible range. In contrast, Holmgren (1960), who accompanied Jenson and others on the same field visits, thought that the danger of accidental fire in blackbrush would be higher in areas that previously burned than in unburned areas if high winter rainfall had produced more red brome biomass and other fine fuels. Prior to the invasion of red brome and cheatgrass during the late 1800s to early 1900s (Brooks 2000a; Young 2000), fine fuel loads were likely not as great in either burned or unburned blackbrush stands. This resulted in fewer fires, possibly smaller fires, and fewer reburns.

Fire Regime Condition Classes

Fire Regime Condition Class 1 (FRCC 1)

FRCC 1 is characterized by vegetation and fire regime attributes within the natural range of variation. The risk of losing key ecosystem components, such as high blackbrush cover and associated protection from soil erosion, is low. Mature blackbrush stands fall into this condition class. These stands are typically late seral with occasional early seral patches created by infrequent stand-replacing fires.

The fire regime for FRCC 1 is active crown fire carried primarily by blackbrush and perennial grasses (*Achnatherum* spp. and *Pleuraphis* spp.) on deep silty soils. Burns are complete and stand-replacing, fire intensity is high, and fire return intervals are >100 years. Long fire return intervals allow for the typically slow process of blackbrush re-establishment. Approximately 20 percent of the blackbrush at the Mojave Desert-Colorado Plateau ecotone in southern Utah, southern Nevada, and northwestern Arizona is currently in this condition class.

Fire Regime Condition Class 2 (FRCC 2)

FRCC 2 is characterized by vegetation and fire regime attributes that have been moderately altered from their natural range. The risk of losing key ecosystem components, such as high blackbrush cover and associated protection from soil erosion, is moderate. Blackbrush stands with an intermix of late seral unburned patches and early seral burned patches are in this condition class. Blackbrush stands with patches that have been degraded by overgrazing, prescribed burning in the mid-1900s, or other forms of surface disturbance also fall into this condition class. These disturbances reduce cover of blackbrush and increase cover of early seral shrubs such as rabbitbrush (*Chrysothamnus* spp.), snakeweed (*Gutierrezia* spp.), and wild buckwheat, early seral herbaceous perennials such as desert globemallow (*Sphaeralcea ambigua*) and milkvetch (*Astragalus* spp.), and non-native annual plants such as red brome, cheatgrass, and red-stemmed filaree. Burned stands without livestock over-grazing that are situated on deep, silty soils can also have a large perennial grass component (*Achnatherum* spp. and *Pleuraphis* spp.).

Fires in late seral patches are active crown fires carried by blackbrush, perennial grasses (*Achnatherum* spp. and *Pleuraphis* spp.), and non-native annual grasses (*Bromus* spp.). Burns are complete and fire intensity is high. Fires within early seral patches are passive crown fires or surface fires carried by perennial grasses and non-native annual grasses between the sparse cover of early seral shrubs. Burns are patchy and fire intensity is low to moderate.

Fire return intervals in FRCC 2 stands are approximately 50 to 100 years. This shorter fire return interval, and over-grazing pressure from livestock, helps to maintain dominance by early seral species and may prevent re-establishment by blackbrush. Approximately 70 percent of the blackbrush at the Mojave Desert-Colorado plateau ecotone is in this condition class.

Fire Regime Condition Class 3 (FRCC 3)

Condition Class 3 is characterized by vegetation and fire regime attributes that have been significantly altered from their natural range. The risk of losing key ecosystem components, such as high blackbrush cover and its associated protection from soil erosion, is high. Blackbrush stands that burned during the 1900s, and have reburned at least once, fall into this condition class. These stands are typically dominated by non-native annuals and early seral perennials. Recurrently burned stands without livestock over-grazing can also have a large perennial grass component (*Achnatherum* spp. and *Pleuraphis* spp.).

The fire regime for this condition class is typically surface fire carried primarily by non-native annual plants. Burns are patchy, fire intensity is low, and fire return intervals are <50 years. Re-establishment by blackbrush is highly unlikely under this fire regime. Approximately 10 percent of the blackbrush at the Mojave Desert-Colorado plateau ecotone is in this condition class, but this percentage could increase if ignition rates increase with burgeoning human populations and if fine fuel continuity increases due to increased dominance of non-native annual grasses that could occur if atmospheric CO_2 and rainfall levels increase (Brooks and Pyke 2001).

Recommended Treatments

In FRCC 1 areas suppress human-caused fires, but consider allowing lightning-caused wildfires to burn unless significant populations of red brome or cheatgrass are present, or there are other reasons for suppression. Prescribed fires should not be conducted except for small research burns designed to evaluate fire behavior, fire effects, and fire management techniques and treatments. Only apply fuels management treatments at the wildland urban interface to reduce fire hazard, or in wildland areas where fuel breaks are deemed necessary to achieve management goals. Realize that regular maintenance may be required to maintain these managed fuel zones because fuel treatments that involve replacement of late seral woody fuels with early seral fine fuels will reduce fire intensity, but may increase susceptibility to ignition and rates of fire spread.

In FRCC 2 areas suppress all wildfires. Prescribed fires should not be conducted except for small research burns. Minimize livestock grazing and other surface disturbances on early seral stands, or where early seral and late seral stands are intermixed. Do not apply fuels management treatments on late seral stands, except possibly at the wildland urban interface to reduce fire hazards or in wildland areas where fuel breaks are deemed necessary to achieve management goals. Realize that regular maintenance may be required to maintain these managed fuel zones because fuel treatments that involve replacement of late seral woody fuels with early seral fine fuels will reduce fire intensity, but may increase susceptibility to ignition, fire spread rates, and fire frequency. Fuels treatments on early seral stands dominated by non-native annual plants may include the use of grass-specific herbicides. Carefully managed livestock grazing may reduce fine fuel loads temporarily, but may hinder the re-establishment of blackbrush and other late seral species and thus, may be counterproductive in the long-term.

In FRCC 3 areas suppress all wildfires. Prescribed fires should not be conducted except for small research burns. Extreme measures may be required in these stands. Revegetation with blackbrush and other late seral shrubs and perennial grasses, and exclusion of livestock grazing and other surface disturbances, may be necessary. Control of non-native annual grasses using herbicides or early season prescribed fire implemented immediately before revegetation treatments may improve initial establishment rates of revegetated plants.

Interior Chaparral

Introduction

Interior chaparral (Arizona chaparral) is characterized by moderate cover, ranging from 40 percent at dry sites to 80 percent at wetter sites, of moderately tall statured (1 to 2.5 m [3.3 to 8.2 ft] tall), woody evergreen shrubs with dense crowns (fig. 5) (Carmichael and others 1978). It is best represented in the foothills and mountain slopes and canyons in the sub-Mogollan region of central Arizona (Pase and Brown 1994). Disjunct stands also occur in the mountains of southwestern Utah, southern Nevada, northwestern and southeastern Arizona, southern New Mexico, southwest Texas, and northern Mexico. It generally occurs between 1,000 to 2,000 m (3,280 to 6,560 ft) in elevation above the desert grassland/shrub-steppe zone and below the pinyon-juniper/ponderosa pine (*Pinus ponderosa*) zone (Keeley 2000).

Within the Mojave-Colorado Plateau ecotone, interior chaparral is best represented in the Pine Valley Mountains, Bull Valley Mountains, and Zion National Park in Washington County (Utah), the Virgin Mountains of Mojave County (Arizona), and the Spring Mountains and Gold Butte regions of Clark County and various other mountain ranges in Lincoln County (Nevada). In this region, it is characterized by turbinella live oak

Figure 5—Interior chaparral dominated by turbinella live oak (*Quercus turbinella*).

(*Quercus turbinella*), buckbrush (*Ceanothus greggii*), pointleaf manzanita (*Arctostaphylos pungens*), Wright silktassel (*Garrya wrightii*), and narrowleaf yerba-santa (*Eriodictyon angustifolium*). Understory grass species, such as grama (*Bouteloua* spp.) and three-awn (*Aristida* spp.), occur throughout the range of interior chaparral.

Interior chaparral is an important vegetation community for wildlife. Game animals such as javelina, deer, and bighorn sheep use several plant species as forage and others as cover. Distinct from the communities above and below it, interior chaparral provides a significant amount of wildlife habitat diversity.

Most chaparral species are adapted to a fire-prone system, but methods of response vary. Turbinella live oak is well adapted to survive fire, typically resprouting vigorously from the root crown and rhizomes in response to fire or other disturbance (Pase 1969), while buckbrush and pointleaf manzanita regenerate from long-lived seeds that accumulate in the soil and germinate prolifically following fire. Areas dominated by non-sprouting chaparral species, including pointleaf manzanita, may develop a persistent cover of herbaceous species following fire, especially where *Bromus* spp. are present. Interior chaparral dominated by turbinella live oak, have fire return intervals from 74 to 100 years. At least 20 years may be required before these sites can reburn (Cable 1975).

Historical Conditions

In some areas prior to the 1900s, interior chaparral was considered good livestock range, but overgrazing has removed much of the perennial grass component since then (Nichol 1937). Most people now view interior chaparral as a nuisance or obstacle to human activity. Ranchers have routinely attempted to get rid of the brush that competes with forage grass and makes travel difficult. Livestock grazing and fire suppression have at least partly caused shrub encroachment into grassland and woodland/forest encroachment into shrublands during the 1900s in the southwestern United States (Hastings and Turner 1965; Leopold 1924; Miller and Rose 1999).

Many lower elevation interior chaparral sites have been managed for livestock grazing since the 1880s (Pase and Brown 1994). Where fire was used to maintain grass forage, interior chaparral probably did not encroach into lower elevation grasslands. However, where fire was not used, and the removal of fine fuels by livestock grazing and fire suppression further decreased the frequency of wildfire, interior chaparral very likely did encroach into lower elevation grasslands. Large areas near the early settlements of Prescott and Globe, AZ, were reported to be grasslands in the 1860s and became dense stands of interior chaparral by 1936 (Cable 1975). Aldo Leopold (1924) reported a substantial increase in "brush" cover since the 1880s at the expense of herbaceous plant cover after 40 years of livestock grazing.

Aldo Leopold made additional observations at the interior chaparral–grassland ecotone in southern Arizona, which sheds some light on the pre-settlement fire regime of this region. He noted during the early 1920s that there were multiple fire scars on ancient juniper stumps embedded in even-aged chaparral stands consisting of shrubs <40 years old. This suggests that the fire scars were created during low intensity grassland fires that pre-dated the current chaparral stands (Leopold 1924). Based on observations such as these, Leopold concluded that there had been no widespread fires in the chaparral–grassland ecotone in southern Arizona between the early 1880s and early 1920s. He further hypothesized that previous grassland fires at these same sites occurred at intervals of approximately once every 10 years before the advent of widespread livestock grazing.

Higher elevation interior chaparral sites likely did not receive as much grazing pressure, but fire suppression, especially at the interface with ponderosa pine forests, may have resulted in forest encroachment into chaparral shrublands. For example, where old chaparral stands intergrade with woodlands or forests at higher elevations, chaparral species such as Pringle manzanita (*Arctostaphylos pringlei*) and Fendler ceanothus (*Ceanothus fendleri*) may be replaced by ponderosa pine, emory oak (*Quercus emoryi*), or Arizona oak (*Quercus arizonica*) after long fire-free intervals (Pase and Brown 1994).

Historical fire return intervals in interior chaparral were likely 50 to 100 years (Cable 1975). However, this is an average over its entire range, and local intervals probably vary widely.

Current Conditions

Interior chaparral presents a more complex management challenge than the forests above or the deserts below, facing some of the challenges of both. Like the ponderosa pine forests, chaparral communities are fire-dependant. Exclusion of fire from interior chaparral can lead to encroachments by woodland and forest species. Like the creosotebush scrub and blackbrush desert shrublands, non-native annual grasses can increase fire frequency to the point where even the fire-adapted interior chaparral cannot recover. Thus, interior chaparral requires fire, but not too much fire.

Conventional wisdom in the fire suppression community is that chaparral either does not burn, or when it does burn, it burns very intensely. Due to threats to humans and their property, as well as to other high value resources, full suppression remains the primary response to wildfires that occur under the hot, dry conditions of summer in the southwest.

Fire Regime Condition Classes

Fire Regime Condition Class 1 (FRCC 1)

FRCC 1 is characterized by vegetation and fire regime attributes within the natural range of variation. The risk of losing key ecosystem components, such as cover for wildlife, is low. These stands are typically late seral with occasional early seral patches created by infrequent stand-replacing fires. The fire regime for FRCC 1 is active crown fire carried primarily by turbinella live oak, buckbrush, and manzanita species. Burns are complete and stand-replacing, fire intensity is high, and fire return intervals are between 50 and 100 years.

Long fire return intervals allow for the re-establishment of seed banks and the development of the fuel loads and spatial continuity necessary for fire to occur. Approximately 40 percent of the interior chaparral at the Mojave Desert-Colorado plateau ecotone in southern Utah, southern Nevada, and northwestern Arizona is in this condition class.

Fire Regime Condition Class 2 (FRCC 2)

FRCC 2 is characterized by vegetation and fire regime attributes that have been moderately altered from their natural range. The risk of losing key ecosystem components, such as cover for wildlife, is moderate. There are two types of chaparral within FRCC 2: 1) sites where the fire return intervals are greater than the natural range of variation (>100 years) and encroachment of interior chaparral by conifer woodlands has occurred (10 percent of the Mojave Desert-Colorado plateau ecotone region) and 2) sites where fire return intervals, or other landscape-scale disturbances, are within or slightly less than the natural range of variation (≤50 to 100 years) and the post-disturbance community contains a significant intermix of late seral unburned patches and early seral burned patches dominated by non-native grasses (20 percent of the Mojave Desert-Colorado plateau ecotone region). These sites typically retain significant amounts of area with late seral interior chaparral characteristics, but the trend is toward FRCC 3.

Fire Regime Condition Class 3 (FRCC 3)

FRCC 3 is characterized by vegetation and fire regime attributes that have been significantly altered from their natural range. The risk of losing key ecosystem components, such as cover for wildlife, is high. Chaparral stands that have burned repeatedly and lost most of the shrub cover, have lost the shrub seed bank, and have significant amounts of *Bromus* spp. or other non-native grasses fall into this condition class. Non-native annuals and early seral perennials typically dominate these stands.

The fire regime for this condition class is typically surface fire carried primarily by non-native annual plants. Burns are patchy, fire intensity is low, and fire return intervals are <20 years. Re-establishment by native interior chaparral shrub species is diminished under this fire regime and becomes less likely as native chaparral seed banks disappear.

Approximately 30 percent of the interior chaparral at the Mojave Desert-Colorado plateau ecotone is in this condition class, but this percentage could increase if ignition rates increase with burgeoning human populations and if fine fuel continuity increases with increased dominance of non-native annual grasses that could occur if atmospheric CO_2 and rainfall levels increase.

Recommended Treatments

In FRCC 1 areas suppress human-caused fires. Consider allowing lightning-caused fires to burn unless significant populations of *Bromus* spp. are present, especially near stands of non-sprouting chaparral species, or there are other reasons for suppression. Prescribed fires should not be conducted except for small research burns designed to evaluate fire behavior, fire effects, and fire management techniques and treatments. Apply fuel management treatments at the wildland urban interface to reduce fire hazard and in wildland areas where fuel breaks are deemed necessary to achieve management goals. Where possible, use indirect fire suppression tactics that provide a reasonable containment strategy to protect human life, property, and other valuable resources while allowing some acres to burn during the high-intensity summer fire season where deemed appropriate.

Since this is a fire-dependent ecosystem, counter the effects of fire suppression in FRCC 1 areas by implementing burn treatments unless a significant threat of conversion to a *Bromus*-dominated system is present. The goal of these treatments would be to replicate normal patch size/mosaic patterns to provide diversity and to reduce the potential size of wildfires. Treatment frequency should not occur more often than 20 years at any site. Treatment areas should not be reseeded with herbaceous vegetation unless necessary to compete with non-native annual grasses. Treatments on sites dominated by seed propagators (for example, pointleaf manzanita) may need to be seeded with chaparral species.

Realize that regular maintenance may be required to maintain FRCC 1 managed fuel zones because fuel treatments that involve replacement of late seral woody fuels with early seral fine fuels will reduce fire intensity, but may increase susceptibility to ignition and rates of fire spread.

Treat FRCC 2 sites that have had fire excluded and are being encroached by conifer woodlands in a manner similar to FRCC 1, except where non-native grasses are a significant threat. In other words, suppress human-caused fires, but consider allowing lightning-caused fires to burn unless significant populations of *Bromus* spp. are present or there are other reasons for suppression. Prescribed fires should not be conducted except for small research burns. Apply fuels management treatments at the wildland urban interface to reduce fire hazard and in wildland areas where fuel breaks are deemed necessary to achieve management goals. Where possible, use indirect fire suppression tactics that provide a reasonable containment strategy to protect human life, property, and other valuable resources while allowing some acres to burn during the high-intensity summer fire season.

Treat FRCC 2 sites where disturbances have occurred, but the post-disturbance community has changed to one with a significant intermix of late seral unburned patches and early seral burned patches with non-native grasses, in a manner similar to FRCC 3. In other words, suppress all fires; revegetate with live oak, buckbrush, manzanita, and other late seral shrubs and perennial grasses; exclude livestock grazing and other surface disturbing activities at least until plants become established, and control non-native annual grasses using herbicides).

In FRCC 3 areas suppress all wildfires. Prescribed fires should not be conducted except for small research burns. Revegetation may be necessary using turbinella live oak, *Ceanothus* spp., and manzanita species. and other late seral shrubs and perennial grasses, along with exclusion of livestock grazing and other surface disturbances. Control of non-native annual grasses and other invasive plants using herbicides immediately before revegetation treatments may improve initial establishment rates of revegetated plants and increase survival rates of regenerating plants.

We recommend research in adapted plant materials, propagation techniques, and planting techniques for restoration of these FRCC 3 sites. The grass/fire cycle creates a feedback loop of decreasing habitat quality for wildlife and livestock. Livestock grazing to reduce fuel loads may be counterproductive in the long run if it also hinders the re-establishment of late seral native plants.

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