BIOLOGY, ECOLOGY, AND MANAGEMENT OF
WESTERN JUNIPER

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Biology, Ecology, and Management of
Western Juniper
(Juniperus occidentalis)

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Cover: A lone old-growth tree surrounded by post-settlement western
juniper on top of Fredricks Butte, Lake County, Oregon.

CONTENTS

Summary

The rapid expansion of western juniper into neighboring plant communities during the past 130 years has caused considerable concern because of increased soil erosion, reduced stream flows; reduced forage production; altered wildlife habitat; changes in plant community composition, structure, and biodiversity; and the replacement of mesic and semi-arid plant communities with woodlands. However, the impacts of post-settlement woodland expansion are not always clear or consistent across sites and have led to debate and legal challenges over control projects and management plans for western juniper.

This publication represents a synthesis of what is known about the history, biology, ecology, and management of western juniper. Western juniper occupies 9 million acres in central and eastern Oregon, northeastern California, southwestern Idaho, and northwestern Nevada, and occurs in a few outlying stands in southern Washington. Presettlement changes in woodland abundance and distribution are largely attributed to long-term changes in temperature, amounts and distribution of precipitation, and the extent and return intervals of fire. Evidence supporting rapid post-settlement expansion is derived from old surveys, photographs, the distribution of relict presettlement woodlands, and tree-ring chronologies.

Western juniper represents the northwestern portion of the piñon and juniper region in the Intermountain West. The tree is submonoecious and develops male cones in early spring, which attain full size the first summer and mature during the second summer. Female cones persist on trees for nearly 2 years. Seeds are dormant and germination potential is greatly enhanced by prolonged cool-moist stratification, which is cumulative from year to year. Seed dispersal of western juniper occurs through gravity, overland flow, and animals. At least 12 species of birds feed on the fruits and as a group are the most important disseminator's of western juniper seed. Western juniper grows on a wide variety of parent materials and soils including materials derived from aeolian (e.g., pumice sands), sedimentary, and igneous sources (e.g., rhyolite, andesite, basalt). Soil textures range from clay to sandy and soil temperature regimes from mesic to frigid.

Western juniper communities may be separated into presettlement (old-growth) or post-settlement (expansion) communities. We suggest 1870 as a cut-off to separate the two age classes. Western juniper is a long-lived species (more than 1,000 years). However, old-growth represents only a small proportion of the population throughout most of its range with the exception of the Mazama Ecological Province. Old-growth trees and stands can easily be separated from post-settlement stands based on morphological and stand structure characteristics. The majority of post-settlement communities are still in a state of transition. The stage of woodland succession (defined in this publication as Phases I, II, and III) directly affects plant community structure, composition, seed pools, wildlife habitat, and ecological processes including hydrologic and nutrient cycles. The phase of woodland development also affects the selection of management treatment, response following treatment, follow-up management, and treatment cost. As the tree layer increases in dominance, the shrub and herb layer decline. The degree that the herb layer is depleted is dependent upon soil depth to a restrictive layer. The minimum time for the tree overstory to begin suppressing the understory is 45–50 years and to approach stand closure 70–90 years on cool wet sites and 120–170 on dry warm sites. Western juniper expansion into sagebrush grassland

and diversity also decline. Western juniper has significantly increased in density and distribution since the late 1800's and if left unchecked can have significant impact on soil resources, plant community structure and composition, water and nutrient cycles, and wildlife habitat. As a result, control of western juniper has been a major concern of land management since the early 1960's. In the 1960's through the early 1970's chaining and dozing were the most common forms of western juniper control. In the 1970's, chainsaws became a widespread tool used for juniper control. In the 1990's, the use of prescribed fire for juniper control also increased. This document evaluates different western juniper control practices including fire, mechanical, chemical, seeding, and post-treatment grazing. Specific concerns regarding the justifications used to support western juniper removal are also discussed. A large concern in woodland control treatments is weed infestation. Weed response following woodland conversion projects is site-specific and depends heavily on the initial floristics of each plant community. The ecological site (especially where it fits along the gradient of warm-dry to cool-moist), initial floristics, and the stage of woodland development are very important factors that will influence the response of a site following thinning or total removal of trees. A framework of questions are defined that will help land managers and private landowners select the most appropriate management action. There are some commercial uses of juniper but profit margins are often marginal. To date, products include firewood, chips for particle-flake board and animal bedding, decking, interior paneling, doors, cabinetry, rustic furniture, picture frame molding, small gifts, Christmas decorations, and the female cones are used as flavoring for gin. A great deal has been learned about the ecology, biology, history, and management of western juniper over the past several decades. However, not all of the questions have been answered in some areas, reducing but not totally limiting our ability to manage western juniper on an ecosystem basis.

can affect the spatial distribution of soil organic matter, carbon, and nutrients. The loss of nutrients will also increase if woodland development results in accelerated erosion. Changes in hydrologic processes and water balance as tree abundance and dominance increase are not well understood. Evidence suggests that juniper can impact infiltration rates, sediment loss, and soil water storage and depletion rates. Accelerated soil water depletion rates in western juniper-dominated stands can decrease the length of the growing season by as much as 4–6 weeks. However, the impacts of western juniper on the water balance at the watershed or basin level have not been determined, nor have the effects of woodlands on subsurface flow into streams and springs. A large variety of wildlife species use early transitional states of woodlands that still contain an understory of shrubs and herbs. However, as structural diversity declines with increasing tree dominance, wildlife abundance

Introduction

Western juniper *(Juniperus occidentalis var. occidentalis Hook.)*[1] has occupied its current range for several thousand years. Its rapid expansion into neighboring communities the past 130 years has caused considerable concern because of increased soil erosion; potential reduced stream flows; reduced forage production; altered wildlife habitat; changes in plant community composition, structure, and biodiversity; and the replacement of mesic and semi-arid plant communities with woodlands[2]. However, western juniper has been reported to be a valuable source of wildlife habitat throughout the literature and has aesthetic appeal. The wood has been used for various products including firewood, fence posts, and commercial energy production. The impacts of post-settlement woodland expansion are not always clear or consistent and have led to debate and legal challenges over control projects and management plans for western juniper. This document represents a synthesis of what is known about the history, biology, ecology, and management of western juniper. We hope to dispel some of the myths, identify knowledge gaps, sort out some of the issues related to woodland expansion, and increase the overall understanding of western junipers place and function in the northern Great Basin. This synthesis will provide guidance for defining long-term goals, setting management priorities, and developing management plans and strategies related to western juniper

This publication is separated into six major sections: 1) distribution and history of woodland expansion, 2) life history and biology, 3) ecology; 4) hydrology, 5) restoration and management, and 6) management guidelines. Subsections within each category allow readers to easily refer to specific subject areas related to western juniper. We cite some literature associated with other juniper species to help put western juniper communities into a larger context of juniper and piñon woodlands in the American West and to support ecological concepts and management action. However, the focus of this paper is on western juniper.

Distribution and History of Woodland Expansion

Distribution

Juniper and piñon woodlands currently occupy over 74 million acres in the western United States (West 1999). The northwestern portion of the piñon and juniper region is represented by western juniper. Western juniper occupies 9 million acres in central and eastern Oregon, northeastern California, southwestern Idaho, and northwestern Nevada, and occurs in a few outlying stands in southern Washington (Table 1, Fig. 1) (USDA Forest Service 1981, Gedney et al. 1999, Miller and Tausch 2001, Azuma et al. 2004). Western juniper is usually the only conifer species occupying a site except where western juniper woodlands adjoin ponderosa pine *(Pinus ponderosa)* forests. Precipitation across most of the western juniper zone varies between 10 and 15 inches (Gedney et al. 1999), most of which falls during the winter and spring (October through June). However, western juniper can grow in areas receiving as little as 7 inches or exceeding 20 inches of precipitation annually. It grows over a wide array of environments and occupies elevations ranging from 600 to 8,000 ft (Sowder and Mowat 1958, Miller and Rose 1995, Gedney et al. 1999, Miller et al. 2000). Nevertheless, most of western juniper woodlands and savannas are found between 2,000 and 6,000 ft (Gedney et al. 1999). Western juniper is usually not found above 7000 ft because its foliage is damaged by extreme winter temperatures (Miller and Rose 1995).

A second variety of western juniper, Sierra juniper *(J. occidentalis var. australis)*, extends along the eastern slopes of the Sierra Nevada mountain range south of Susanville, California, and east and south from the Anchorite Hills (south of Walker Lake in Nevada near the California border) to the Panamint and San Bernardino mountains (USDA Forest Service 1981)(Fig. 1). This species is usually found growing as widely scattered trees mixed with other conifers at elevations between 4,100 and 9,100 ft. Recent work has documented small pockets of Sierra juniper growing in the mountains of central and eastern Nevada (Charlet 1996). Although stands typically occur well above Utah juniper *(J. osteosperma)* in this region, mixed stands including hybrids of Sierra and Utah juniper are occasionally found along drainages at lower elevations (Charlet 1996, Terry et al. 2000).

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[1] All scientific names used throughout the text are from Cronquist, A.A., et al. 1972–1996. *Intermountain-flora: vascular plants of the Intermountain West, USA.*

[2] Western juniper woodlands are defined as having more than 10 percent tree canopy compared to savannas that have less than 10 percent tree canopy (Gedney et al. 1999).

Presettlement Expansion

The distribution and density of western juniper changed significantly across the Intermountain West around the late Pleistocene and into the Holocene. Changes in woodland abundance and distribution are largely attributed to long-term changes in temperature, amount and distribution of precipitation, and the extent and return interval of fire (Davis 1982, Thompson and Hattori 1983, Mehringer 1987, Van Devender et al. 1987, Wigand et al. 1995). During much of the Pleistocene, 45,000–12,500 years BP (before present), western juniper had a much more southerly distribution, with the northern boundary near Kings Canyon National Park, California (Cole 1983). Towards the end of the Pleistocene, 12,000–15,500 years BP, its northern-most distribution was the Winnemucca Lake Basin in Nevada (Thompson 1984). It was also located on the eastern shore of Pluvial Lake Lahontan (Thompson et al.1986). Only prostrate juniper (*J. horizontalis*) and common juniper (*J. communis*) occupied southeastern Oregon at the end of the Pleistocene (Wells 1983). As temperatures warmed during the early Holocene, western juniper began migrating north into its present range. Macrofossils (leaves, twigs, and seeds) from pack rat middens found in caves at the Lava Beds National Monument in northern California date its arrival around 5,300 years BP (Mehringer and Wigand 1984). In Oregon, the earliest evidence of western juniper (pollen from pond and lake sediment cores) was dated 6,600 years BP in the Fort Rock Basin in south-central Oregon (Bedwell 1973) and 4,800 years BP at Diamond Craters in eastern Oregon (Wigand 1987).

Since the arrival of western juniper in central and eastern Oregon, northeastern California, and southeastern Idaho, its abundance and distribution have fluctuated (Mehringer 1985, Mehringer and Wigand 1990, Miller and Wigand 1994). Following a very dry period during the mid-Holocene, 7,500–5,000 years BP, western juniper rapidly expanded into its new range. Precipitation increased while temperatures remained warm between 5,000 and 4,000 years BP (Davis 1982, Mehringer 1986, Wigand 1987). Between 4,000 and 3,000 years BP climatic conditions were relatively wet and cool. Western juniper continued to increase, but retreated from higher elevations and expanded to lower elevations during this period. Western juniper reached most of its current geographic range approximately 3,000 years BP (Wigand et al. 1995). Severe drought and major fires during the late Holocene, 2,500–1,500 years BP, resulted in regional declines in western juniper (Mehringer and Wigand 1987, Wigand et al. 1995). Around 1,200 years BP summer precipitation increased, resulting in increases in abundance of both grasses and western juniper. A drying period between 900 and 700 years BP again reduced woodland abundance (Wigand et al. 1995).
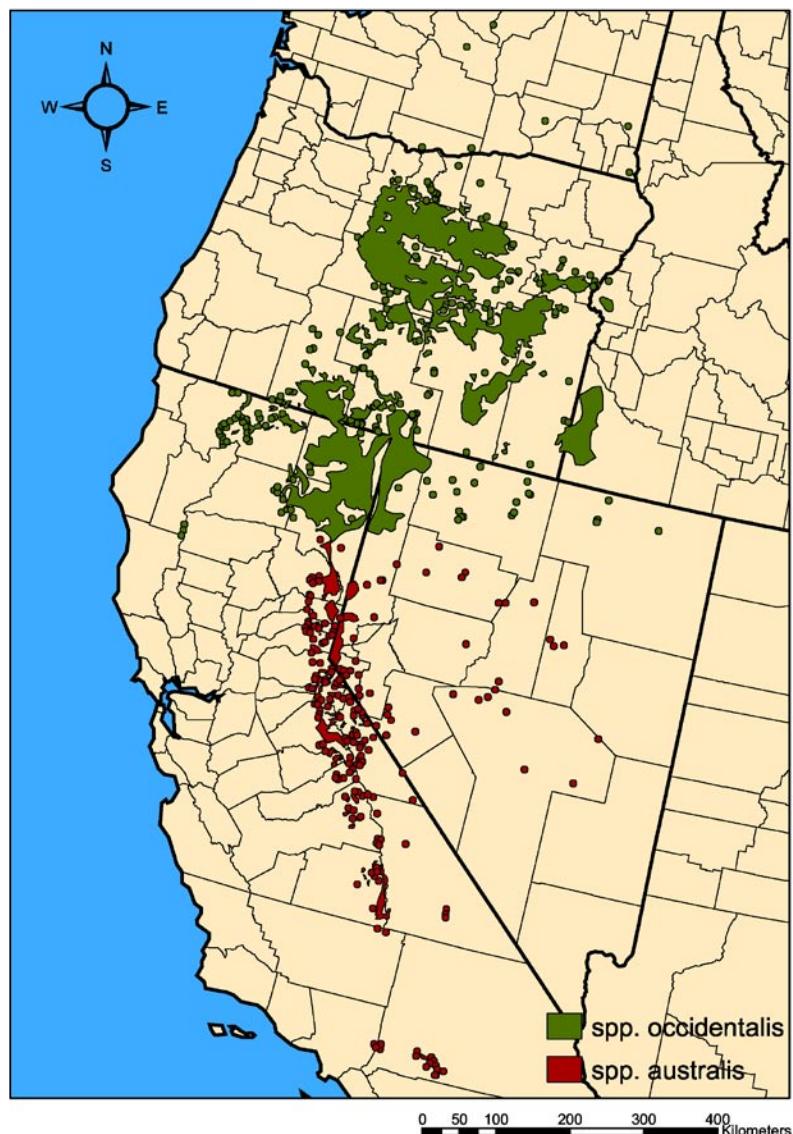
Table 1. Estimated area occupied by western juniper woodlands (tree canopy more than 10 percent cover) and shrub steppe savannas (tree canopy less than 10 percent).

State	Community structure	Acres	Reference
California	woodland	1,284,000	Bolsinger 1989
	savanna	797,000	Bolsinger 1989
Idaho	woodland	250,000	Chojnacky 1995
Nevada	mixed	100,000	estimate[1], no reference
Oregon	woodland	2,239,000	Gedney et al. 1999
	savanna	2,818,000	Gedney et al. 1999
Washington	savanna	trace	
Total		**7,488,000 (9,000,000)[2]**	

[1]Estimated from USGS 1:250,000 maps by R.F. Miller, EOARC, Oregon State University.
[2]An in-progress juniper inventory has increased the acreage in Oregon an additional 1.5 million acres bringing the total to nearly 9 million acres (Azuma et al. 2004)

Figure 1. Distribution map of western juniper *(Juniperus occidentalis* var. *occidentalis)* and Sierra juniper *(J. occidentalis* var. *australis)* (derived from Griffin and Critchfield 1972, Charlet 1996, Gedney et al. 1999, and USGS 1:250,000 maps; *developed by Steve Petersen, Department of Rangeland Ecology and Management, Oregon State University, Corvallis, Oregon).*

The Little Ice Age, 700–150 years BP, was the wettest and coolest period during the last half of the Holocene. Increased grass cover during this period (Wigand et al. 1995) probably supported higher fire frequencies (Gruell 1999, Miller and Rose 1999), which limited woodland distribution and abundance (Wigand 1987, Miller and Wigand 1994). The abundance of juniper pollen has gradually increased since 1500 A.D., fluctuating in the early 1800's and sharply increasing in the mid-1900's (Mehringer 1987). Since the end of the Little Ice Age around 1850, annual temperatures have been slowly but steadily rising (Ghil and Vautgard 1991). Relict juniper woodlands, tree age chronology data, down and dead trees and stumps, and historic documents (i.e., surveys) generally indicate that presettlement western juniper trees were typically confined to rocky ridges, low sagebrush (*Artemisia arbuscula)* flats, and pumice soils where fine fuels were too low in abundance to carry fire (Burkhardt and Tisdale 1976, Vasek and Thorne 1977, Holmes et al. 1986, Miller and Rose 1995, Waichler et al. 2001). The physiognomy of most stands was sparse and savanna-like (less than 10 percent tree canopy cover) on the rocky shallow soils and open-canopy woodlands (10–25 percent tree canopy cover) in the pumice region.

Figure 2. Keystone Ranch east of Prineville, Oregon, in Crook County on Ochoco Creek. Majority of trees are juniper with a few ponderosa pine. The smaller trees in the foreground of Figure 2a appear to be about 10 to 25 years old, and larger trees 60 to 70 years. *Photo by Stu Garrett.*

Figure 2a. Keystone Ranch, about 1890.

Figure 2b. Keystone Ranch, 1989.

Post-settlement Expansion

During the past 130 years, western juniper has been expanding within its geographic range at unprecedented rates compared to any other time period during the Holocene (Miller and Wigand 1994, Miller and Tausch 2001). Historical expansions of western juniper and other piñon and juniper species throughout the West are well documented in the literature (Cottam and Stewart 1940; Burkhardt and Tisdale 1976; Tausch et al. 1981; Tausch and West 1988, 1995; Miller and Rose 1995, 1999; Gedney et al. 1999; O'Brien and Woudenberg 1999; Soulé and Knapp 1999; Tausch and Nowak 1999; Coppedge et al. 2001; Soulé et al. 2004). For western juniper, evidence supporting rapid post-settlement expansion is derived from old surveys, photographs (i.e., Fig. 2), the distribution of relict presettlement woodlands, and tree ring chronologies. Limited evidence suggests western juniper began increasing its range following the end of the Little Ice Age in 1850 (Mehringer personal communication, Johnson 2005). However, its rapid increase in abundance and expansion since the late 1800's (Table 2) has largely been attributed to anthropogenic factors (Miller and Wigand 1994, Knapp et al. 2001b, Miller and Tausch 2001).

Western juniper is a long-lived species[3] and presettlement woodlands have been in place for hundreds and thousands of years (EOARC[4], unpublished data). However, presettlement western juniper stands outside of the Mazama Ecological Province[5] are estimated to account for only 10 percent or less of present day woodlands (Miller et al. 1999a, Johnson 2005). Most woodlands have developed during the past 130 years. Western juniper woodlands in eastern Oregon with more than 10 percent canopy cover increased from 456,000 acres in 1936 (Cowlin et al. 1942) to 2.2 million acres in 1988 (Gedney et al. 1999). Other evidence supporting the post-settlement expansion of western juniper is the sharp rise in pollen in the mid-1900's, which Mehringer (1987) detected in lake sediment cores. The presence of old stumps and logs, which can persist on a site for hundreds of years in this semi-

[3]The oldest western juniper aged to date is 1600 years old, located on Horse Ridge, Oregon.

[4] EOARC Eastern Oregon Agricultural Research Station, Burns, Oregon, jointly operated by Oregon State University and USDA Agricultural Research Service.

[5]An ecological province is a subdivision of a region having a distinctive combination of geological features and ecological sites (Anderson et al. 1998).

[6]Alliance is a physiognomically uniform group of plant associations sharing one or more dominant or diagnostic species, which, as a rule, are found in the uppermost stratum of the vegetation (Grossman et al. 1998).

arid climate, are good indicators as to whether woodlands were present on a site prior to the 1860's. Old stumps and logs in post-settlement western juniper woodlands associated with aspen *(Populus tremuloides)*, riparian, and the majority of the mountain big *(Artemisia tridentata* var. *vaseyana)* sagebrush alliance[6] across the High Desert and Klamath Ecological Provinces are absent or rare (Miller and Rose 1995, 1999; Miller et al. 2000; Miller and Tausch 2001; Wall et al. 2001).

The strongest evidence for the post-settlement expansion of western juniper is from tree-ring chronologies (Table 2, Fig. 3). These chronologies, which describe the age composition and establishment of woodlands over time, show a rapid increase in establishment since the 1870's (Miller and Tausch 2001, Soulé et al. 2004). In southeastern Oregon and southwestern Idaho, peak establishment in some closed woodland stands[7] occurred between 1900 and 1920 (Fig. 3b) (Miller and Rose 1999,

EOARC unpublished data). This was a period of above-average precipitation. A similar peak was reported for woodlands of Utah juniper in Nevada (Tausch et al. 1981).

A similar pattern of western juniper encroachment has occurred in aspen communities throughout the range of western juniper (Fig. 3d) (Miller and Rose 1995, Wall et al. 2001). In southeastern Oregon, northeastern California, and northwestern Nevada, 12 percent of the aspen stands (n = 100) measured were completely replaced by western juniper (Wall et al. 2001). These stands were identified as previously being dominated by aspen based on the high density of dead aspen logs in the understory. In addition, post-settlement western juniper was the dominant tree species in 23 percent of the stands and common to codominant in 42 percent of the aspen stands measured. Western juniper began invading aspen stands in the 1890's, with peak establishment occurring between 1900 and 1940 (Table 2). No western juniper in these aspen stands exceeded 130 years in age.

In much of its range, western juniper has increased the area it occupies by an estimated 10-fold in the past 130 years (Miller et al. 1999a) and has the potential to occupy far more area than it now does (West and Van Pelt 1986, Betancourt

[7]Closed stands are sites where western juniper is the dominant vegetation layer and the primary species controlling ecological processes on the site. The physiognomy of closed stands are characterized by a dominant overstory layer of mature trees and often suppressed subcanopy trees, and a shrub canopy of less than 5 percent cover.

Table 2. Decadal initiation of western juniper expansion, location of study, and period of peak establishment based on tree-ring data. Sample size = number of trees sampled (from Miller and Tausch 2001).

Cover type	Initiation	Peak	Location	Sample size	Reference
Sagebrush Mountain big	1860's	1880–1920	e OR	<1,000	Gedney et al. 1999
sagebrush	1890's	1902–1936	Silver Lake, OR	228	Adams 1975
	1870's	1910–1940	Owyhee Mt, ID		Burkhardt & Tisdale 1976
	1880's	1900–1910	Prineville, OR	>1,000	Eddleman 1987
	1850's		Juniper Mt & South Mt, ID	>500	Johnson 2005
	1850's	1900–1920	Juniper Mt, ID	>1,000	Johnson 2005
	1880's		Steens Mt, OR	>500	Johnson 2005
	1880's		Steens Mt, OR	>1,000	Miller & Rose 1995
	1870's		Kiger Gorge, OR	240	Unpublished data
	1870's		Hart Mt, OR		Gruel 1999
	1870's	1905–1925	Paisley, OR	>1,000	Miller & Rose 1999
	1870's		Lava Beds NM, CA	715	Miller et al. 2003
Wyoming & low sagebrush	1880's	1890–1910	nw CA	<100	Young & Evans 1981
Low sagebrush	1870's		Paisley, OR	500	Miller & Rose 1995
Aspen	1890's	1910–1940	se OR, ne CA, nw NV	>1,000	Wall et al. 2001

1987, Miller et al. 2000). Most of the 9 million acres occupied by western juniper is still in transition from shrub-steppe to western juniper woodland (Miller et al. 2000) and the species continues to expand its range and increase in density (Miller and Rose 1995, 1999; Knapp and Soulé 1998; Wall et al. 2001), even in the absence of livestock grazing (Soulé et al. 2004).

Factors affecting post-settlement expansion

Factors most frequently attributed to the increase in both density and area of piñon and juniper are climate, the introduction of livestock, industrial increases in atmospheric CO_2, and the reduced role of fire (Fig. 4).

Climatic influences

From 1850 to 1916, winters became milder and precipitation was greater than the current long-term average in much of the Great Basin (Antevs 1938, Wahl and Lawson 1970, LaMarche 1974, Graumlich 1987). There is some indication that woodland expansion was initiated between 1850 and 1870 in some areas prior to European settlement (Fig. 3b) (P.E. Mehringer, Department

of Anthropology and Geology, Washington State University, personal communication; Johnson 2005). However, expansion across the majority of areas sampled occurred in the late 1800's (Table 2; Fig. 3a, c, d). Annual tree ring growth in western juniper is strongly related to local climatic conditions (Pohl et al. 2002). Soulé et al. (2004) reported that western juniper annual ring growth across five sites in eastern Oregon were above-average from the late 1800's through the early 1900's. This wet period coincides with post-settlement establishment and the peak period of woodland establishment for closed stands (Table 2). Wet, mild conditions promote vigorous growth in western juniper (Fritts and Wu 1986, Holmes et al. 1986).

Livestock grazing

Introduction of livestock in the 1860's and the large increase of animals from the 1870's through the early 1900's (Oliphant 1968, Miller et al. 1994) coincide with the initial expansion of western juniper woodlands. Season-long grazing by the large numbers of domestic livestock during this period is believed to have reduced fine fuel loads, thus contributing to a significantly reduced role of fire in the northern Great Basin (Burkhardt and Tisdale
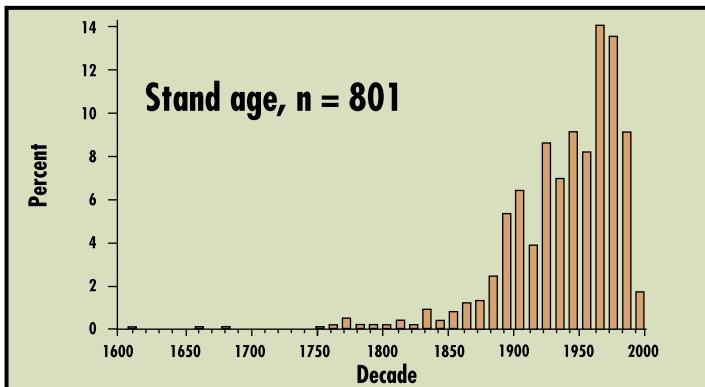
Figure 3a. Lava Beds National Monument in northern California.
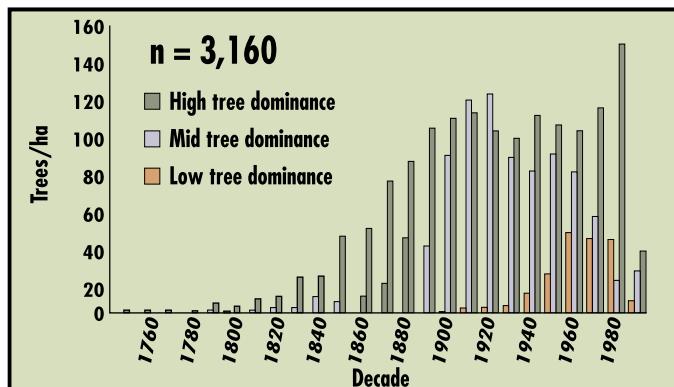
Figure 3b . Combined chronologies from 42 miles of transects in Steens Mountain southeastern Oregon, and South Mountain and Juniper Mountain in southwestern Idaho.
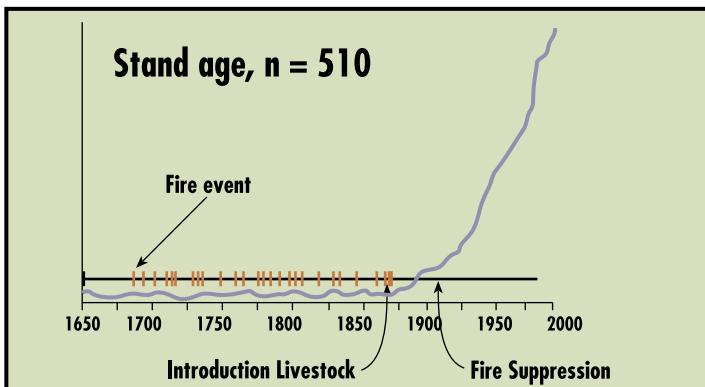
Figure 3c. Chewaucan River basin in the Paisley Ranger District, Fremont National Forest in south-central Oregon.
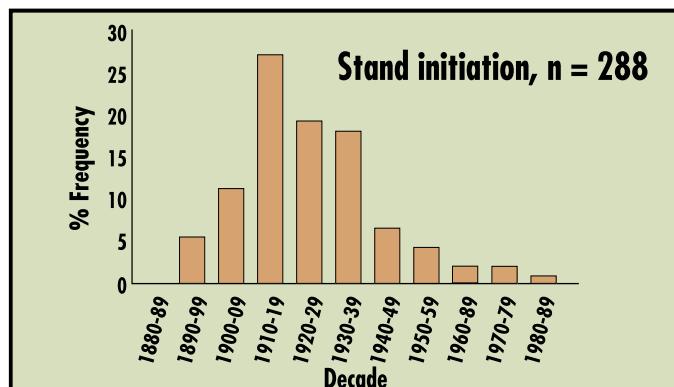
Figure 3d. When juniper encroachment began (based on the three oldest western juniper in the stand) in 96 aspen stands in southeastern Oregon, northeastern California, and northwestern Nevada. Chronologies are based on tree ring data.

1976, Miller and Rose 1999, Miller and Tausch 2001). Fire occurrence and fire size declined dramatically in the late 1800's. Miller and Rose (1999) reported a large decrease in fire occurrence in southeastern Oregon shortly after large numbers of livestock were introduced in the late 1860's (Fig. 3c). The lack of fire and decreased competition from herbaceous species probably contributed to an increase in shrub density and cover, thus providing a greater number of safe sites for western juniper establishment (Miller and Rose 1995, 1999). The role of livestock as a mechanism for western juniper seed dispersal appears to be minimal (Burkhardt and Tisdale 1976).

Atmospheric CO_2

Rising levels of atmospheric CO_2 seem to have enhanced the increase in woody species throughout the West (Johnson et al. 1993, Knapp and Soulé 1999b). Increases in atmospheric CO_2 levels do not coincide with the initial increase or peak periods of western juniper establishment (Table 2). However, elevated atmospheric CO_2 during the last half of the 20th century may be an important contributing factor accelerating tree canopy expansion and establishment in some areas (Knapp and Soulé 1996, 1998, 1999b; Soulé et al. 2004). Annual sapwood growth in western juniper has been significantly greater since the 1950's compared to prior years (Knapp et al. 2001a, b), suggesting accelerated growth. The authors suggest elevated CO_2 levels may have a drought-ameliorating effect by increasing water use efficiency.

Fire

Fire is considered to have been the most important factor in limiting conifer encroachment into shrub-grassland communities in the Intermountain West prior to European settlement (West 1999, Miller and Tausch 2001). However, only a few studies have documented fire regimes across shrub-steppe communities and woodlands throughout this region. Unlike ponderosa pine, junipers seldom repeatedly scar in response to fire; thus it is difficult to determine or describe presettlement fire regimes across many shrub-steppe and woodland communities. Fire scars on western juniper are occasionally found, but most presettlement trees do not grow on sites representative of more productive deeper-soil sites, which now support expanding post-settlement woodlands. Old-growth western juniper is commonly found on relatively fire-safe sites (i.e., rocky surfaces, shallow soils, limited effective moisture) characterized by low production with limited fine fuels (Burkhardt and Tisdale 1976; Vasek and Thorne 1977; Young and Evans 1981; Holmes et al. 1986; Miller and Rose 1995, 1999). Evidence that woodland expansion was limited by fire events prior to settlement includes: (1) sites supporting old-growth trees are usually fuel-limited,

(2) most young stands occupy the more productive communities where fine fuel loads could carry a fire, and (3) the time sequence of woodland expansion is synchronous with the decline in fire occurrence.

In productive mountain big sagebrush plant associations in the Northwest, such as those characterized by Idaho fescue (*Festuca idahoensis*), MFRIs (mean fire return intervals[8]) typically ranged between 10 to 25 years (Table 3) and large fires every 38 years. Potential natural vegetation resulting from these short fire return intervals would probably have been dominated by Idaho fescue with an open, scattered canopy of mountain big sagebrush. MFRIs were determined from fire scars collected on ponderosa pine or Douglas-fir (*Pseudotsuga menziesii*) growing in or adjacent to mountain big sagebrush communities (Fig. 5). In two studies, where presettlement MFRIs were 12–15 years, fire-free intervals varied between 3 and 29 years (Gruell 1999, Miller and Rose 1999). However, fire occurrences were less frequent in the more arid plant associations in the mountain big sagebrush alliance. Based on tree growth, age structure, and the scarcity of presettlement trees or the presence of large dead wood, the maximum MFRI in the mountain big sagebrush/Thurber needlegrass (*Stipa thurberiana*) plant association was probably 50–70 years. Fire return intervals up to 50 years were probably adequate to limit western juniper encroachment into the mountain big sagebrush alliance (Burkhardt and Tisdale 1976, Miller and Rose 1999). The probability that western juniper will establish and successfully mature greatly increases

[8]MFRI = arithmetic average of the number of years between fire events determined for a designated area during a designated time period.

Figure 4. Conceptual model illustrating factors influencing the expansion of western juniper since the late 1800's and throughout the 1900's (Miller and Tausch 2001).
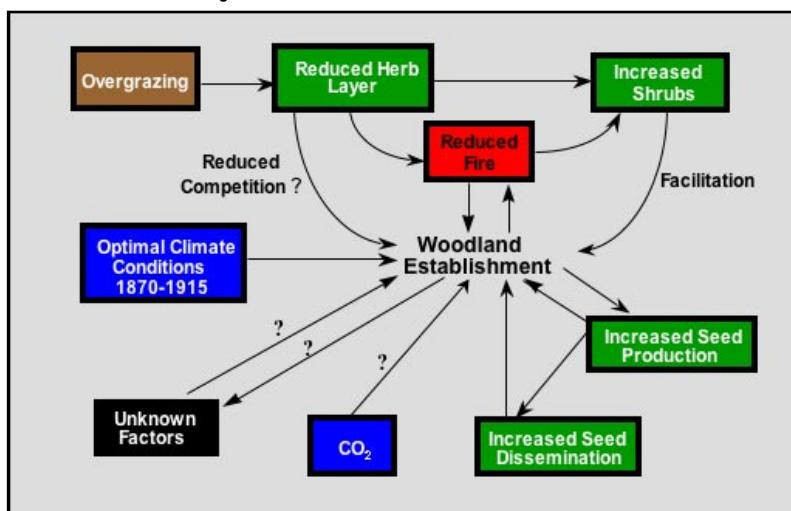
Table 3. Presettlement mean fire-return intervals (MFRI = average number of years between fire events) in sagebrush and aspen cover types associated with western juniper. Change indicates the decade when the MFRI increased (Miller and Tausch 2001).

Plant Association	MFRI (yrs)	Decade of change	Location	Reference
Mountain big sagebrush/				
Idaho fescue	20	Late 1800's	Lava Beds National Monument, CA	Martin & Johnson 1979
	11	1910	Owhyee Mt, ID	Burkhardt & Tisdale 1976
Idaho fescue	12–15	1870's	Chewaucan–Paisley, OR	Miller & Rose 1999
Idaho fescue	13	Late 1800's	Hart Mt, OR	Gruell 1999
Idaho fescue	13–15	1870's	Pine Mt, OR	Miller et al. 2001
Idaho fescue	16	1860's	Summer & Silver Lake, OR[1]	Miller et al. 2001
Idaho fescue	12	1880's	Fort Rock, OR[1]	Miller et al. 2001
Idaho fescue	17	1870's	Devils Garden, CA	Miller et al. 2001
Idaho fescue	6	1870's	Silver Lake, OR[1]	Miller et al. 2001
Idaho fescue	16.5		Silver Lake northwest, OR[1]	Miller et al. 2001
Idaho fescue, with some ponderosa pine	8–10	1870	Lava Beds National Monument, CA	Miller et al. 2003
Western Juniper/				
western needlegrass	150+		Lava Beds National Monument, CA	Miller et al. 2003
Low sagebrush/				
Not reported		1860	northwestern CA	Young & Evans 1981
Sandberg bluegrass	138	1870	Chewaucan–Paisley, OR	Miller & Rose 1999
Aspen	60[2]	1870's	eastern OR, northeast CA, & northwest NV	Wall et al. 2001

[1]General location of stand studied.
[2]Stand replacement interval based on aspen age structure, disturbance may not be fire.

Figure 5. Ponderosa pine with over a dozen pre-1900 fire scars in a densely wooded community at Lava Beds National Monument, northeastern California. The pre-1900 plant community was an open ponderosa pine stand with an understory dominated by Idaho fescue. The mean fire return interval was between 8 t   y of mountain mahogany soon to be overtaken by western juniper (greater than 100 trees/acre).
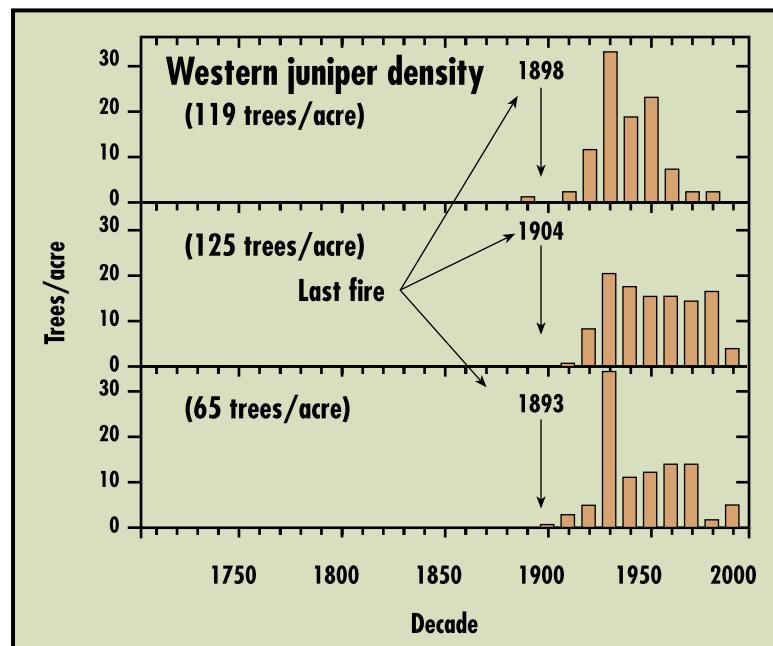
as MFRIs become more than 70 years. A fire free period of more than 70 years will also increase the potential for leaving large-diameter charred wood consisting of heartwood (that can persist on the site for more than 100 years), resulting from the development of mature trees on the site. Small trees consisting of mostly sapwood usually decompose in several decades. In northern California, a plant community identified as a western juniper/ mountain big sagebrush/western needlegrass (*Stipa occidentalis*) plant association burned in 1856 (Miller et al. 2003). Intact charred wood and fire-killed trees are still present on the site. On Juniper Mountain in eastern Oregon, trees killed by fire in 1717 still persist in the stand.

A number of studies in mountain big sagebrush communities in the Intermountain West have reported significant declines in fire events since the late 1800's (Table 3) (Miller and Tausch 2001). Several studies have shown a close relationship between the early expansion of western juniper in the late 1800's and the sudden decline in fire occurrences in the mountain big sagebrush alliance (Figs. 3c, 6) (Miller and Rose 1999; Miller et al. 2001, 2003).

MFRIs reported for the low sagebrush/Sandberg bluegrass (*Poa sandbergii*) association (Table 3) were considerably longer than for neighboring mountain big sagebrush communities (Young and Evans 1981, Miller and Rose 1999). Fire-free periods of 90 (Young and Evans 1981) and 138 years (Miller and Rose 1999) were reported for this plant association in northern California and south-central Oregon and it is not unlikely that fire-free periods exceeded 150 years for some sites. This plant association can be characterized by a low density of widely scattered old-growth western juniper, which suggests infrequent fires. Tree growth rates are relatively slow with the average age of a 3-m-tall tree ranging from 75 to 90 years. Fire return intervals of 100 to 150 years would probably be adequate to maintain a low-density stand of widely scattered trees in this plant association. In the absence of fire, western juniper will slowly increase in density in this plant association.

Fire also played an important role in the maintenance of healthy mixed-age aspen stands in the semi-arid West (Bartos and Campbell 1998). In the northwestern Great Basin, Wall et al. (2001) reported that encroachment of western juniper into these communities began around 1900. Fire was probably the primary disturbance factor limiting western juniper invasion into these aspen communities. Based on the composition and distribution of age of aspen in two large stands in southeastern Oregon, presettlement mean

Figure 6. Tree densities and age chronologies of western juniper and date of last fire for three cinder buttes sampled on the Lava Beds National Monument, northern California. The lack of large dead juniper wood on these sites suggests mature juniper did not occupy these sites prior to 1900. Potential natural vegetation is Idaho fescue grassland with widely scattered ponderosa pine and mountain big sagebrush and mean fire return interval prior to 1900 was 8–10 years (Miller et al. 2003).

disturbance intervals were determined to be 16 years within portions of these stands. Wall et al. (2001) estimated that total stand replacement in these two aspen communities occurred around 60–100 year intervals.

Climate and fire

In eastern Oregon, large presettlement fires in sagebrush-steppe communities were usually preceded by at least one year of above-average growing conditions (Miller and Rose 1999). In these semi-arid ecosystems, fuels are often limited in abundance and continuity. A series of wet years allows fuels to accumulate and become more contiguous. Wetter than average conditions in the late 1800's would have resulted in the accumulation of fine fuels. However, high livestock stocking rates and season-long or heavy grazing during this period reduced fine fuel accumulations and thus significantly decreased the potential for fire (Burkhardt and Tisdale 1969, Miller and Rose 1999). The combination of reduced fire occurrences (Miller and Tausch 2001) and optimal climatic conditions for conifer establishment (Fritts 1974, Fritts and Wu 1986) at the turn of the century were probably the two dominant factors that initiated post-settlement western juniper expansion.

Life History and Biology

Western Juniper Varieties

Morphology of western juniper

Western juniper is submonoecious[9]. Trees are pyramidal to round in shape and typically reach 13–32 ft in height at maturity, but will occasionally reach 65 ft in height. Trunks are usually composed of a single erect stem 13.7–27.6 inches in diameter (maximum of 74.8 inches) (Vasek 1966, Cronquist et al. 1972). The largest reported western juniper, located in the Lost Forest in northern Lake County, Oregon, is 78 ft tall with a trunk circumference of 19 ft. Bark is typically gray but can turn reddish in some old trees (more than 300 years). Mature western juniper leaves are 0.039–0.118 inches in length, compressed to the stem and overlapping the next leaf (Fig. 7). Leaves occur as opposite pairs or in whorls of three. Each scale has a conspicuous resin gland on the dorsal side of the leaf (Fig. 7). In contrast, juvenile leaves are not compressed to the stem and are spiny tipped. Seed bearing can begin as early as 10–20 years of age, but significant fruit production usually starts at 50–70 years of age (Miller and Rose 1995). The yellowish-brown male cones are 0.12–0.16 inches long and occur at the end of a branchlet (Fig. 7). Male cones develop during the late summer and early fall and shed their pollen early the following spring (Vasek 1966). Female cones are bluish to bluish-black at maturity, covered with a resinous pulp, and contain two to three seeds (occasionally one seed). These cones begin development in early spring, attain full size the first summer and mature during the second summer. Female cones persist on trees for nearly 2 years. Morphological characteristics of western juniper and Utah juniper are usually distinct. Utah juniper lacks the resin gland on the back of the leaf scale and the female cones are brownish with a mealy to fibrous covering (Cronquist et al. 1972). However, in northwestern Nevada, where the distribution of the two species overlap, differences become less apparent due to hybridization (Vasek 1966, Terry et al. 2000).

Morphology of Sierra juniper

Sierra juniper, a variety of western juniper, is located primarily south and southeast of the range of western juniper. Sierra juniper is distinguished from western juniper in that it is mostly dioecious[10], has reddish-brown bark rather than gray bark (Cronquist et al. 1972), can attain a larger size at maturity, and grows in different plant associations, higher elevations, and different climatic conditions. However, the bark on older western juniper trees also often attains a reddish color. Charlet (1996) reported that Sierra juniper material collected in Nevada was distinct from western juniper and suggested a taxonomic reevaluation of the variety. Further, the largest recorded Sierra juniper is 83 ft tall and 40 ft in trunk circumference, located in the Stanislaus National Forest, east-central California.

Seed Production, Dissemination, Germination, and Establishment

Although seed production occurs in most years (Sowder and Mowat 1958), western juniper seed-crop production is highly variable across sites and years. The environmental variables that trigger the initiation of male and female cones have not been identified. Research on factors influencing seed production and seedling establishment will be required to predict annual seed-crop production and better understand woodland dynamics (Chambers et al. 1999b).

Figure 7. Western juniper male cones and foliage showing white dried resin exuded from the resin gland located on the dorsal side of the leaf scale.

[9]Submonoecious—male and female cones are borne on the same individual; however, some trees will produce predominantly male or female cones.

[10]Dioecious—male and female cones are borne on different individuals.

Seed dispersal of western juniper occurs through gravity, overland flow, and by animal transport. At least 12 species of birds feed on the fruits and as a group are the most important disseminator of western juniper seed (Fig. 8) (Gabrielson and Jewett 1940, Maser and Gashwiler 1978). American robins *(Turdus migratorius)* and Townsend's solitarie *(Myadestes townsendi)* often winter in woodlands and consume the female cones (Lederer 1977, Podder and Lederer 1982, Reinkensmeyer 2000). Townsend's solitarie can consume over 80 female cones/day. Mountain bluebirds *(Sialia currucoides)*, cedar waxwings *(Bombycilla cedrorum)*, and Steller's *(Cyanocitta stelleri)* and western scrub-jays *(Aphelocoma californica)* have been observed consuming female cones. Most birds have limited gut-retention times and fly short distances to perch and process the fruit, thus limiting the distance of most seed dispersal (Schupp 1993, Chambers et al. 1999b). After feeding on Ashe juniper *(Juniperus ashei)* fruits, American robins flew an average distance of 145 ft to a post-foraging perch, which could be another tree, shrub, or on the ground beneath a woody canopy (Chavez-Rameriz and Slack 1994). In Spain, Santos and Telleria (1994) reported birds feeding on juniper berries were more likely to visit large stands of trees and less likely to feed in small isolated juniper stands. Coyotes *(Canis latrans)*, cottontail rabbits *(Sylvilagus* sp.), and several rodent species also consume and scatter western juniper seeds (Chambers et al. 1999a). Mule deer *(Odocoileus hemionus)* have also been observed to eat western juniper fruits during winter months when preferred foods are unavailable (Leckenby 1968, Trout and Thiessen 1968). However, successful establishment of seed dispersed by mammals is probably limited, because seeds are deposited at high densities in microsites where establishment is poor (Schuppe 1993; Schuppe et al. 1997; Chambers et al. 1999a, b).

Western juniper seeds are initially dormant immediately following seed drop (Johnson and Alexander 1974). Germination potential is greatly enhanced by prolonged cool-moist stratification, which is cumulative from year to year (Young et al. 1988). This suggests germination of a particular seed crop may span several years. Seeds of several other juniper species are also long lived with an extended dormancy, resulting in highly persistent seed banks (Chambers et al. 1999a, b).

Little information is available on percent seedling survival or climatic conditions that influence seedling establishment. However, two studies indicate survival rates for western juniper seedlings are high (Burkhardt and Tisdale 1976, Soulé et al. 2004). In addition, Soulé et al. (2004) reported that wet cool summers may lower western juniper seedling mortality.

Much of successful western juniper seedling establishment occurs beneath shrubs (Burkhardt and Tisdale 1976, Eddleman 1987, Miller and Rose 1995, Soulé and Knapp 2000, Soulé et al. 2004). This may be attributed to a disproportionate amount of seed dropped by perching birds and/or more favorable growing conditions beneath the shrubs compared to the interspace. Growth rates of young trees beneath mountain big sagebrush canopies were greater (1.34 inches/year) than in the interspace (0.95 inches/year) (Miller and Rose 1995). Compared to bare soils in the interspace, soils beneath a sagebrush canopy can have nearly twice the moisture content and nitrification (Roberts and Jones 2000). Cooler temperatures and higher relative humidity beneath the sagebrush canopy also provide more favorable growing conditions for juvenile foliage, which has poorer stomatal control and lower water use efficiency than adult foliage (Miller et al. 1992). Safe microsites that modify the environment may be responsible for greater seedling survival rates under relatively dry conditions. Many seeds also germinate beneath the tree canopy; however, survival and growth rates are low because of high intraspecific competition from the overstory tree. No evidence suggests competition from associated shrubs or herbs limits the success of western juniper seedling establishment (Burkhardt and Tisdale 1976, Miller et al. 2000). However, an increase in bare ground and mature western juniper cover was negatively correlated with successful tree establishment across the mountain big sagebrush alliance in Oregon and California (Miller et al. 2000). This may be the result of intraspecific competition from overstory trees and limited safe sites for seedling establishment as woodlands approach late successional stages.

Figure 8. Mountain bluebirds consuming juniper berries early in the spring.
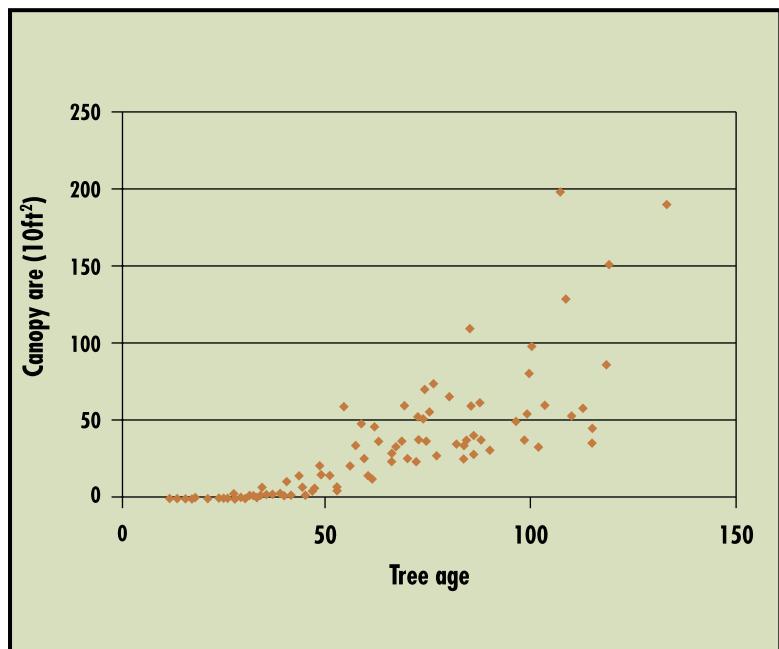
Photo by Rick Vetter.

Figure 9. Juniper roots in deep soils can have both large lateral and fine roots in the upper 24 inches, and a deep taproot. Taproots are often missing in shallow soils.

Photo by Hugh Barrett.

Figure 10. Canopy area and age for individual western juniper trees showing an increase in canopy expansion for trees over 45 years of age (from Miller and Tausch 2001).

Roots

During the first 10 years of growth, western juniper directs most of its effort into developing a taproot with only limited lateral root development (Kramer 1990). After 10 years, lateral root development increases, accounting for about 65 percent of the root biomass in trees 30–35 years old. Root: shoot ratios for young trees vary from 0.55 to 0.76 (Miller et al. 1990). Taproot development declines as trees begin to lose their juvenile foliage (Young et al. 1984) on shallow soils. However, taproots have been observed on some sapling and mature trees growing in deep soils (Fig. 9). Trees develop a massive fine root mat system with age. Young et al. (1984) reported most of tree roots were located in the upper 30 inches of the soil profile in a soil that is 40 inches deep. Large lateral roots commonly extend a distance that equals the height of the tree, but in some cases can extend as much as three times tree height.

Growth

Following germination, aboveground growth is relatively slow, averaging 1.18–1.58 inches/year in height for the first 10 years and increasing to 3.54–6.57 inches/year for older trees up to 100 years old (EOARC, unpublished data). Root development appears to far exceed aboveground growth during early development. Leaf canopy development remains relatively slow during the first 35–45 years. At the age of 45–50 years, the rate of tree canopy development increases (Fig. 10).

Current year's sapwood development begins during the spring and usually ends in early to late August, depending on the site and annual precipitation (Peter 1977). In wet years, ring growth can continue through August. Branchlet and leaf growth are greatest during June and July (Miller et al. 1992). Western juniper typically approaches its maximum height at 80–100 years of age across its geographic range (EOARC, unpublished data). Depending on site potential and competition from other trees, mean height of western juniper at 80 years of age will vary from 19.7 to 49.2 ft (Gedney et al. 1999). Site index curves that describe tree age and height relationships for western juniper varies widely across sites. Height for trees 80 years old at breast height (approximately 4.25 ft above the ground) ranged from 15 ft on scablands to 35 ft on sites associated with ponderosa pine (Sauerwein 1982). In central Oregon, mean height growth rate varied from 3.5 to 6.6 inches/year for dominant trees (Eddleman 1987). Several authors have developed regressions estimating western juniper leaf area, leaf biomass, and total standing crop using tree basal and sapwood areas (Gholz 1980, Miller et al. 1987). In a fully developed woodland in eastern Oregon,

Gohlz (1980) estimated foliage biomass of 4,550 lb/acre, total standing crop biomass of 23,300 lb/acre, and a leaf area index (LAI) of 2 (2 units of leaf area to 1 unit of ground area) for stands with a mean density of 608 trees/acre). Primary production was 1,200 lb/acre, about half that of adjacent ponderosa pine communities and 10 percent of Douglas-fir communities in the Cascades.

Leaf Morphology

Young western juniper trees (usually less than 25 years) have needle-like leaves, which are different than leaves on older trees (De Laubenfels 1953). Leaves on mature trees are triangular with minutely serrated margins and have a low surface-to-volume ratio (Fig. 11a) (Miller and Schultz 1987). Leaf margins are slightly cupped, which seals one leaf against the other and forms a chain-like cylinder. The leaf epidermis is heavily cuticularized (waxy covering on the leaf surface), which greatly reduces water loss through the leaf surface. Most of stomates are located on the protected side of the leaf surface facing the stem (Fig. 11c). Stomates on the outer surface are located at the base of the leaf and are covered by the adjacent subtending leaf (Fig. 11b). The leaf morphology of western juniper allows for maximum drought avoidance through low leaf area, low surface-volume ratios, thick cuticle layer, and protected stomata. Mean maximum leaf conductance (transpiration, measured as inches of water/second passing through the leaf surface to the atmosphere) per unit leaf area was lower (0.03–0.05 inches/second) than values reported for several other conifer species (0.05–0.16 inches/second) (Miller and Schultz 1987).

Water Use and CO_2 Assimilation

Ecophysiological (Moore et al. 1999) and morphological (Miller and Shultz 1987) adaptations allow western juniper to tolerate relatively large environmental changes. In addition, allocation of resources in young trees partially explains the species ability to compete successfully with other native species (Miller et al. 1990). By reducing allocation of resources to branches and trunks, juvenile and small adult western juniper allocate larger portions of dry mass to foliage and roots to optimize photosynthetic capacity and uptake of water and nutrients than mature trees.
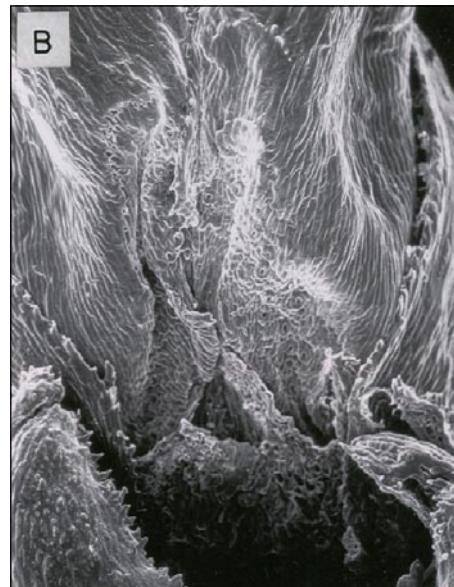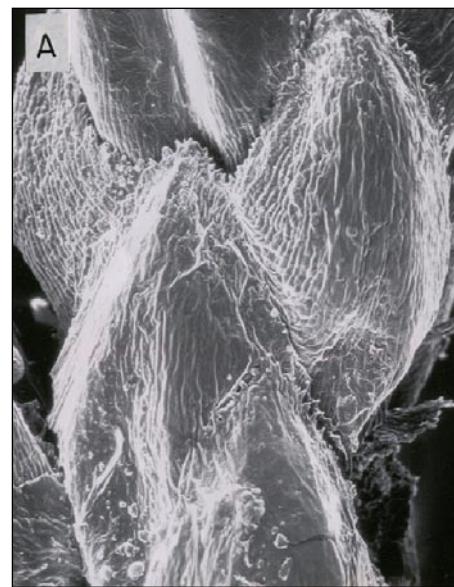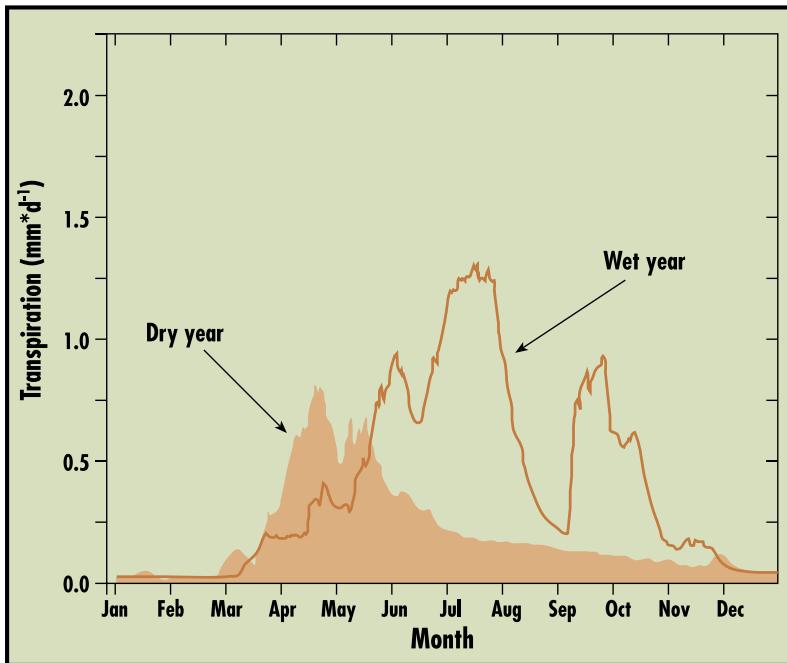
Figure 11. Leaf scales on the top (abaxial) [A and B at 150x] and lower (adaxial) [C at 250x] surfaces of western juniper. Most of the stomata are located on the adaxial leaf surface and a few are at the base of the abaxial surface, which is covered by the lower overlapping scale (Miller and Schultz 1987).

During winter, cold soil temperatures limit water use by western juniper (Miller and Schultz 1987). As soil temperatures drop below 40°F, water uptake at the root surface significantly decreases. As soil temperatures increase in March, trees begin to actively transpire and grow. In warmer climates, such as in the John Day or Mazama ecological provinces, more moderate soil temperatures may allow western juniper to transpire water during any month in the winter. In central Oregon, Jeppesen (1977) reported greater winter soil water loss at 20-inch soil depth in woodlands compared to thinned stands. Leaf conductance is strongly influenced by soil temperature and vapor pressure deficit during the spring. During the summer, soil water availability and vapor pressure gradient[11] are the primary factors influencing water use and CO_2 assimilation in western juniper (Miller et al. 1992, Angell and Miller 1994, Moore et al. 1999). Stomata closed when stem water potentials decreased to –2.0 MPa (mega pascals) (Miller and Schultz 1987). In a dry year, the greatest amounts of water were transpired during April and May, compared to June and July in a wetter-than-average year (Fig. 12) (Angell and Miller 1994). In a moderately

[11] The water vapor concentration gradient from inside the stomata to the open atmosphere.

Figure 12. Modeled daily transpiration for western juniper during a drought year (1990) and wet year (1984) (from Angell and Miller 1994).

stocked stand of 30 trees/acre and 1.6 LAI, the water-use model predicted western juniper would extract 2 inches of soil water in a dry year and 5.6 inches in a wet year. These predictions suggest soil water depletion rates will significantly shorten the growing season on the site, a point confirmed by Bates et al. (2000). They reported the growing season of the understory was shortened by as much as 6 weeks in uncut western juniper stands, compared to adjacent cut stands.

Juveniles with the awl-shaped leaves have higher leaf conductance, transpiration, and greater total CO_2 assimilation per unit of leaf weight during the growing season than sapling and mature trees (Miller et al. 1992). The change from juvenile to mature foliage reduces the amount of carbon assimilated per unit leaf area but also reduces the amount of water lost to transpiration by 40 percent (Miller et al. 1993).

Insects

Artichoke-like galls located on the branchlets of western juniper (frequently misidentified as reproductive structures) are formed by midge larvae *Walshomyia* spp. (Purrington and Purrington 1995)(Fig. 13). Moth larvae *Heinrichiesa sanpetella* were found to inhabitat 40 percent of these galls, over-wintering and pupating there in early spring. Other moth caterpillars that feed on western juniper are the sequoia sphinx (*Semiothisa* spp., *Sphinx sequoiae*), cedar streak *(Lithophane logior)*, and *Mitoura grynes barryi* (Miller 1995). Other insects known to feed on western juniper include long-horned beetle *(Styloxus bicolor)*, juniper bark beetle *(Phloeosinus serratus)*, round-head borers *(Callidium califonrnicum* and *C. juniperi)*, wood-boring beetle *(Melonophila miranda)*, and grasshoppers *(Melanoplus* sp.).

The western juniper bark beetle is typically attracted to wounded or felled trees (personal communication, Jane L. Hayes, USDA US Forest Service Research Station, La Grande, OR). Insect attacks usually do not result in the killing of live trees, however in the 1920's and 1930's in addition to drought, areas of western juniper were killed by insects in central Oregon (Furniss and Carolin 1977). Current work has identified 25 species of bark and woodboring beetles feeding on western juniper (Hayes, unpublished work in progress).

During the grasshopper outbreaks near the John Day Fossil Beds in eastern Oregon in the late 1970's, the tops of some western juniper trees were nearly totally defoliated. On dead or dying juniper, round-

head borers or long-head beetles (*Creambydidae* spp.) deposit eggs in the bark (Swan 1996). Upon hatching, the larvae bore into the wood, deriving nourishment from the soluble carbohydrates in wood particles and/or fungal tissue.

Associated Nonvascular Plants

Mistletoe

Mistletoes that commonly infect western juniper are juniper mistletoe (*Phoradendron juniperinum*), and dense mistletoe *(P. densum)* (Geils et al. 2002). Juniper mistletoe is the primary species found on western juniper and is the most widespread mistletoe infecting juniper species throughout the West. Juniper mistletoe has leafless stems and pinkish-white colored berries about 0.16 inches in diameter. Dense mistletoe occurs in the southwestern range of western juniper. This species has white- to straw-colored berries 0.16 inches in diameter and is easily differentiated from juniper mistletoe in that it has leaves. Birds feed on the fleshy mistletoe berries and are the primary dispersers of the sticky seeds. Birds that commonly feed on the berries include American robins, Townsend's solitaires, cedar waxwings, flycatchers, mountain bluebirds, and thrushes (Sutton 1951). The mistletoe foliage is high in nutritional value (Urness 1969). Juniper mistletoe usually occurs in a patchy distribution with only a few heavily infected trees. Although it can stress the tree by absorbing relatively large amounts of water and nitrogen, the tree is rarely killed.

Mosses, fungi, and lichens

Limited information is available on the ecology and life histories of nonvascular plants associated with western juniper. We also know very little about the effects of western juniper expansion or removal on biological crusts. *Tortula ruralis* is commonly associated with mature western juniper trees where it grows beneath the tree canopies. Four species of wood-rotting fungi, *Antrodia juniperina, Pyrofomes demidoffii, Diplomitoporous rimosus,* and *Phellinus texanus,* may cause heart rot in western juniper (Knapp and Soulé 1999a). These fungi

typically enter openings in the heartwood or in dead sapwood. Knapp and Soulé (1999a) reported a widespread occurrence of heart rot (suspected to be *Antrodia juniperina)* between 1730 and 1749 in western juniper across eastern Oregon and northeastern California. Heartwood rot is most commonly found in trees more than 150 years old (EOARC, unpublished data). Western juniper roots can be infected with symbiotic fungi mycorrhizae (Trappe 1981). Roberts and Jones (2000) reported higher levels of vesicular-arbuscular mycorrhiza fungi under western juniper canopies than under sagebrush or grass canopies.

Two species of foliose lichens commonly associated with western juniper are *Letharia columbiana* and *L. vulpina.* These lichens are brilliant fluorescent yellow-green or chartreuse in color, and highly branched. Both species are nearly identical in form except that *L. vulpina* lacks the small disk-like fruiting bodies (soredia). Both species can occur on a single tree and are often most abundandant on dead, barkless branches or snags.

Figure 13. Artichoke-like gall located on the branchlet of a western juniper (frequently misidentified as a reproductive structure) is formed by midge larvae Walshomyia species.

Ecology

Soils

Western juniper grows on a wide variety of parent materials and soils (Driscoll 1964a). Parent materials are derived from aeolian (e.g., pumice sands), sedimentary, and igneous (e.g. rhyolite, andesite, basalt) sources. Soil textures can range from heavy clays to sandy soils. Soil depths vary from bare rock to more than 3 ft, and soil temperature regimes are mesic and frigid[12] (limited cryic[13]). Western juniper roots are able to penetrate fractured basalt bedrock, allowing it to occupy rock outcrops and soils less than 12 inches deep. The wide range of soils has a large impact on potential erosion, woodland development, overstory-understory interactions, and response to disturbance across the range of western juniper.

Western Juniper Communities

Numerous classifications have been proposed for western juniper plant associations[14] and communities (Driscoll 1964a, b; Hall 1978; Hopkins 1979; Johnson and Clausnitzer 1992). In addition, the Natural Resource Conservation Service (NRCS) is developing ecological site classification with western juniper in the plant association name. However, it is not always clear in these classifications if western juniper was a part of these communities prior to European settlement or has encroached since settlement. Western juniper communities may be separated into presettlement (old-growth) or post-settlement (expansion) communities. We suggest 1870 as a cut-off to separate the two age classes. The date separating pre- and post-settlement is based on the approximate time when fire regimes changed (1870's) and the arrival of livestock in eastern Oregon, southwestern Idaho, and northeastern California (late 1860's) (Oliphant 1968, Miller et al. 1994, 1999a).

Common associated diagnostic species

Western juniper is associated with a wide range of plant communities including forest, riparian, aspen, and shrub-steppe. Within these community groups, it has actively expanded into numerous plant alliances and associations defined by ponderosa pine, aspen, willow *(Salix* spp.), mountain big sagebrush *(Artemisia tridentata* ssp. *vaseyana)*, Wyoming big sagebrush *(A.t.* ssp. *wyomingensis)*, basin big sagebrush *(A.t.* ssp. *tridentata)*, low sagebrush, stiff sagebrush *(A. rigida)*, bitterbrush *(Purshia tridentata)*, and mountain mahogany *(Cercocarpus ledifolius)*. Scattered stands of western

juniper in Siskiyou, Trinity, Shasta, and west Lassen counties in California are associated with Oregon white oak *(Quercus garyana)*, buckbrush ceanothus *(Ceanothus cuneatus)*, and several other conifer species (Vasek and Thorne 1977). Common understory diagnostic species are Columbia needlegrass *(Stipa columbiana)*, needle-and-thread needlegrass *(S. comata)*, western needlegrass, Thurber needlegrass, Idaho fescue, bluebunch wheatgrass *(Agropyron spicatum)*, Ross sedge *(Carex rosii)*, and Sandberg bluegrass.

Old-growth (pre-settlement) communities

It is estimated that less than 10 percent of existing western juniper individuals established prior to the 1870's (USDI–BLM 1990, Miller et al. 1999a, Johnson 2005). However, the proportion of old-growth varies across ecological provinces and few presettlement stands have been inventoried or separated out from post-settlement stands. Old-growth western juniper is associated with a variety of soils, landforms, and plant associations throughout its range. Old-growth communities typically occupy rock outcrops and soils that are shallow, rocky, and often high in clay or sand, and fine-textured sedimentary soils. Examples include the shallow claypan soils occupied by low sagebrush and Sandberg bluegrass common in the High Desert, Klamath, and Humboldt ecological provinces. However, it is also associated with the ashy-sandy-pumice soils associated with mountain big sagebrush, western needlegrass, and needle-and-thread grass in the Mazama Province and sedimentary soils in the John Day province. The common factor linking this wide array of soils and landforms that support old-growth stands

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[12]*Mesic:* a soil temperature regime that has mean annual soil temperature of 46-59° F, and more than 43° F difference between mean summer and mean winter soil temperatures at 20 inches below the surface, or at a densic, lithic, or paralithic contact, whichever is shallower.

Frigid: a soil temperature regime with warmer summer temperatures than cryic, with mean annual soil temperatures less than 46° F, and more than 43° F difference between mean summer and mean winter soil temperatures at 20 inches below the surface, or at a densic, lithic, or paralithic contact, whichever is shallower.

[13]*Cryic:* a soil temperature regime that has mean annual soil temperature of 0° F but lower than 8° F difference between mean summer and mean winter soil temperatures at 20 inches below the surface at 20 inches below the surface, or at a densic, lithic, or paralithic contact, whichever is shallower.

[14]Plant association is defined by the dominant/diagnostic overstory and understory species (e.g., mountain big sagebrush/ Idaho fescue plant association).

is their low production potential, which limits the accumulation of fuels. Thus, fire events were typically limited to one or several trees, or stand replacement, and mixed-severity fire events were infrequent (more than 150 years). For a definition of old-growth western juniper woodland, see Appendix 1.

Single tree perspective

Old-growth is a relative term, and has been based on morphological characteristics, actual age, or general period of establishment (pre- and post-settlement). As trees age they change morphologically. Compared to younger trees, old trees have approached their maximum size, height growth has ceased, and the tree crowns may be in various stages of decline. As trees mature, their inverted-cone-shaped canopy becomes increasingly unsymmetrical in appearance with rounded tops and spreading canopies that may become sparse and contain dead limbs or spike tops (Figs. 14, 15). In addition, the bark on the trunk becomes deeply furrowed, fibrous (Fig. 16), and reddish in color. Bark on trees less than 150 years is scaly and furrows are shallow or lacking. Branches near the base may be very large (more common in open stands), and branches are covered with bright green arboreal fruticose lichens. The cambium layer (live wood tissue) may also die around portions of the tree trunk, leaving only a narrow strip connected to a single live branch. An additional characteristic that helps distinguish older trees is limited terminal leader growth on branches in the upper 25 percent of the tree canopy. Younger trees, between 80 to

Figure 14. An 800-year-old western juniper tree with spreading rounded top and large lower limbs, Connelly Hills, south-central Oregon.

Figure 15. Old-growth western juniper/low sagebrush/Sandberg bluegrass plant association, occupying a shallow heavy clay soil on the Modoc Plateau in northern California.

Figure 16. Bark characteristics of three different aged trees: At 75 years, bark is thin and flaky; at 152 years, bark layer is thickening and beginning to develop vertical furrows; and at 270 years, bark is thick, fibrous with well-developed furrows.

Figure 16a. A 75-year-old tree.

Figure 16b. A 152-year-old tree.

Figure 16c. A 270-year-old tree.

130 years, typically have terminal branch leader growth ranging between 2 to 6 inches in the upper portion of the canopy. Many of these traits that separate old and young trees usually begin to develop at 150 (± 30) years of age but can vary across different sites. For example, on Green Mountain in northern Lake County, Oregon, trees older than 200 years still retain symmetrical inverted-cone-shaped canopies. However, terminal and lateral leader growth in the upper canopy was less than 0.5 inches and vertical furrows in the bark were present.

Community perspective

Old-growth is usually defined at the community level based on structural components that are easily identifiable. In the absence of major disturbance, structural characteristics that increase over time include morphological characteristics of old trees, standing and down dead, canopy decadence (dead branches), abundance of lichen in the tree canopies, hollows, and cavities. For example, a stand that is 150–400 years old generally has little standing or down dead wood. However, as stands

Figure 17. An old growth juniper growing on a rocky ridgetop with a young post-settlement stand in the background on Steens Mountain, Oregon.

Figure 18. Old-growth western juniper woodland in the Mazama Ecological Province.

mature to over 500 years, standing and down dead wood in the community accumulates due to slow decomposition rates. Structure of the tree layer in old-growth western juniper communities (e.g., percent cover, tree density, size, etc.) will vary with site conditions and the history of past disturbances. Most old-growth stands can be separated into three general categories based on stand structure: (1) isolated stands of one to several trees located on rocky outcrops and ridges (Fig. 17); (2) low sagebrush grasslands with widely scattered trees (i.e., savannas, Fig. 15), and (3) woodlands with tree canopy cover typically less than 20 percent (Fig. 18), but occasionally exceeding 35 percent. Old-growth savannas probably account for the largest land area of old-growth but tree densities are usually very low. Trees are widely dispersed and primarily shrubs and herbs influence the tree interspace, with little interference from western juniper roots. A typical example is a low sagebrush-Sandberg bluegrass tableland (Fig. 15). Old-growth woodlands are defined as stands where tree root competition is dominant over shrubs and dominant or co-dominant over grasses in extracting resources in the tree canopy interspace. The most extensive area of old-growth woodland occurs in the aeolian sands in the Mazama Province and the northwestern edge of the High Desert Province. Composition and structure of these old-growth communities varies widely across the range of western juniper. Mean density of overstory trees varies from 80 trees/acre in the pumice region (Waichler et al. 2001), 96 trees/acre in southwestern Idaho (Burkhardt and Tisdale 1969), and 146 trees/acre on Juniper Mountain in southeast Oregon (EOARC, unpublished data). Cover of tree canopies ranges from 10 to 60 percent on these sites.

Old-growth types

Three primary regions of old-growth can be differentiated by soils derived from three different parent materials: igneous, sedimentary, and aeolian soils. The three types usually differ in community structure and composition.

Igneous soils. Soils derived from igneous parent materials dominate much of the landscape in the High Desert, Klamath, southwestern portion of the Snake River, and the Owyhee Plateau region in the Humboldt ecological provinces. In these provinces, old-growth western juniper typically grows in widely spaced stands on shallow, rocky, heavy clay soils, or rock outcrop, which support limited fuels to carry fire (Fig. 15, Fig. 17) (Vasek and Thorne 1977, Miller and Rose 1995, West 1999). Old-growth western juniper is estimated to make less than 10 percent of the western juniper population across this region (Miller et al. 1999a, Johnson 2005). The low sagebrush claypan communities probably account for the greatest land area occupied by old-growth western juniper across these provinces. The dominant grass in these low sagebrush

tablelands is Sandberg bluegrass, with Idaho fescue frequently growing beneath the tree canopy. These communities often occupy extensive flats with slopes typically less than 5 percent, but sometimes approaching 30 percent. The rocky, shallow heavy clay soils originate from ancient volcanic flows of basalt, andesite, and rhyolite. Although soils are shallow (less than 20 inches) on these tablelands or nearly non-existent on rock outcrops, western juniper roots are capable of penetrating fractured bedrock, usually basalt (EOARC, unpublished data).

Tree canopy cover on the low sagebrush clay-pan sites is highly variable and may approach 20 percent, but usually is less than 5 percent (EOARC, unpublished data). These communities are usually rich in herbaceous species with a high diversity of forbs. Trees are usually uneven aged. On the Devils Garden in northern California, 63 percent of the presettlement trees aged varied between 200 and 500 years old. More than 30 percent were older than 500 years. The oldest trees aged to date on igneous parent material ranged between 1,000 and 1,400 years old, and were located north of Fredrick's Butte in southeastern Deschutes County, Oregon. Low presettlement tree densities in these communities may be attributed to limited tree establishment, slow growth rates due to poor site conditions, and occasional fires. Mean fire intervals of 80–150 years were probably adequate to create a stand of widely scattered western juniper trees (Young and Evans 1981, Miller and Rose 1999), but single-tree lightening fires were more common occurrences across these western juniper-low sagebrush claypan communities. Tree densities in these communities have increased since the late 1800's (Young and Evans 1981; Miller and Rose 1995, 1999).

On deeper (more than 20 inches) igneous soils, fire limited the development of old-growth western juniper woodlands (Miller and Tausch 2001). These soils typically support mountain big sagebrush grassland communities. Mean fire intervals of 10–25 years occurred in the more productive plant associations of this alliance (Houston 1973, Burkhardt and Tisdale 1976, Martin and Johnson 1979, Miller and Rose 1999). However, Juniper Mountain, located east of Alkali Lake in Harney and Lake counties, Oregon is an exception. This site may serve as a model as to what the more productive mountain big sagebrush plant associations would look like if fire had played a minor role in the sagebrush ecosystem (Fig. 19). On the north and northeast aspects tree canopy cover ranged between 35 and 60 percent. On south and southwest aspects tree cover ranged between 25 and 40 percent. Preliminary work indicates the age of overstory trees ranged between 350 and 600 years (EOARC, unpublished data). Understory trees 3–5 ft tall were between 100 and 200 years old. Shrub cover accounted for less than

one percent of the understory cover. Dominant herbaceous species were Idaho fescue on the north aspect, Thurber needlegrass on the south aspect and bluebunch wheatgrass on the west aspect. In August of 2001, a stand-replacement fire occurred on the northeast aspect of Juniper Mountain.

Sedimentary soils. Little work has been conducted on old-growth western juniper on these soils. Most of these soils occupied by old-growth western juniper occur in the John Day Ecological Province with limited amounts occurring in other provinces. These soils usually support a low density of trees and a sparse understory incapable of carrying fire. The accumulation of both down and standing dead and decadent trees on many of these sites indicates the presence of very old stands. Dead trees may remain standing for hundreds of years. Old-growth also occurs on shallow rocky soils and rock outcrops in this province. Old-growth stands probably account for less than 5 percent of the western juniper woodland component in these provinces.

Aeolian soils. The aeolian soil region, primarily composed of pumice sands, is located in the Mazama and northwestern portion of the High Desert ecological provinces. This region supports extensive old-growth western juniper woodlands (Fig. 18). Although not inventoried, these woodlands are estimated to account for more than 10 percent of the area occupied by western juniper woodlands in the Mazama Province. These stands are characterized by very sandy pumice soils derived from the eruption of Mount Mazama, 7,600 years ago. In the northwestern corner of the High Desert Province, soils are mixtures of wind-blown sands from Pleistocene lakebeds and pumice from Mount

Figure 19. The largest and most dense old-growth western juniper woodland in the High Desert Province, located on Juniper Mountain about 4 miles southeast of Alkali Lake, Oregon. The dense north-face side of the mountain burned by a lightning-initiated stand-replacement wildfire in 2001.

Mazama and Newberry Crater. Stand structure varies across these provinces, but is generally open, with tree canopy cover typically ranging from 10 to 15 percent in central Oregon (Waichler et al. 2001). Density of live trees ranged from 15 to 25/acre, standing dead were usually less than or equal to 6/acre, and down dead ranged from 1 to 7/acres. At the Lava Beds National Monument in northern California, tree densities in old-growth stands were 50/acre. The oldest western juniper tree aged to date is 1,600 years old and is located in this pumice region on Horse Ridge in Deschutes County, Oregon (Fig. 20). Past fires in this zone were typically small, burning single to several trees within a stand. However, old fire scars on these landscapes indicate occasional, extensive stand-replacement fires did occur. In the Lava Beds National Monument in northeastern California, a large stand-replacement and mixed-severity fire occurred in old-growth

Figure 20. This 1,600-year-old western juniper tree is the oldest aged to date. Located on Horse Ridge in central Oregon, east of Bend.

stands. Western needlegrass and needle-and-thread are usually the dominant grasses characterizing stands with very old trees. However, bluebunch wheatgrass and Idaho fescue are occasionally the diagnostic understory species. Plant associations included in the old-growth western juniper type are usually very low in both forb diversity and abundance. In the Bend-Redmond area, which lies below 3,500 ft in elevation, rabbitbrush (*Chrysothamnus* spp.) and cheatgrass (*Bromus tectorum*) will dominate the understory on sites that have been overgrazed or mechanically disturbed.

Woodland Succession

Most plant communities occupied by post-settlement western juniper are in a transitional state, ranging from open stands of trees with a dominant understory of shrubs and herbs to mid- or late succession, where trees are beginning to dominate the site and tree canopies are approaching full coverage (Miller et al. 2000). It is important to identify the woodland transitional state in resource evaluations or inventories and when developing management strategies. The state of woodland development directly affects plant community structure, composition, seed pools, wildlife habitat, and ecological processes including hydrologic and nutrient cycles. The stage of woodland succession will also directly affect the selection of management treatment, response following treatment, follow-up management, and treatment cost. In addition, continued changes in structure and composition in developing woodlands over time should be considered when developing resource plans and setting management priorities.

Identification of the woodland stage of succession

We have separated woodland succession into three transitional phases (Fig. 21):
• Phase I, trees are present but shrubs and herbs are the dominant vegetation that influence ecological processes (hydrologic, nutrient, and energy cycles) on the site (Fig. 22a);
• Phase II, trees are codominant with shrubs and herbs and all three vegetation layers influence ecological processes on the site (Fig. 22b);
• Phase III, trees are the dominant vegetation and the primary plant layer influencing ecological processes on the site (Fig. 22c, d).

There are several characteristics that can be used to define the phase of woodland development, regardless of the plant association or site potential (Table 4) (Miller et al. 2000). These traits relate to the degree of western juniper dominance on the site. Early signs of western juniper domination on a site are canopy mortality of the shrubs in the interspace and the reduction of leader growth (Fig. 23) on sapling size (less than 10 ft tall) trees.

Rates of woodland development

The rate of woodland succession from initial encroachment to fully developed woodlands is a function of the rates of tree establishment and growth. There is a high degree of variability in woodland succession rates across and within plant associations. In eastern Oregon, all three transitional phases of western juniper stand development can be observed where encroachment began in the late 1800's (Miller and Rose 1999, Miller et al. 2000). The minimum time for the tree overstory to begin suppressing the understory is 45–50 years and to approach stand closure 70–90 years on cool wet sites (i.e. mountain big sagebrush/Idaho fescue and or Columbia needlegrass) (Fig. 24) (Johnson 2005). On warm dry sites (mountain big sagebrush/Thurber needlegrass and/or western needlegrass), stand closure occurs in 120–170 years. Dense fully developed post-settlement woodlands that initiated establishment in the late 1800's had reached Phase III by the 1950's and early 1960's, based on tree growth rates (Fig. 25). In closed stands in southwestern Idaho, a significant decrease in growth of annual tree rings occurred during the 1950's, suggesting the onset of intra-specific competition. In adjacent open stands of trees in Phases I and II, tree-ring growth did not decline.

The primary factor controlling the number of years between initial encroachment and stand closure is establishment rate of tree seedlings. This is largely determined by seed input and the abundance of safe sites for seedling establishment. There may be a lag period of tree establishment immediately following fire, because of the reduction in shrubs (Erdman 1970, Burkhardt and Tisdale 1976). Shrubs provide desirable microsites for tree establishment (Burkhardt and Tisdale 1976, Miller and Rose 1995) and perching sites for avian seed dispersers.

Stand structure in closed stands

Canopy cover and density of overstory trees at stand closure varies among and within plant associations (Table 5). The density of large dominant trees in fully developed woodlands can vary from as low as 32 trees/acre on dry sites to more than 500 trees/acre on cool moist sites. Height and basal diameters are usually smaller in the denser stands of western juniper. Tree densities can exceed over 500 trees/acre if subcanopy trees are included. In closed woodlands shrub cover is typically less than 1 percent on the drier sites. On sites with higher effective precipitation that support both wax current *(Ribes cereum)* and snowberry *(Symphoricarpos oreophilis)*, mean shrub cover is reduced to less than or equal to 5 percent in closed woodlands.

Understory dynamics

Shrubs

As western juniper begins to dominate a site, shrubs begin to decrease (Figs. 21, 26) (Burkhardt and Tisdale 1969, Adams 1975, Bunting et al. 1999, Miller et al. 2000, Roberts and Jones 2000, Schaefer et al. 2003). This has a significant impact on ladder fuels, ground- and shrub-nesting birds, seed pools, and structural complexity of the plant community. At a site near Silver Lake, Oregon, 71 percent of the trees established during 1900–1936 (Adams 1975). The rapid decline in bitterbrush and sagebrush on these sites began in 1948. In the John Day Province near Prineville, Oregon, shrub cover in untreated western juniper plots was 0.4 percent compared to 9.4 percent cover in adjacent plots cut 18 years earlier (Eddleman 2002d). The decline in mountain big sagebrush is not proportional to the increase in western juniper. As western juniper approaches 50 percent of maximum potential, cover of mountain big sagebrush declines to about 20–25 percent of maximum potential (Miller et al. 2000). Tausch and West (1995) also reported a disproportionate decline; shrubs declined to one-fourth of maximum when single-leaf piñon *(Pinus monophylla)* and Utah juniper cover reached 50 percent of maximum in Nevada.

Figure 21. A conceptual model illustrating the relationship between shrub canopy cover, tree canopy cover, relative growth rates (i.e., ratio of annual ring width:mean ring width), and management strategies during the three phases of woodland development.
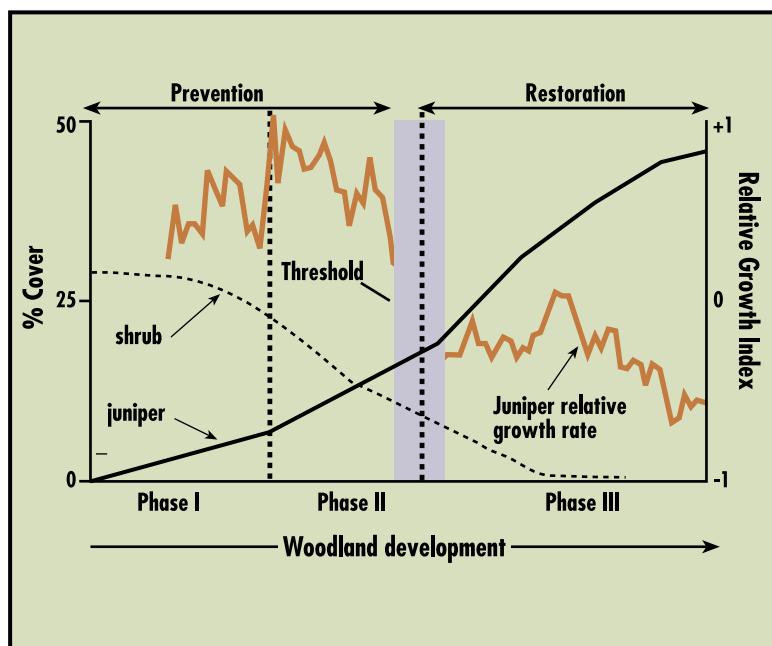
Figure 22. Three phases of woodland succession in mountain big sagebrush communities.

Figure 22a. Subordinate—Phase I.

Figure 22b. Co-dominant—Phase II.

Figure 22c. Dominant—Phase III on a south aspect with a soil restrictive layer at 16–18".

Figure 22d. Dominant—Phase III on a north aspect and deep well-drained soil.

Table 4. Stand characteristics differe
needlegrass (maximum juniper cover = 25–41%, Idaho fescue (maximum juniper cover 34–58%), and Columbia needlegrass (maximum cover = 60–75%) (derived from Miller et al. 2000).

Characteristics (post-settlement stands)	Phase I (early)	Phase II (mid)	Phase III (late)
Tree canopy (% of max. potential)	Open, actively expanding ≤10%	Actively expanding 10 to 30%	Expansion nearly stabilized >30%
Leader growth (dominant trees) (cm/yr)	terminal >10 lateral >10	terminal >10 lateral 5 to >10	terminal >10 lateral <5
Crown lift[1] (dominant trees)	Absent	Absent	Lower limbs dying or dead where tree canopy >40%
Potential berry production	Low	Moderate to high	Low to near absent
Tree recruitment	Active	Active	Limited
Leader growth (understory trees) (cm/yr)	terminal >10 lateral >8	terminal 5 to >10 lateral 2 to >8	terminal <5 lateral <2
Shrub layer	Intact	Nearly intact to significant thinning	≥75% dead

[1] Crown lift is the mortality of lower tree limbs, usually due to shading by neighboring trees.

Grasses and forbs

Although it is often stated that the herbaceous layer declines as western juniper increases in dominance, only a few studies have evaluated this relationship for western juniper. Two types of experiments support the hypothesis that western juniper overstory significantly affects production, diversity, and cover of the herbaceous layer: (1) spatial, comparing different transitional states within plant associations (Bunting et al. 1999, Miller et al. 2000), and (2) temporal, comparing herbaceous response over time between cut and uncut western juniper plots (Bates et al. 2000, Eddleman 2002d).

Miller et al. (2000) reported that the relationship between herbaceous cover and western juniper canopy cover differed among plant associations. Herbaceous vegetation in plant associations characterized by Thurber needlegrass, which often had a restricted subsoil layer or strong argillic horizon, was the most sensitive to increasing tree dominance (Fig. 27a). Mean herbaceous cover, in early states of woodland development, was 16 percent, compared to 5 percent in late stages of development. However, herbaceous cover was not significantly different between different stages of woodland development in plant associations characterized by Idaho fescue (Fig. 27b). In central Oregon, the presence of western juniper was associated with an increase in bare ground and smaller, more widely spaced grass clumps on relatively shallow soils (Roberts and Jones 2000) and a significant decrease in ground cover (Knapp and Soulé 1998). This was consistent with results from southwestern Idaho, where herbaceous cover also decreased in the mountain big sagebrush alliance as western juniper dominance increased (Bunting et al. 1999). However, changes in species richness across the transitional phases of woodland development were not consistent. In southwestern Idaho and southeastern Oregon, species richness did not change as western juniper increased in dominance (Bunting et al. 1999, Miller et al. 2000). In contrast, species richness declined in Thurber needlegrass communities in Oregon and in Idaho fescue communities in northeast California (Miller et al. 2000). Herbaceous species diversity and richness also significantly increased following western juniper removal on a mountain big sagebrush/Thurber needlegrass plant association (Bates et al. 2000).

Figure 23. Terminal leader growth is obvious throughout the outer canopy.

Figure 23a. A 35-year-old tree with little competition of other trees.

Figure 23b. Indistinct for trees older than 150 years.

Two studies that compared cut and uncut treatments reported significant increases in herbaceous cover and biomass, when trees were removed. On Steens Mountain, Oregon, herbaceous cover was only 2 percent and biomass 34 lbs/acre in closed woodland on a mountain big sagebrush/Thurber needlegrass site with a duripan 16–20 inches below the surface (Bates et al. 2000). On adjacent cut sites herbaceous biomass was 293 lbs/acre and basal herbaceous cover increased to 6 percent 2 years after cutting. The total number of species was 46 on the cut plots and 26 on the uncut. The greatest difference in species richness was for perennial herbs; 7 in woodland plots and 15 in cut plots. In a mountain big sagebrush/bluebunch wheatgrass site in central Oregon, cover and production of tall perennial grasses were 4.6 percent and 58 lbs/acre, respectively in woodland plots and 14 percent and 347 lb/acre on adjacent cut plots (Eddleman 2002d).

Studies suggest the decline in herbaceous vegetation in the presence of increasing tree dominance is largely dependent upon depth to restrictive layers including cemented ash, duripans, and clay layers in the B horizon (Miller et al. 2000). This is in part a result of western juniper roots being concentrated near the soil surface (Fig. 28). Although the literature is limited on this topic, we suggest the negative effect of western juniper on herbaceous cover and biomass increases from wet to dry sites (e.g. increasing from Idaho fescue to bluebunch wheatgrass to Thurber needlegrass) and with the presence of soil-restrictive layers. In addition, seed production of herbaceous species is reduced in closed woodlands (Bates et al. 2002).

Thresholds

A threshold can be viewed as a transition from one "state" (or plant community) to another. In plant ecology, thresholds generally represent a transition that is difficult to reverse. Often the transition involves not only a change in plant species, but also a change in processes (erosion, infiltration, water balance, fire, etc.). Multiple thresholds, abiotic, biotic, and economic are crossed as juniper dominance increases in shrub-steppe and aspen communities. Examples of biotic thresholds include modification of fuel loads and structure resulting in a changed fire regime, loss of native plant seed pools, and replacement of native plants by exotics. Changes in community structure and/or composition may reach a point where habitat is no longer suitable for some wildlife species. Abiotic thresholds include loss of topsoils by erosion and changes in soil characteristics, the hydrologic cycle, and fire regime.

Although we have a conceptual picture of thresholds (Fig. 21), we have no quantitative definitions defining or identifying the point at which different thresholds are crossed. Most thresholds are probably crossed as the role of western juniper shifts from codominant to dominant (transition from Phases II to III). This is the point where western juniper begins to control many of the community processes on the site. As a community approaches a threshold, our ability to predict outcomes following disturbance greatly decreases. For example, our ability to predict the pathway of plant succession following the removal of western juniper by fire or mechanical treatment is relatively high in communities with abundant native grasses and forbs in the prevention stage (Phases I and II, Fig. 21) unless the disturbance is severe. Successional pathways and plant composition following wildfire are also predictable in communities in poor condition that have clearly crossed a threshold into a new steady state. In a state and transition model, both communities are in a steady state. However, as communities are in a transitional state and nearing thresholds, responses following disturbance become less predictable as

to which steady state the community will shift following disturbance. Type, intensity, and frequency of disturbance or management can push the site across a threshold to a new steady state [e.g., cheatgrass or medusahead *(Taeniatherum asperum)* dominated site] or back from the threshold (e.g., dominance of native herbaceous and shrubs increases). If continued degradation occurs (e.g., loss of soils) on juniper dominated sites abiotic rather than biotic components will control the processes on the site.

Nutrient and Organic Matter Cycling

Nutrient and carbon (C) cycling in western juniper woodlands has received limited research attention. Most studies have focused on spatial distribution of soil nutrients and C and the monitoring of short-term changes in soil nutrient availability. There is a lack of long-term assessment of changes to soil processes as plant communities convert to juniper woodlands. In addition, the effects of juniper control treatments on soil nutrient cycling and nutrient capital have not been well documented. A key element in the rehabilitation of shrub-grassland communities from juniper-dominated systems will be to maintain site nutrient capital.

Spatial distribution of carbon and nutrients

Studies in western juniper woodlands indicate that greater amounts of soil nutrients and C are accumulated in litter and soils beneath juniper canopies compared to interspace soils. The spatial variability of soil nutrient and organic matter (OM) content is characteristic of many arid and semiarid systems. Higher concentrations of nutrients and OM measured in soils and litter layers beneath shrub/tree canopies compared to adjacent interspaces result in the formation of "resource islands" or "fertility islands". Formation and maintenance of resource islands is thought to be advantageous for shrubs and trees in these systems. In several systems, particularly African savanna and southwestern ecosystems of the United States, resource island formation by woody plants appears to be important for maintenance of site fertility and may enhance productivity of the herbaceous understory.

Resource islands beneath western juniper canopies appear in a few cases to enhance understory cover and productivity. In some juniper communities, for example, presence and cover of Idaho fescue is greater under juniper canopies than in the surrounding interspace zone. However, this may be the result of a resource island effect, a moderated microclimate, or a combination of both factors. In most western juniper woodlands, resource islands do not confer any benefits to the herbaceous and/or shrub understory as long

Figure 24. Conceptual model illustrating the hypothesized time periods required for western juniper to reach (1) minimum tree stocking densities to fully occupy a site (Phase I), (2) co-dominance (Phase II), and (3) dominance (Phase III) on sites of varying potential. The dashed line represents cool wet sites, and the dotted line warm dry sites.
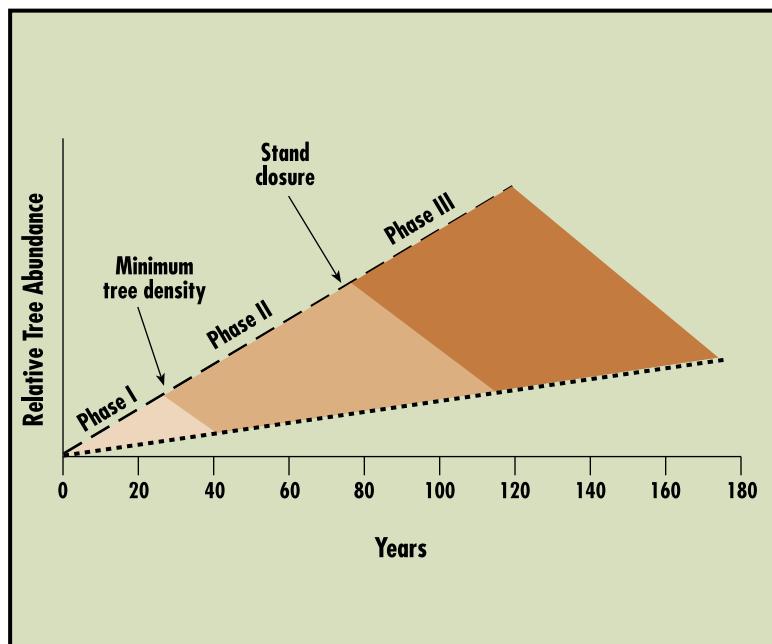
Figure 25. Annual relative growth index (ratio of annual ring width : mean ring width) based on a composite of 10 mature western juniper trees that established in the late 1800's in what is now a Phase III woodland on Juniper Mountain in Owyhee County, Idaho. The decline in growth during the 1950's is probably a result of increased competition between trees as the woodland developed. Overstory (dominant trees) density on this site is 240 trees/acre.
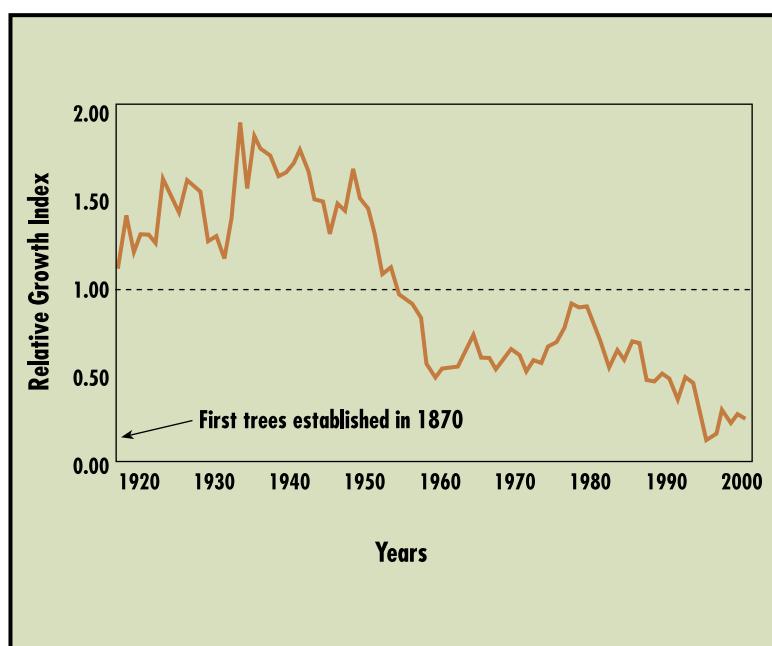
Figure 26. Developing western juniper woodland in a mountain big sagebrush/Thurber needlegrass plant association that has crossed the threshold and entered Phase III. Numerous sagebrush skeletons are scattered across the site. Picture was taken on Tule Mountain south of Alturas, northeastern California.

as trees remain in place. This is confirmed in successional studies by Miller et al. (2000) where cover of herbaceous and/or shrub species declined as woodlands developed. Generally, the benefits of higher resource availability are not realized unless trees are removed by fire, cutting, or other mechanical means. When trees are removed, herbaceous productivity and cover are significantly greater in canopy-influenced soils (resource islands) compared to interspace zones (Vaitkus and Eddleman 1987, Bates et al. 1998).

Litter and belowground accumulation

Nutrient and litter distribution are altered when juniper invades and dominates sites. In central Oregon higher concentrations of calcium (Ca) and potassium (K) occurred under mature juniper trees compared to interspace soils (Doescher et al. 1987). The greatest concentrations of nitrogen (N) and OM were in soils under juvenile (less than 40 years old) tree canopies. Bates et al. (2002) measured

Table 5. Characteristics of cl
dominant and subdominant trees and total shrub cover.

Plant association	No. sites	% Juniper cover	# Overstory trees/ac	% Shrub cover	Location	Province
JUOC/ARTRV /FEID[1]	11	50 (41–58)	146 (73–202)	2.9	Juniper Mt, OR	High Desert
ARTRV/STCO	4	64 (54–81)	101 (67–169)	2.6	Steens Mt, OR	High Desert
ARTRV/STTH	6	34 (25–41)	66 (36–145)	0.5	Tule Mt, CA	Klamath
ARTRV/FEID	15	53 (34–66)	82 (50–145)	3.0	Tule Mt & Devils Garden, CA	Klamath
ARTRV/STTH	3	24 (20–30)	92	0.5	Steens Mt, OR	High Desert
ARTRV/FEID	19	53	407 (288–588)	1.0	Juniper Mt, ID	Humboldt
ARTRV/AGSP	7		220 (102–450)	1.0	Juniper Mt, ID	Humboldt
ARTRV/FEID	4		390 (324–496)	1.0	South Mt, ID	Humboldt
ARTRV/AGSP		38	300 (191–487)	1.0	South Mt, ID	Humboldt
ARTRV/FEID			231 (93–595)	1.0	Steens Mt, OR	High Desert
ARTRV/AGSP			112 (47–192)	1.0	Steens Mt, OR	High Desert
ARTRV/AGSP	1		100		Combs Flat, OR	John Day

[1]JUOC = western juniper, ARTRV = mountain big sagebrush, FEID = Idaho fescue, STCO = alpine needlegrass, STTH = Thurber needlegrass, AGSP = bluebunch wheatgrass

significantly greater concentrations of N and organic carbon (OC) in canopy-influenced soils at 0–4 inches in soil depth than at the same depth in interspace soils in an 80-year-old juniper woodland. These patterns were attributed to greater litter accumulations under trees than in the interspace zones, which largely consisted of bare soil (Bates 1996). In aspen stands invaded by juniper, higher soil C:N ratios (13.3 vs. 12.3) and pH (7.3 vs. 6.8) were measured under juniper canopies than under aspen trees (Wall et al. 2001). There were also higher concentrations of salts, lime, and sulfate and lower concentrations of magnesium (Mg), iron (Fe), manganese (Mn), and copper (Cu) in juniper-influenced soils than aspen-influenced soils.

Josiatis (1990) evaluated effects of juniper occupancy on soil characteristics in southwestern Idaho. Trees were separated into three age classes: young (52–60 years), mature (81–108 years), and old (180–450 years). Across all age classes, concentrations of Ca, K, sodium (Na), magnesium (Mg), and percent soil OM content, total N, and nitrate (NO_3^-) were significantly greater in soils under tree canopies than in the interspace. Soil N, NO_3^-, OM, and Ca increased significantly with tree age. Soil cation exchange capacity (CEC) and pH were significantly greater in soils beneath trees compared to the interspace and both CEC and pH increased in soils under older trees.

Klemmedson and Tiedemann (2000) assessed litter and soil nutrient concentrations and OC over a range of tree age classes (21–231 years) in central Oregon. Their results showed an increase in the accumulation of OC, N, phosphorus (P), and sulfur (S) with greater tree age in the litter horizon beneath the canopy. In addition, greater concentrations of soil OC, N, and P occurred under tree canopies compared to interspace locations. Concentrations of exchangeable Ca and K in soils also tended to increase with tree age class. They concluded that exchangeable Ca and Mg and possibly N were being depleted in the interspace soils and being redeposited in soils and litter beneath tree canopies. Both Klemmedson and Tiedemann (2000) and Doescher et al. (1987) measured the highest nutrient concentrations in surface soil horizons, and found that concentrations declined with horizon depth.

Aboveground accumulation

Woodland formation also results in substantial accumulations of C, N, and S in aboveground biomass of individual trees (Tiedemann and Klemmedson 2000). These accumulations in juniper biomass occur at the expense of the shrub and herbaceous layers. Values reported from Tiedemann and Klemmedson (2000) indicate that biomass accumulations of nutrients in a juniper-dominated system would be greater than in pure sagebrush grassland. However, the methods used for biomass

Figure 27. Photos showing two very different responses of the herb layer to a site fully occupied by western juniper (Phase III).

Figure 27a. The mountain big sagebrush/Thurber needlegrass plant association has a restrictive layer at 16–20 inches of soil depth that limits tree rooting depth (see Fig. 28).

Figure 27b. In contrast, the mountain big sagebrush/Idaho fescue plant association has a deep (greater than 30 inches) Pachic Haploxerolls soil lacking a restrictive layer. Western juniper roots are distributed throughout this soil profile.

calculations make it difficult to extrapolate up to a community scale. The study was conducted on an individual tree basis and made no comparisons among woodland successional stages.

Tiedemann (1987) compared N accumulations in biomass among low-shrub grassland, big sagebrush grassland, and piñon-juniper woodland. While his data indicate greater above- and belowground biomass accumulation in the piñon-juniper woodland, caution must be used when interpreting the results. He compared plant communities that are very different in site potential. To date, no comparisons have been made in overall nutrient and C accumulations among shrub-grassland and woodland successional stages on sites with similar potentials and characteristics.

Figure 28. Compressed layer of juniper roots in the upper 12 inches of soil caused by a cemented ash layer at 12 inches.

Soil nutrient availability: undisturbed versus disturbed woodlands

The availability of soil nutrients for uptake by plants is a critical element influencing ecosystem productivity in arid and semiarid environments. Availability of nutrients for plant uptake is controlled by an array of variables, interacting over time and space. Variables include soil characteristics, soil moisture and temperature, populations of soil organisms, and extreme weather events, such as drought. The interactive effects of drought and wet cycles significantly influence soil microbial and faunal activity as well as plant growth, all of which in turn impact availability of soil nutrients for plant uptake. Nitrogen availability[15] has received the most attention in the literature because it is assumed that N is the most limiting soil nutrient in wildland systems. Thus, this section will focus on N availability in soils for plant uptake in western juniper woodlands.

[15] Plants take up N in two forms from soils, ammonium (NH_4^+) and/or nitrate (NO_3^-). Ammonium is derived during mineralization (ammonification) of organic matter by soil fauna and microorganisms and from nitrogen fixation processes. Nitrate is evolved from activities of nitrifying bacteria and other soil organisms utilizing NH_4^+ in decomposition of organic matter. Available N fractions are said to immobilize when soil microorganism N demand exceeds the rate that these fractions become available. Microbial demand for available N may exceed supply if organic matter quality is low (organic matter with high carbon/N ratio) and when fresh litter substrates are added to soils such as by root turnover or leaf drop. This is not to say that plants cannot take up N during periods of net immobilization. Nitrogen in soils is never completely immobilized as biological activities are constantly renewing available N fractions into the soil solution.

Undisturbed woodlands

In controlled (constant temperature and moisture) laboratory incubations, N availability was greater in canopy-influenced soils than tree interspace soils collected in semiarid and arid shrub lands and woodlands (Charley and West 1977, Everett et al. 1986, Klopatek 1987). However, laboratory incubations measure the potential of soils to supply N and do not reflect conditions in the field where soil temperature and moisture fluctuate, thereby influencing soil microbial activity, plant uptake, and soil N availability. These studies suggest the nutrients are sequestered beneath the juniper canopy but may not necessarily make them more available for understory plant use.

Roberts and Jones (2000) compared N mineralization among four types of microsites (bare soil, grass, sagebrush, and juniper) in a central Oregon juniper-sagebrush-grass community. Nitrogen mineralization analysis was conducted in a laboratory environment and results indicated that available N was greater in soils under grass and sagebrush than in soils that are bare or under juniper. Nitrogen availability and mineralization rates were similar between bare patches and soils under juniper. Bates et al. (2002) derived similar results in field incubations during an above-average precipitation year in an eastern Oregon woodland.

Several field studies measuring N availability have been conducted in western juniper woodlands. Myrold et al. (1989) recorded net rates of N immobilization in a yearlong field study in central Oregon juniper woodlands. Unfortunately they did not report monthly measurements, so we are unable to determine seasonal availabilities. In another field trial in western juniper woodlands, Bates et al. (2002) consistently measured greater levels of net N mineralization (ammonification and nitrification) rates and extractable NO_3^- levels in interspace soils, compared to canopy-influenced soils. This result indicated a lack of a resource island effect for available N in canopy-influenced soils. The form of C input appears to be a controlling factor in determining N availability in the woodland. Carbon inputs to the interspace zone are derived primarily from root turnover. In the canopy zone, C is input from root turnover and deposition of low-quality, aboveground juniper litter. Juniper leaf litter is of low quality (high C:N ratio, 77:1, Wall et al. 2001) and decomposes very slowly (Bates 1996). Soil microbial demand for N increases with lower quality litter, which explains the lower rates of N mineralization in canopy soils. Tiedemann and Klemmedson (1995), using a bioassay approach, also found no difference in uptake of plant available N between canopy and interspace in western juniper soils.

Potential N losses from denitrification in soils of western juniper communities appear to be limited. Measurement of denitrification rates and potentials were found to be low in western juniper woodlands

when compared to other forest types (Vermes and Myrold 1991).

In a greenhouse study, Josiatis (1990) compared the emergence and growth of bitterbrush, ceanothus, Idaho fescue, and bluebunch wheatgrass between soils collected from beneath the juniper canopy and the interspace. Except for ceanothus, all other species had greater emergence rates in canopy-influenced soils than in soils from the interspace zone. Above-ground biomass of bitterbrush, Idaho fescue, and bluebunch wheatgrass tended to be greater in canopy-influenced soils compared to the interspace soils. Belowground biomass of ceanothus was greater in canopy-influenced soils compared to the interspace soils. Biomass of ceanothus, Idaho fescue, and bluebunch wheatgrass also tended to be greater in soils collected beneath older trees, perhaps indicating increased availability of nutrients. The results indicate the potential growth benefits that juniper resource islands may provide to associated species. Because this study was conducted in a noncompetitive environment with frequent watering, results cannot be directly extrapolated to a field situation. The results do indicate that after juniper removal, growth potentials for associated plant species may be enhanced in canopy-influenced soils when compared to interspace soils. This was confirmed in field trials conducted in central Oregon by Vaitkus and Eddleman (1987) and on Steens Mountain, Oregon by Bates et al. (1998, 2000, 2002).

Disturbed woodlands

In forested systems, disturbance such as tree cutting or fire often stimulates a flush of nutrient availability (Vitousek and Mellilo 1979). Cutting trees in western juniper woodland increased N availability and rates of N mineralization in soils the first year following treatment (Bates et al. 2002). However, in the second year post-cutting, there were few treatment differences in either available N levels or net N mineralization between cut and uncut woodland. Although treatment differences attributed to cutting were measured, the major influence on the year-to-year variation in available N were seasonal weather patterns, which likely overwhelmed the treatment effect. The first sample year was a moderately dry year and the second sample year was considered a very wet year. The high rates of N mineralization and buildup of available N in soils the first year after cutting coincided with very dry soil conditions. Under these conditions N uptake by plants was limited, which contributed to the accumulation of inorganic N in soils. The second year after cutting was wetter and cooler. The low levels of available N in the second year probably resulted from uptake from soil N by plants and soil microorganisms. Greater plant growth and inputs of plant litter in the wet year would provide an increased source of soluble

C from aboveground biomass and root turnover for soil fauna and microorganisms, thus increasing demand for N. Others have measured similar patterns in N mineralization and available N pools during drought and wet cycles in semiarid systems (Birch 1960, Ingham et al. 1986, Fisher et al. 1987). Ingham et al. (1986) established that increased plant growth and root turnover during wet periods stimulated microbial demand for N and thus reduced N availability in prairie soils in Wyoming.

After trees were cut, soils beneath the cut trees had lower nitrification and N mineralization rates compared to the interspace (Bates et al. 2002). The lower rates of N mineralization and nitrification in soils under juniper debris compared to the interspace around the cut trees may result from inputs of low-quality (high C:N) litter from cut trees (Bates 1996). Carbon:nitrogen values were 55:1 in juniper leaves and 240:1 in the twigs and branches, significantly greater than soil C:N. Immobilization of available N fractions by soil microflora increases with input of litter with a high C:N ratios (Schimel and Firestone 1989, Davidson et al. 1992).

A management concern after tree cutting in forested systems is the potential for increasing outflows of soil N, primarily in the form of NO_3^-, which is highly mobile in the soil (Vitousek and Mellilo 1979, Knight et al. 1991, Parsons et al. 1994). Bates et al. (2002) measured large increases in available NO_3^- in soils the first growing season after cutting, but the increase was transient. The high levels of soil NO_3^- were immobilized in soils by the second winter after cutting. Because of the methods used to assess available N and N mineralization, Bates et al. (2002) concluded that most of the available N in soils was taken up by soil microorganisms and not lost via leaching or denitrification. In a further study, Vermes and Myrold (1991) also concluded that denitrification potential in soils of western juniper communities were low.

Litter decomposition

The effects of felling juniper trees on litter decomposition and N release were examined over a 2-year period on Steens Mountain, Oregon in a basin big sagebrush/Thurber needlegrass plant association (Bates et al. 2002). Juniper leaf litter decomposition was 37 percent greater in cut woodland than in uncut woodland. Greater litter inputs and higher litter quality from juniper slash were thought to have caused a priming effect, resulting in higher decomposition rates in cut woodlands. The increase in litter decomposition in the cut treatment did not result in an earlier release of litter N. Nitrogen was limiting for decomposers under juniper slash, resulting in the importation and immobilization of litter N. The retention of N in litter in the early stages of decomposition following cutting may serve as an important sink

that conserves N on a site. In the uncut woodlands, 20 percent of litter N was removed, indicating that N was not limiting during decomposition. Despite retention of N in juniper litter in the cut woodland, there was no indication that N was limiting for plant growth. Herbaceous plants in the cut treatment had significantly higher N concentrations than in the uncut treatment. Total aboveground biomass N uptake in the herbaceous in cut woodland layer was nine times greater than uptake by the herbaceous layer in the uncut woodlands.

Soil arthropods play an important role in the initial breakdown of litter and OM. Arthropod biomass was lower in a juniper community than in a sagebrush-grass community (Roberts and Allen 2000). The lower biomass levels in the juniper community were attributed to lower structural diversity (greater amounts of bare ground and less cover of shrubs and grasses than in the sagebrush community) rather than direct affects of the juniper canopy. Alternatively, the greater soil arthropod numbers in sagebrush-grassland plots than in areas with juniper may reflect differences in litter quality and availability. Wall et al. (2001) measured litter depth and litter C:N in aspen stands dominated by juniper. Aspen leaf litter was of higher quality (C:N=45:1) compared to juniper (C:N=77:1). Litter depth was 4 inches under juniper compared to 1.5 inches under aspen, which suggests both reduced decomposition rates and lower rates of nutrient turnover in stands dominated by juniper.

Nutrients in juniper debris and organic layer: site restoration considerations

There is increasing debate over the impacts that juniper treatments have on site nutrient capital, particularly when trees are removed for commercial energy production and wood products or when juniper debris is burned after cutting. Debris removal is often required by the Oregon Department of Forestry when juniper-cutting treatments are within fire-protection zones. Loss of nutrients could potentially affect long-term site productivity and recovery. Unfortunately, this subject has received very limited study and is an area that requires additional research effort.

Burning of piñon-juniper debris and litter can influence the concentration of soil nutrients. Gifford (1981) measured an increase soil N, P, K, and OC following cutting, piling, and burning piñon-juniper debris in Utah the first year after treatment, but not the second year. DeBano and Klopatek (1988) measured increased phosphatase activity and P concentration after burning piñon-juniper litter when soils were wet. Phosphatase activity was reduced when sites with dry soils were burned; resulting in a 50 percent reduction of P in the litter.

Tiedemann and Klemmedson (2000) argued that removal of western juniper slash, such as by burning or removal of trees, may result in substantial nutrient loss, particularly N, contained in debris and organic soil layers. Leaving juniper debris and slash in place after mechanical treatment retains nutrients on site and may be important in maintaining site productivity (Bates et al. 2002). Tiedemann and Klemmedson (2000) estimated that N contained in trees and organic layers on one site in central Oregon was 16–30 percent of the amount contained in a 12-inch soil profile. However, the estimated amount of N contained in aboveground biomass will vary depending on site, phase of woodland succession, and measurement depth. Bates (1996) estimated that N contained in juniper biomass represented 6 percent of the total amount contained in the top 4 inches of the soil profile. Tiedemann (1987) estimated that N in piñon-juniper biomass and organic layers was about 18 percent of the total amount of N contained in a 24-inch-deep soil profile.

Alternatively, as tree canopy dominance increases, the potential for nutrient loss off-site may also increase. In many juniper woodlands, the level of bare ground connectivity increases in tree interspace zones resulting in greater potential soil loss from increased overland water flow and erosion. Loss of nutrients by erosion and runoff may offset inputs from atmospheric deposition and accumulation of nutrients in juniper biomass and litter layers. As described in the hydrology section, cutting of juniper resulted in an almost complete reduction in overland flow and sediment yield off site (Pierson et al. 2004). Increasing the density of ground cover in treated woodlands is an important component in retaining soil nutrients on site.

Summary

Western juniper expansion into sagebrush grassland has demonstrated the potential for woodland formation to alter the spatial distribution of soil OM, C, and nutrients. If erosion increases as juniper woodlands develop, the potential loss of nutrients off site in sediment will ultimately cause a reduction in community productivity. Nutrient dynamics in woodlands have received so little research emphasis that developing firm conclusions from the limited number of studies is problematic. Only one study (Bates et al. 2002) evaluated N availability across multiple years. While cutting did produce a flush of available N the first year after treatment, the effect was transient. In addition, the main factor influencing N dynamics in cut and uncut woodlands was determined to be seasonal macroclimate variation, which included the effects of both drought and above-average precipitation events.

Hydrology

Impacts of Western Juniper on Hydrology and Erosion

The impacts of expanding western juniper woodlands or juniper control treatments on hydrology and water balance are not well understood. Variability in soils, geology, slopes, and spatial and temporal variations in climate and precipitation that occur across the range of western juniper make it difficult to generalize the hydrologic response from one watershed to another (Hawkins 1987). Little or no research has been conducted on how western juniper directly influences any of the specific components of the water budget for a given watershed. In the absence of pertinent research results, proxy and anecdotal evidence and research from other parts of the United States are commonly used to fuel the debate on important questions such as: (1) does western juniper intercept and/or transpire excessive moisture and thereby dry up springs and reduce stream flow; and (2) does western juniper increase surface runoff and erosion? The following is a discussion of the impacts of western juniper and western juniper control practices on effective precipitation, base stream/spring flow, and hillslope runoff and erosion.

Effective precipitation

The amount of precipitation that enters the soil and is stored with the soil profile at a given point within a watershed is termed effective precipitation. This amount can be different than the amount of precipitation that falls from the sky. Effective precipitation is reduced in western juniper woodlands through leaf interception. The amount intercepted is determined by percent of tree canopy cover and the duration, intensity, and type of precipitation. The potential for interception of precipitation is greater in juniper woodland than in a shrub-steppe community (Eddleman et al. 1994). The reduction of precipitation reaching the ground directly beneath a mature western juniper canopy was 20 percent at the canopy edge and 50 percent halfway between the canopy edge and 70 percent at the trunk (Young et al. 1984). With 9.25 inches of precipitation, these authors estimated that 42 percent of the total was intercepted by the juniper canopy. However, storm intensities and duration were not accounted for in this study. In central Oregon, up to 74 percent of precipitation was intercepted when measured directly beneath the tree canopy (Eddleman 1986, Larson 1993). Stem flow represented a small portion of the total precipitation (Young et al. 1984). Larson (1993) reported stem flow was absent in small storm events (less than 0.26 inches of precipitation). Once the

tree canopy reached saturation (at about 0.4 inches of rainfall), water started to flow down the stem. In a 1.0-inch event, stem flow accounted for up to 5 percent of the precipitation. The remaining moisture that does not reach the ground by throughfall or stem flow is lost to the atmosphere by way of evaporation or sublimation. Studies suggest the juniper canopy can significantly reduce the amount of precipitation reaching the soil surface.

In areas with significant snowfall, changes in vegetation density and structure associated with western juniper encroachment may significantly change how much snow is trapped and stored on different parts of the landscape, and may alter the timing and rate of snowmelt. To date, no work has documented the direct impact of western juniper woodlands on the dynamics of snow accumulation and melt. Studies from other plant communities have shown that change in vegetation height and distribution can strongly influence snow redistribution by wind. Sturges (1975) reported that removal of brush changed snow movement within sagebrush communities. On windy sites, snow accumulates to the height of the dominant vegetation, with additional snow being redistributed to other topographic positions or areas with taller or denser vegetation (Sturges and Tabler 1981, Pomeroy and Gray 1995).

Stream/spring flow

The base water flow of a stream or spring comes from water moving laterally underground. The source of the water is from precipitation that infiltrates the soil and moves along an impermeable layer until it once again reaches the surface in a stream channel or spring. The source of such groundwater can come from areas great distances from the stream channel or spring. No watershed-scale studies have ever been conducted in western juniper areas to investigate the impact of western juniper on ground water flow. Only anecdotal evidence exists for streams, springs, and meadows that have dried up due to the increase of western juniper, or again began to flow following western juniper removal.

Two long-term studies in the Southwest demonstrated that removal of Utah juniper generated modest increases in base stream flow (Clary et al. 1974, Baker 1984). Over an 8-year period following treatment, annual stream flow increased by 157 percent. Stream flow returned to pretreatment levels after dead trees were removed from the watershed. Peak stream discharge from a

Figure 29. Average cumulative runoff for juniper-dominated sites (n = 8) (Phase III) and sites where juniper had been removed for 10 years (n = 8). Rainfall was applied at 2.1 inches/hour, which approximates a 2-year return interval thunderstorm after 5 minutes and a 50-year return interval thunderstorm after 30 minutes. Site is a basin big sagebrush/Thurber needlegrass plant association on Steens Mountain, Oregon.
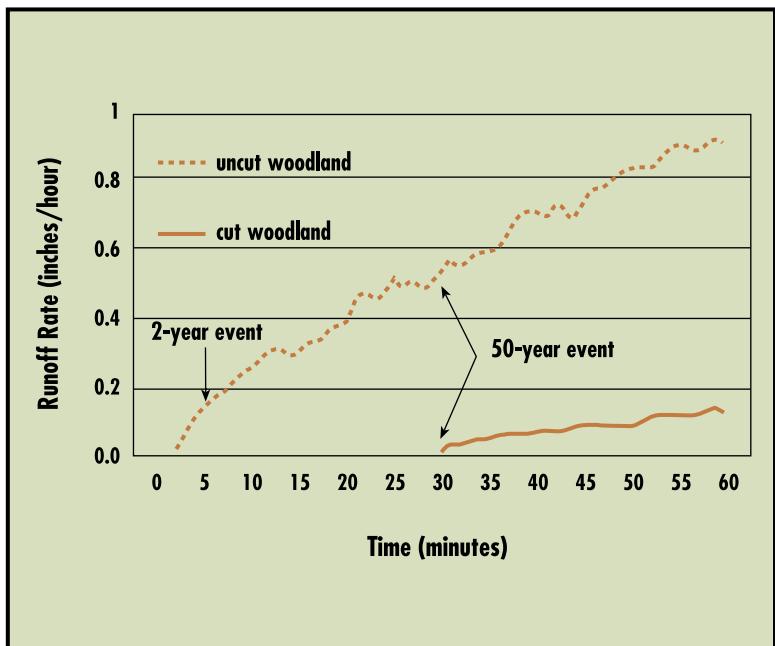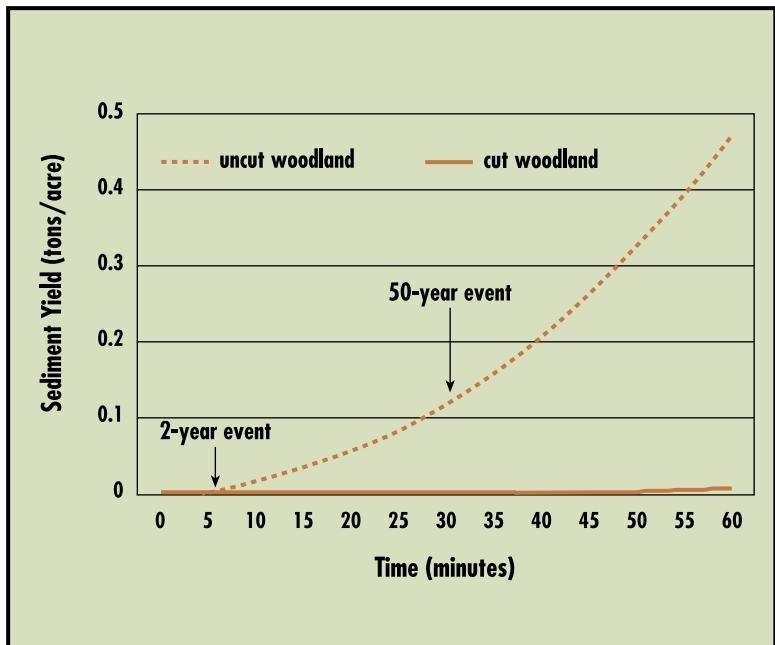
Figure 30. Average cumulative sediment yield for juniper-dominated sites (n = 8) and sites where juniper had been removed for 10 years (n = 8). Rainfall was applied at 2.1 inches/hour, which approximates a 2-year return interval thunderstorm after 5 minutes and a 50-year return interval thunderstorm after 30 minutes. Plant association is a basin big sagebrush/Thurber needlegrass plant association on Steens Mountain, Oregon.

large thunderstorm, however, was 1.3 times higher for the herbicide treated watershed compared to the woodland control. Wilcox (2002) summarized the relationship between Ashe juniper in Texas and Utah juniper in New Mexico and concluded that Ashe juniper control could increase stream flow based on two factors: (1) junipers have a high canopy interception of precipitation; and (2) junipers often exist on shallow soils underlain by permeable rock layers conducive to lateral movement of ground water. Hibbert (1983) evaluated the potential for increased water yield through vegetation management within different western rangeland plant communities. He concluded that little potential exists for increasing stream flow in watersheds receiving less than 18 inches of annual precipitation. In watersheds dominated by winter snow, the annual precipitation threshold for increasing water yield may be as low as 16 inches. The response of stream flows following the removal of western juniper is likely dependent upon amounts and timing of precipitation, geology, soil characteristics, size of the watershed, successional phase of the developing woodland, and area occupied by western juniper.

Hillslope runoff and erosion

In semiarid environments, most surface overland flow and erosion is generated during thunderstorms when high-intensity rainfall rates exceed the infiltration, interception, and surface storage capacities of the soil and vegetation (Wilcox 2002). Pierson et al. (2002) studied the impact of high-intensity rainfall on hillslope hydrologic response within various rangeland plant communities across the western United States and concluded that most hillslope hydrologic responses were a result of complex vegetation-soil interactions. A change in the pattern and density of vegetation cover can alter soil properties and retention of overland flow resulting in changes in hillslope infiltration and surface runoff patterns (Wood 1988). Greater plant density, cover, and dispersion offer greater soil protection from raindrop impact and can slow the movement of water flowing across the soil surface, resulting in less erosion (Blackburn et al. 1994). Studies from sagebrush areas have shown that over time, shrubs produce modified microsites with decreased bulk density, increased soil organic matter, and increased aggregate stability that in turn cause a several-fold increase in infiltration capacity and dramatic decreases in erosion compared to the soil between shrubs. The areas between the shrub canopies have less herbaceous and litter cover, so the soil is more susceptible to raindrop impact that

increases soil crusting, decreases infiltration, and increases erosion (Blackburn 1975, Blackburn et al. 1992, Pierson et al. 1994).

Dominance of western juniper on a site has been shown to decrease cover of shrub and herbaceous vegetation, particularly in areas with shallow root-restricting layers (Burkhardt and Tisdale 1969, Adams 1975, Knapp and Soulé 1998, Bunting et al. 1999, Miller et al. 2000, Roberts and Jones 2000). Bates et al. (2000) showed that cutting western juniper resulted in increased shrub and herbaceous cover on sites in southeastern Oregon. On the same sites, Pierson et al. (2003) compared the hillslope hydrologic response of western juniper-dominated hillslopes with hillslopes where the western juniper had been removed 10 years earlier. They found that hillslopes dominated by western juniper produced runoff from small thunderstorms that occur in the area every 2 years. Hillslopes where western juniper had been removed by cutting only produced runoff from large thunderstorms that occur once every 50 years (Fig. 29). For a large 50-year return interval thunderstorm, sheet erosion on western juniper hillslopes produced over 275 lb/acre of sediment compared to 0 lbs/acre on the hillslopes without western juniper (Fig. 30). During large thunderstorms, rill erosion on the western juniper hillslopes was over 15 times greater than on the hillslopes without western juniper.

Research from piñon-juniper watersheds in the Southwest has consistently demonstrated the strong relationship between vegetation cover and soil erosion by wind and water (Wilcox 1994, Baker et al. 1995, Reid et al. 1999). Runoff and soil erosion are highest in the bare ground areas found in the intercanopy zone between juniper trees and lowest near the base of trees protected by canopy and high amounts of ground cover (Fig. 31) (Wilcox et al.1996, Reid et al. 1999). Davenport et al. (1998) presented a conceptual framework that suggests the rate of soil erosion is a balance between soil erosion potential (SEP, a function of climate and soil properties) and ground cover condition. A threshold of accelerated erosion is crossed when ground cover is reduced to a point where runoff can move along a continuous flow path through connected intercanopy zones. High SEP is a result of soil properties such as texture and aggregate stability, increased slope, high rainfall intensity, and low intercanopy ground cover. Management practices that maintain adequate plant cover density on piñon-juniper hillslopes reduce soil loss and sustain site productivity (Baker et al. 1995).

Figure 31. A juniper dominated site (Phase III) that has eroded to a restrictive layer in the intercanopy zone.

Hydrologic impacts of western juniper control treatments

Mechanical

Mechanical treatments such as chaining can affect both the hydrologic and erosion condition of a site by altering surface soil and vegetation characteristics (Gifford 1975). The magnitude and duration of impact of a mechanical treatment is directly proportional to the degree that the treatment disrupts critical soil and vegetation properties (Gifford and Skau 1967, Brown et al. 1985). Mechanical treatments can positively impact the process of infiltration by altering such soil properties as bulk density, soil structure, and macroporosity (Blackburn 1983, Hutten and Gifford 1988). However, such treatments also may reduce infiltration rates and increase runoff on sagebrush rangeland due to surface sealing of bare soil exposed to raindrop impact (Gifford and Skau 1967, Tromble 1976, Gifford 1982, Brown et al. 1985). Surface runoff is impacted by disruption of surface drainage patterns, increases in detention storage caused by furrows, dikes, and dams created by the implement, and increased surface roughness (Tromble 1976, Brown et al. 1985, Hutten and Gifford 1988, Clary 1989). The susceptibility of the site to erosion can be increased by changes in soil erodibility and loss of plant cover to protect the soil surface from splash erosion by raindrop impact (Brown et al. 1985, Hutten and Gifford 1988). Maintenance of vegetation cover, soil organic matter, and surface litter all reduce the proportion of

bare soil impacted by rainfall and can help to reduce the negative hydrologic impacts of mechanical treatments (Gifford and Skau 1967, Blackburn and Skau 1974, Blackburn 1983, Brown et al. 1985, Hutten and Gifford 1988).

Chemical

Herbicides can be used to kill the overstory of juniper with little impact to the understory vegetation or the soil surface. Baker (1984) killed all the piñon-juniper on a 360-acre watershed in Arizona with herbicide. The treatment resulted in a 730-lb/acre increase in herbaceous biomass compared to the adjacent untreated watershed.

Fire

Few studies have examined the impact of fire on rangeland hydrology. Most studies have shown an increase in runoff and erosion rates the first year following fire (prescribed or wild), and returning to pre-fire rates within 5 years (Wright and Bailey 1982). Roundy et al. (1978) studied the impact of prescribed fire on hillslope hydrology of a piñon-juniper woodland on loamy soils in eastern Nevada. They found that fire had the greatest impact on areas directly below the juniper and sagebrush canopies with high surface litter accumulations. Water repellency under unburned trees in the

juniper duff was greater than where the duff layer had burned. Across the site fire had little effect on infiltration rates, but did significantly increase soil erosion. Pierson et al. (2003) summarized results of studies on the impact of fire within coarse-textured sagebrush-dominated systems and concluded that the greatest impact of the fire was on overland flow dynamics and rill erosion. Fire induced significant water repellency, particularly in areas dominated by shrubs with large accumulations of litter (Pierson et al. 2002). However, such systems were also found to have a high degree of natural water repellency when extremely dry (Pierson et al. 2001). The result is that burned or unburned woodlands will rapidly generate runoff under intense rainfall in the absence of vegetation in the tree canopy interspace. The immediate effect of fire is the reduction of ground surface barriers, which include shrub, herbaceous vegetation, and litter. The water then concentrates and increases in velocity resulting in greater erosive energy (Pierson et al. 2003). Water moves more rapidly down slope and ultimately into stream channels, impairing water quality and potentially causing downstream flood damage. An important component in evaluating the impacts of fire on the hydrology of a site is the vegetation response following fire, especially recovery of vegetation structure and surface litter.

Wildlife

Shrub-steppe communities in Phases I and II, (Fig. 22 a, b) of woodland encroachment contain a high degree of vertical diversity, and are attractive to wildlife. These transitional communities are used by 83 species of birds and 23 species of mammals (Maser and Gashwiler 1978). Sixteen wildlife habitats were defined on the basis of structure and diagnostic plant species across eastern Oregon (Maser et al. 1984a, b; Puchy and Marshall 1993). Of the 16 types, the western juniper/sagebrush/bunchgrass type ranked third in having the highest number of wildlife species using this habitat for feeding or reproducing. However, these summary papers do not address woodland dynamics, successional states, or separate presettlement and post-settlement woodlands. Communities containing western juniper can range from open stands with a diverse understory of shrubs and grasses to closed woodlands with little understory vegetation. Open western

juniper/big sagebrush/bunchgrass stands are mid-successional (Phases I and II), and characterized by herbaceous, shrub, and tree layers. As western juniper dominance increases, structural diversity declines with the loss of shrubs and some herbs (Miller et al. 2000). Old-growth stands also differ structurally from post-settlement woodland, including having a greater density of cavities, which significantly influences cavity nesting species. The replacement of aspen, riparian, and mountain big sagebrush communities by western juniper may have detrimental effects on wildlife populations dependent upon these habitats. In summary, low levels of western juniper can be beneficial for many wildlife species but increasing dominance at both the community and landscape levels will result in a general decline in landscape and plant community diversity, resulting in a decline of wildlife abundance and diversity.

Large Herbivores

Western juniper's primary influence on large herbivore habitat relates to cover and food. Lechenby et al. (1971) and Leckenby and Adams (1986) reported mule deer heavily used stands of western juniper during severe winter conditions. They found that weather conditions were less severe in western juniper woodlands with 30 percent cover of trees at least 15 ft tall compared to adjacent shrub communities. Leckenby et al. (1982) concluded that dense stands of trees or shrubs over 5 ft tall provided optimal thermal cover. However, these stands provide minimal food resources. Deer will browse western juniper during the winter if little else is available. Both digestibility and levels of available protein are low in western juniper. Increased western juniper dominance across the landscape will result in a decline in browse resources (Adams 1975, Miller et al. 2000, Schaefer et al. 2003). In northeastern California, the decline of mule deer populations in the late 1960's may in part be related to the concurrent increase in western juniper dominance and the decline in shrubs (Schaefer et al. 2003). The large decline in mule deer populations in southwestern Idaho in the late 1950's and 1960's also coincides with woodland transition from Phases II to III, resulting in the rapid decline in shrub cover. Replacement of big sagebrush with western juniper will also have negative impacts on fawning habitat. In the spring on Steens Mountain, Oregon, mountain big sagebrush with a canopy cover greater than 23 percent was the most preferred fawning habitat (Sheehy 1978).

The relationship between western juniper and other large herbivores such as elk *(Cervus elaphus)* and wild horses has received little attention. Trainer et al. (1983) reported that antelope (Antilocapra americana) rarely used western juniper woodlands during the winter or spring in eastern Oregon, preferring more open shrub-steppe communities or stands with only scattered trees.

Birds

Until recently, few studies have evaluated the effects of western juniper and woodland structure on avian populations. Throughout the piñon-juniper biome, past work has not recognized woodland transitional states, separated old-growth from young post-settlement stands, or evaluated the effects of western juniper encroachment into various habitats on avian populations.

Winter habitat and food source
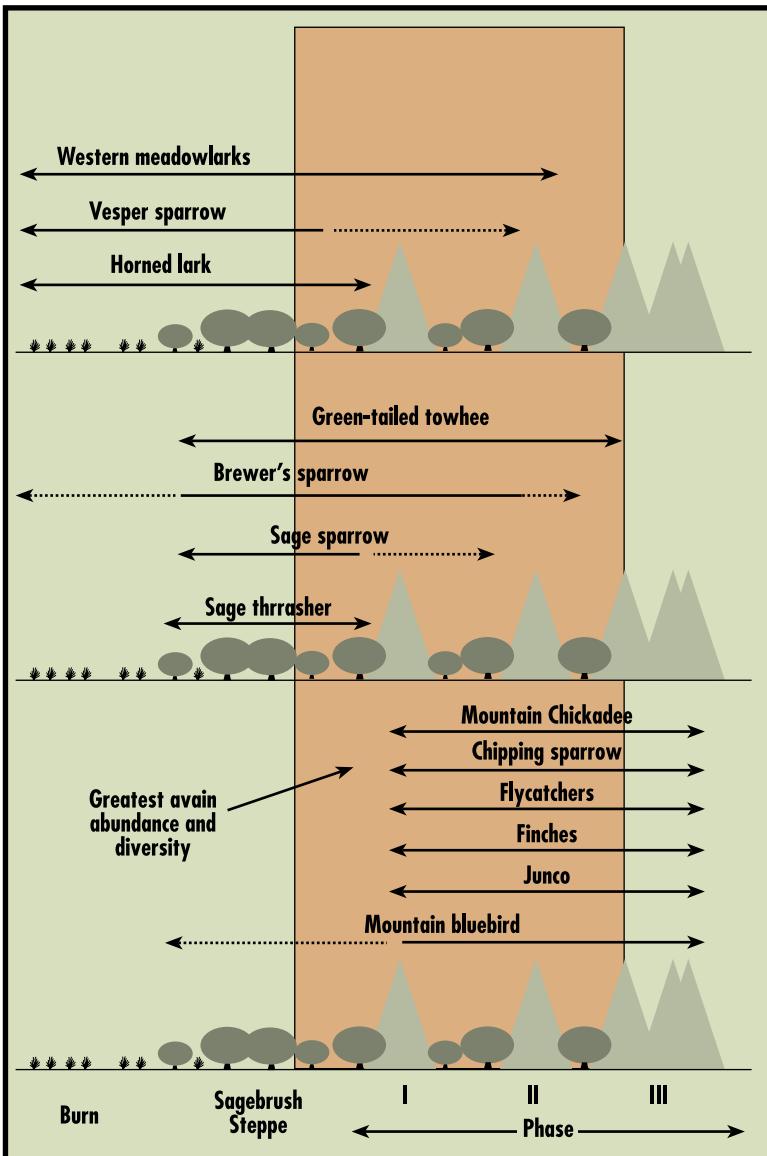
Stands of old-growth western juniper and open post-settlement western juniper/sagebrush/bunchgrass communities provide important winter bird habitat. Densities of winter (December through February) birds in open western juniper woodlands were significantly greater than in adjacent shrub-steppe or grassland communities (EOARC, unpublished data). Large numbers of American robins and Townsend's solitaires use these wooded communities during the winter months. They first arrive in wintering areas late in the fall and remain until April. Western juniper berries (female cones) provide an important source of food for Townsend's solitaires, American robins, mountain bluebirds, cedar waxwings, Steller's jays, and scrub jays (Fig. 8) (Lederer 1977, Solomonson and Balda 1977, Poddar and Lederer 1982). Western juniper berries are the sole winter food used by Townsend's solitaires and make most of the American robin's diet throughout the winter (Lederer 1977, Poddar and Lederer 1982). Solitaires consumed up to 80 ripe berries/day and American robins up to 220/day. Ripe western juniper berries provide a good source of energy but contain low levels of protein (Poddar and Lederer 1982). Birds were observed avoiding the green berries, which are less nutritious. Densities of mountain bluebirds and territory size for Townsend's solitaires are closely related to the abundance of western juniper berries. Fruit production varies greatly among years and locations. As woodlands close and competition between trees increases, female cone production declines. Several species of birds, including cedar waxwings and American robins, also feed on the small pearl-like berries of mistletoe growing on western juniper (Kuijt 1960, Maser and Gashwiler 1978).

Breeding habitat

Species richness (total number of species) and diversity (an index based on the total number of species and abundance of individual species) of birds generally increases with structural complexity of the plant community. The many birds that were reported to use western juniper woodlands in central Oregon (Fig. 32) (Maser and Gashwiler 1978; Maser et al. 1984a, b; Puchy and Marshall 1993) pertained to communities in Phase I and II (Figs. 22 a, b), which still contained a complex understory. In eastern Oregon, avian species diversity and richness were greater in Phases I and II western juniper mountain/big sagebrush

communities compared to mountain big sagebrush communities, where trees were absent. This results from an increase in tree-nesting species, which include the chipping sparrow *(Spizella passerina),* flycatchers *(Empidonax spp.***),** Cassin's finch *(Carpodacus cassinii),* and house finch *(C. mexicanus).* Maximum densities of these species were reached at relatively low densities of western juniper (Noson 2002). No bird species reported

Figure 32. Changes in avian composition across successional stages from grassland to shrub-steppe to juniper woodland. Avian species richness (total number of species) is greatest in Phase I and early to mid-Phase II. Illustration is based on Reinkensmeyer 2000, Noson 2002, and EOARC unpublished data. Dashed line indicates presence but declining use by the species.

in the literature are obligates to closed western juniper woodland. Several shrub-steppe bird species showed differences in sensitivity but an overall negative correlation to increasing western juniper (Noson 2002, Reinkensmeyer 2000). Noson (2000) reported Brewer's sparrows *(Spizella breweri),* vesper sparrows *(Pooecetes gramineus),* and sage thrashers *(Oreoscoptes montanus)* showed a strong negative correlation to increases in western juniper density and to area occupied by western juniper. The sage thrasher was the most sensitive to western juniper encroachment, sharply declining at very low western juniper densities. Reinkensmeyer (2000) reported a 90 percent decline in sage thrasher densities in western juniper stands with only 6 percent tree cover. However, green-tailed towhees *(Pipilo chlorurus)* were positively correlated with increases in western juniper communities, occupying up to 33 percent of the area (Noson 2002). Brewer's sparrows used transitional communities in Phases I and II that contained adequate levels of sagebrush cover (estimated to be more than or equal to 10 percent). Abundance of tree-nesting species including flycatchers, mountain chickadees *(Poecile gambeli),* dark-eyed juncos *(Junco hyemalis),* house wrens *(Troglodytes aedon),* chipping sparrows, and mountain bluebirds increased in the early stages of woodland encroachment (Phase I) (Fig. 32). However, the continued increase in juniper dominance did not result in an increase in these species. These studies suggest that as woodland succession enters Phase III, avian abundance, diversity, and richness will decline with loss of understory species and structural complexity.

Avian communities are also strongly influenced by aspen communities at the landscape level. Species richness and diversity in sagebrush communities were strongly and positively correlated with the presence of nearby aspen stands (Noson 2002). The encroachment and eventual replacement of aspen communities by western juniper would be expected to have a negative effect on this relationship (Wall et al. 2001).

Sagebrush obligate species, including sage-grouse *(Centrocercus urophasianus),* are sensitive to western juniper encroachment into sagebrush communities. The density of juniper at which use by greater sage grouse declines or ceases has not been determined. However, in central Oregon, sage grouse avoided western juniper communities for nesting and winter use (BLM 1994). As tree densities increase and woodland area continues to expand, sage grouse habitat will decline, especially in mountain big sagebrush habitat below 7,000 ft.

Old-growth woodlands

Densities of tree and cavity nesting species were 20 percent higher in old-growth western juniper woodlands compared to western juniper-sagebrush-bunchgrass communities in Phases I and II (Reinkensmeyer 2000). Both tree- and cavity-nesting species accounted for 67 percent of the total bird density and 66 percent of the number of species present in the old-growth western juniper type. The increase in cavity-nesting species in old-growth stands maintained the relatively high species richness, diversity, and evenness of avian populations. Tree cavity-nesting species, including red-breasted nuthatches *(Sitta canadensis),* American kestrels *(Falco sparverius),* ash-throated flycatchers *(Myiarchus cinerascens),* mountain bluebirds, mountain chickadees, and northern flicker *(Colaptes auratus),* were found in greater numbers in old-growth than in post-settlement woodlands (Reinkensmeyer 2000; EOARC, unpublished data). However, there was considerable overlap in tree foliage and cavity-nesting species between the young developing woodlands and old-growth western juniper woodlands. Shrub- and ground-nesting species were absent or scarce in old-growth stands.

Small Mammals

The bushy-tailed *(Neotoma cinerea)* and dusky-footed *(N. fuscipes)* woodrats are commonly associated with western juniper (Verts and Carraway 1998). The bushy-tailed woodrat is found throughout the range of western juniper but is most common in old-growth stands where it nests in hollow tree trunks. The dusky-footed woodrat builds stick houses for nesting, often located at the base of western juniper trees but occasionally in the tree canopy. On the east side of the Cascades, the dusky-footed woodrat is primarily found in Klamath and south Lake counties in Oregon, and Modoc and Lassen counties in California. Woodrats, cottontails, black-tailed jackrabbits *(Lepus californicus),* and porcupines *(Erethizon dorsatum)* utilize western juniper foliage for food during portions of the year (Maser and Gashwiler 1978). During summer drought, 25 percent of the Nuttall's cottontail *(Sylvilagus nuttallii)* diet was composed of western juniper foliage (Verts and Hundermard 1984). Western juniper female cones are consumed by deer mice *(Peromyscus maniculatus),* yellow-pine chipmunks *(Tamias amoenus),* and golden-mantled ground squirrels *(Spermophilus lateralis).* Deer mice open up the nutlets to consume the seeds. They are also known to cache seeds for later consumption (Vander Wall 1990).

Few studies have evaluated the relationships between small mammal populations and western juniper dominance. Possibly the greatest impact of western juniper on small mammal populations is via indirect effects on understory plant species (Miller et al. 2000). Reductions in the shrub layer would impact populations of the Great Basin pocket mouse *(Perognathus parvus),* yellow-pine chipmunk, and desert cottontail *(Sylvilagus audubonii).* Greater numbers of Great Basin pocket mice were captured in Phase II western juniper woodlands that contained a shrub understory than in an old-growth stand with less than one percent shrub cover (Willis and Miller 1999). However, equal numbers of white-footed deer mice *(Peromyscus leucopus)* were found in both communities. Elmore (1984) reported twice as many species and a 60 percent increase in deer mice, piñon mice *(Peromyscus truei),* and Ord's kangaroo rats *(Dipodomys ordii)* in thinned compared to unthinned western juniper stands. Several studies in the Intermountain West have shown small mammal numbers generally increase when western juniper is either thinned or completely cut, provided that the slash remains (Kundaeli and Reynold 1972, O'Mera et al. 1981, Elmore 1984, Severson 1984, Willis and Miller 1999). Thinning or removing western juniper improves food and cover for small mammals by increasing shrub and herbaceous recruitment and seed production (Bates et al. 2000, 2002).

Management Considerations

Western juniper can be an important element in the habitat for many wildlife species, but at densities that allow a healthy understory of shrubs and grasses (Miller 2001). We know of no data suggesting there are juniper-obligate species, or species that require dense, closed western juniper woodlands. Maintaining low densities of western juniper on portions of the landscape increases the abundance, diversity, and richness of avian and small mammal populations in the shrub-steppe. However, as western juniper dominance increases, wildlife abundance, species richness, and diversity decline. This will also occur as the proportion of area dominated by western juniper at the landscape level increases. Noson (2002) concluded that although fire had an immediate negative impact on several shrub-nesting species, periodic burning was important in limiting western juniper encroachment into shrub-steppe communities. Wall et al. (2001) also concluded that fire was an important factor in preventing the conversion of aspen stands to western juniper woodlands. Maintaining small, scattered stands of dense western juniper may be desirable to provide thermal cover from severe winter conditions for large ungulates. However, management strategies that maintain a balance of grasslands, shrub-steppe, and open western juniper woodlands will provide the greatest abundance and diversity of wildlife populations at the landscape level. Old-growth woodlands that provide valuable habitat for cavity-nesting birds should be maintained.

Restoration and Management

Introduction

Previous sections in this overview provide strong evidence that western juniper has significantly increased in density and distribution since the late 1800's and if left unchecked can have significant impact on soil resources, plant community structure and composition, water and nutrient cycles, wildlife habitat, and biodiversity. As a result, control of western juniper has been a major concern of land management since the early 1960's. Justifications used for western juniper control include restoration of preinvasion plant communities, increasing forage production and quality, reducing soil erosion, increasing water capture on site, increasing spring and stream flow, improving wildlife habitat, and increasing biological diversity. In the early years, the emphasis on juniper control was to increase forage production for livestock. However, in the last decade the primary justification for juniper control was to enhance proper site function (i.e., capture and store of water, retain soil nutrient capital, restore shrub-steppe communities, etc.).

In the 1960's through the early 1970's chaining and dozing were the most common forms of western juniper control. However, chaining is not currently used on public lands due to high costs and a perception by the public that the treatment has a high disturbance impact on the site. In the late 1970's, the U.S. Bureau of Land Management (BLM) Prineville district began using chainsaws as its primary method of western juniper control. By the 1980's and 1990's this practice became widespread. Research on the effects of western juniper cutting began in the early 1980's and has steadily expanded. Chemical control of western juniper has been tested but have produced mixed results. In the 1990's, the use of prescribed fire to control western juniper greatly increased. The use of mechanical shears and whole tree chipping of western juniper has increased since 2000 in northeastern California (mainly in Modoc and Lassen counties) and south-central Oregon (Lake County) for providing biofuel for power generation at the Honey Lake power plant in Wendell, California.

Since the 1970's various groups have challenged western juniper control treatments based on the limited scientific evidence that supports western juniper removal. Belsky's (1996) review of western juniper treatments demonstrated that some of the justifications for western juniper control were based on anecdotal evidence and were not supported by experimental evidence. Recent and ongoing research has addressed many of the concerns raised about western juniper control, although knowledge gaps in nutrient cycling and hydrologic processes remain.

Until recently, long-term data sets were lacking. Several studies now provide information from treatments that are older than 10 years. Work in central Oregon provides long-term assessments of vegetation response and successional patterns using several treatment methods.

This section is a synthesis of research evaluating the response of plant communities following various western juniper treatments. In "Guidelines for Management" (page 54), we lay out a framework for developing the appropriate action related to western juniper control. A summary of results from research, BLM, and privately administered western juniper treatments in the 1960's and 1970's is provided in Appendixes 2 and 3.

Assessment of Western Juniper Control Justifications

Before presenting a critique of western juniper control practices, we will address some of the specific concerns that have been raised regarding the justifications that are used to support western juniper removal. These responses primarily refer to juniper-dominated stands. Several of the concerns brought up by Belsky (1996) have been addressed in recent and ongoing research, including questions regarding plant community and wildlife response, hydrologic function, and the identification of pre- versus post-settlement woodlands. There remain several areas where additional work is required to better quantify ecosystem response to the current woodland expansion and western juniper control methodologies.

Does western juniper removal restore plant communities?

There are studies that illustrate both successes (Young et al. 1985, Bates et al. 2000, Eddleman 2002d) and failures (Young et al. 1985) of plant community rehabilitation following western juniper treatment. The three key components that will largely influence success or failure are: (1) site selection, particularly pre-treatment understory composition; (2) method(s) used to control western juniper; and (3) follow-up management. The level and speed of community response depends on several factors including post-treatment weather conditions and management, grazing history, site potential (soils and plant community), seed banks, and plant composition prior to treatment. There are still too few studies that allow us to accurately predict plant succession after treating western

juniper, particularly when one steady state is transitioning to another. Clear guidelines have not yet been developed when a threshold has been crossed. In several studies, the response of exotic annual grasses exceeded the response of remaining native vegetation or seeded perennials (Young et al. 1985, Vaitkus and Eddleman 1987). Warmer, drier sites, especially south- and west-facing aspects and/or sites below 5,000 ft are likely to have an exotic annual grass component (Eddleman 2002a; EOARC, unpublished data)(see section on weeds). These sites historically were big sagebrush (basin or mountain) with bluebunch wheatgrass and/or Thurber needlegrass as the dominant grass. On these drier sites, many of the original plant species can be restored following western juniper control if at least 2-3 deep-rooted perennial grasses per 10ft^2 persist on the site. However, exotic annuals will generally remain a part of the community. Restoring system functionality should be the primary goal on these sites, as it is unlikely that the preinvasion plant community composition will fully return. More productive sites at higher elevations and lower elevation sites with northern aspects are usually more resilient, less susceptible to weed invasion, and have a greater potential to return to pre-woodland community characteristics following treatment, compared to more arid sites (Quinsey 1984; Koniak 1985; EOARC, unpublished data). These sites tend to be characterized by mountain big sagebrush, Idaho fescue, and alpine needlegrass. Eddleman (2002d) estimates that one and two perennial grasses per 10ft^2 are sufficient to allow recovery of these sites following western juniper control.

Does western juniper removal increase forage production and quality?

Productivity of forage species and forage quality can increase after western juniper control, in some cases increasing 8- to 10-fold (Appendix 4)(Young et al. 1985, Vaitkus and Eddleman 1987, Bates et al. 2000). However, the response is primarily driven by pretreatment plant composition and post-treatment management. Crude protein levels of forage species utilized by livestock and wildlife were 50 percent greater in cut versus uncut western juniper woodlands (Bates et al. 2000). Season of available green forage for livestock and wildlife can increase 4–8 weeks for at least the first several years following western juniper control.

Does western juniper removal reduce soil erosion and increase water capture on site?

Recent data on a drier big sagebrush/bluebunch wheatgrass and Thurber needlegrass site indicate that the potential for significant soil erosion and rill formation increases with western juniper dominance ("Hydrology," page 35). Pierson et al. (2003) measured greater runoff, sediment yields, and rill formation in uncut woodlands compared to cut woodlands. Bates et al. (2000) measured increased water capture and storage on cut western juniper woodlands compared to adjacent uncut woodlands.

Does western juniper removal increase spring discharge and subsurface flow?

There have been no experiments designed to link western juniper control to increased spring or stream flows. Anecdotal evidence has for years suggested that removal of western juniper increases spring flows and water table levels (see "Hydrology"). Climatic fluctuations make it difficult to verify this response. It is our opinion that the relationship between western juniper and subsurface flow is site specific, determined by topography, soils, geology, and amount of precipitation.

Does western juniper removal improve wildlife habitat?

Western juniper can be an important habitat element for many wildlife species if a healthy understory of shrubs and grasses is maintained. Maintaining low densities of western juniper on portions of the landscape, resulting in increased structural diversity, will increase the abundance, diversity, and richness of avian and mammal populations in the shrub-steppe. Western juniper cutting has resulted in higher capture rates of small mammals than in adjacent woodlands (Willis and Miller 1999). Closed canopy woodlands also supported lower numbers and diversity of avian species than adjacent treated woodlands (EOARC, unpublished data). On cut plots in Grant County, Oregon, numbers and diversity of avian species were greater on the cut plots where slash remained compared to adjacent closed woodlands (Miller et al. 1999b). However, wildlife response will be highly dependent on vegetation response following treatment.

Does western juniper removal increase biological diversity?

In many cases, biological diversity of herbaceous plants increases following a reduction of western juniper. Increased diversity primarily results from increased emergence of perennial and annual forbs following cutting or fire in western juniper (Bates et al. 2000; EOARC, unpublished data). The effect of western juniper on species diversity may be site dependent (Miller et al. 2000) and cutting may reduce diversity on sites in poor conditions (Young et al. 1985), particularly where cheatgrass or medusahead may become dominant following treatment.

Mechanical Treatments

Chainsaw cutting

The most common method used to control western juniper in recent years has been cutting with chainsaws. Costs for cutting western juniper on BLM lands (based on 2004 bids) ranged from $36 to $80/acre. Though information gaps persist, chainsaw cutting of trees has been the most thoroughly researched method of western juniper control (Appendix 2). Tree cutting has been researched in two of the five Oregon ecological provinces where significant western juniper woodlands are present; these were the High Desert (Steens Mountain vicinity) and John Day (sites in Prineville vicinity and Grant County). Other ecological provinces where western juniper is present but treatments have not been assessed are the Klamath, Mazama, and Snake River provinces. The John Day Province is slightly warmer in the winter with growing seasons beginning 2–4 weeks ahead of the High Desert Province. Weedy annuals, especially at lower elevations and on drier sites, have been more of a concern in woodlands of the John Day Province compared to the High Desert. The High Desert Province has colder winters and shorter growing seasons, and exotic annuals have posed less of a problem following restoration efforts. Research emphasis in the two provinces has differed. Research in the John Day Province has focused on woodland cutting, effects of slash dispersal, and seeding of perennial species. Research in the High Desert Province has centered on combinations of cutting and prescribed fire, post-treatment grazing, and has emphasized natural regeneration rather than seeding. For these reasons we will discuss research results from each province separately.

John Day Ecological Province

Studies in central Oregon have assessed long-term (more than 10 years) post-treatment vegetation dynamics with emphasis on (1) assessing shrub and herbaceous response to western juniper removal on different plant community types, and (2) combining western juniper cutting and slash treatments with seeding of native cultivars and introduced perennial grasses.

Eddleman (2002d) evaluated shrub/herbaceous response after western juniper cutting on three different sites (low sagebrush, basin big sagebrush, and mountain big sagebrush) during an 18-year period. Uncut woodland plots showed little change in herbaceous composition but there were measurable decreases in density and cover of shrubs at the low sagebrush and basin big

sagebrush sites. Cut treatments had large increases in shrub and perennial grass cover and density on all three sites. Early response (2 years after cutting) on these sites demonstrated the potential to increase herbaceous biomass by nearly 300 percent on the shallow-soil sites (low sagebrush) and by 100 percent on deeper soil sites (Vaitkus and Eddleman 1987). The initial increase in biomass was composed of cheatgrass but sites are presently dominated by perennial grasses (Eddleman 2002d). In a second study, perennial forb and grass cover in cut plots was 14 percent, compared to 8 percent in adjacent uncut closed woodlands (Eddleman and Miller 1999).

On many post-settlement woodlands in central Oregon perennial grasses have been depleted and seeding is necessary following western juniper cutting to avoid site dominance by annual grasses. Use of western juniper slash to provide a favorable micro-environment for perennial grass seedling establishment has been evaluated (Eddleman 2002b). Scattering western juniper slash was proven to be a successful method for establishing broadcast-seeded species on a basin big sagebrush/Thurber needlegrass-type site in average to above-average precipitation years. However, in dry years, slash covering had no effect on seedling establishment. After 13 years, seeded grasses that responded favorably to slash cover were Goldar bluebunch wheatgrass, Tegmar intermediate wheatgrass *(Agropyron intermedium),* and Rush wheatgrass *(Agropyron elongatum).* Because of the cost of scattering slash (up to $250/acre) this method should only be considered on highly erodable soils and slopes.

Roller punching to scarify soils followed by broadcast seeding and scattering of slash has also been successfully used to establish perennial grass species (Eddleman 2002 a, c). In dry years, roller punching and slash covering more than 50 percent of the treated area appears to be most beneficial for seedling establishment. In wet years, roller punching with surface slash covering between 0 and 25 percent of the area appears to be adequate for seeded species response.

Eddleman (2002) suggested that for seedings to be successful in the province, precipitation between November and January should exceed 5 inches, with none of the three months individually being below 1.7 inches precipitation. By evaluating moisture conditions over this period, managers could delay seeding until February if conditions were favorable.

Western juniper control in ponderosa pine communities was assessed to determine western juniper influences to pine growth (Rose and Eddleman 1994). Cutting of western juniper increased understory production by 50 percent

after 2 years but did not result in increased pine growth. There may have been a lag response by the pine to the cutting treatment; trees may have adjusted to reduced competition by expanding root systems and leaf area. This study also took place in above-average precipitation years, which may have masked treatment differences. To accurately assess pine response to western juniper removal would necessitate extending the study period beyond the two years used here.

High Desert Ecological Province

Studies in the High Desert Ecological Province have assessed short and long-term (10 years) treatment responses with emphasis on (1) shrub and herbaceous response to cutting and prescribed fire, and (2) effects of cutting on the nitrogen cycle and hydrologic function.

Basin big sagebrush/Thurber needlegrass-bluebunch association (1991–2003)

Vegetation response: Cutting trees in a western juniper-dominated basin big sagebrush/Thurber needlegrass-bluebunch community in 1991 resulted in significant increases in herbaceous cover and biomass in the first 2 years following treatment (Bates et al. 2000). Vegetation response was minimal the first year following cutting. In the second growing season herbaceous cover increased nearly nine-fold (329 lb/acre) in the cut woodlands compared to adjacent uncut woodlands (38 lb/acre). Early successional response indicates restoration requires patience as it may take several years for understory species to respond to the removal of western juniper, particularly during dry periods. Plant diversity was significantly higher in the cut compared to uncut woodlands.
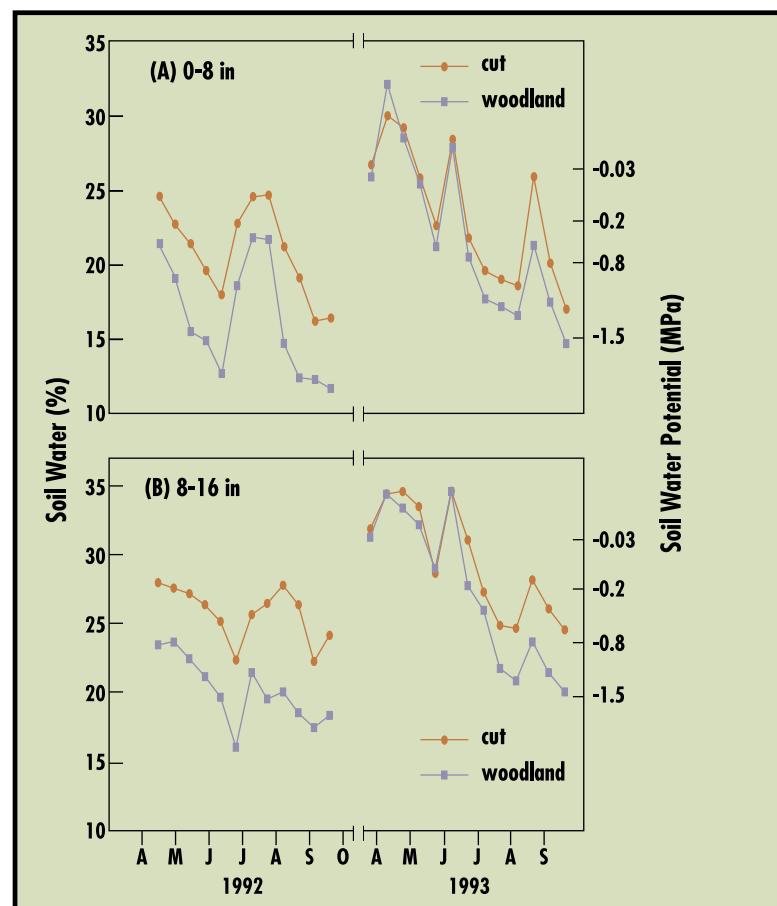
These same plots were measured again in 1994, 1997, 1998, and 2003. Perennial grass density increased between 233 and 300 percent (from 2–3 plants/10ft^2 to 10–14 plants/10ft^2) compared to uncut woodland. Cheatgrass and Japanese brome *(Bromus japonicus)* began increasing in 1994 and increased exponentially in 1997 and 1998 in the cut plots (Bates et al. 2000; EOARC, unpublished data). The increase in annual grasses was mainly confined to litter deposition areas under cut trees and around old stumps. Increases in annual grass in areas of western juniper litter deposition have been observed in studies in central Oregon and in south-facing mountain sagebrush/bluebunch wheatgrass communities on Steens Mountain. Since 1997, annual grass biomass has decreased (from 240 lbs/acre in1997 to 65 lbs/acre in 2003) (EOARC, unpublished data). Native perennial grasses remained the dominant component in the understory (from 550 lbs/acre in 1997 to 890 lbs/acre in 2003).

Soil water: Cutting resulted in increased soil water content and plant water availability the first two growing seasons after cutting (Figs. 33, 34) (Bates et al. 2000). Retaining western juniper debris on site reduced evaporative loss of soil water.

Soil erosion and runoff: Runoff, sediment yields, and rill erosion formation were significantly reduced 10 years (2001) following cutting when compared to adjacent uncut woodlands (refer to hillslope runoff and erosion in "Hydrology," Figs. 29, 30).

Understory response to cutting juniper in four mountain big sagebrush/Idaho fescue sites in various stages of woodland succession (Phases II and III) was evaluated on Steens Mountain in the 1990's (EOARC, unpublished data). Perennial grasses and shrubs were the major functional groups that responded to the cutting treatments. Perennial grass cover in cut late-successional woodlands doubled 5 years after treatment. In closed canopy and mid-successional woodlands there was little change in perennial grass and other understory cover during the same five year

Figure 33. Seasonal soil water potential at 0–8 and 8–16 inches soil depth for cut and uncut western juniper woodland on basin big sagebrush/Thurber needlegrass plant association on Steens Mountain, Oregon (Bates et al. 2000).

period. In all treated areas shrub cover increased significantly and bare ground declined. In northeastern California, bitterbrush leader growth was two to three times greater in western juniper communities where tree cover had been thinned to 5 percent compared to adjacent unthinned woodlands with tree cover of 30–50 percent.

Juniper cutting and debris burning in basin big sagebrush/Thurber needlegrass associations (1997–2002)

Prescribed burning of juniper debris was applied the first and second winters after juniper cutting and were compared to unburned cut plots (EOARC, unpublished data). Conditions under which burning was prescribed were that a) soils were frozen and/or wet (near field capacity); b) suspended juniper litter was dry, and c) herbaceous plants were largely dormant. Burning debris under these conditions successfully removed most of the juniper litter except for large branches and tree boles. Impacts to understory vegetation present beneath burned juniper debris were minimal. There was little measurable loss of established perennial grass species. Comparisons made between burned (1st and 2nd year burns) and unburned debris piles showed no differences after 4 years post-treatment in perennial grass density and cover, or presence of annual grass. Burned areas had significantly higher densities and larger individuals of annual and perennial forbs than unburned debris. Adjacent to this site debris piles were burned with dry soils in the winter and death of perennial grasses was significant compared to unburned debris the first year (EOARC, unpublished data). In central Oregon, Eddleman (unpublished data) reported fall burning of juniper slash under hot dry conditions resulted in a decline in native perennial grasses and large increases in cheatgrass.
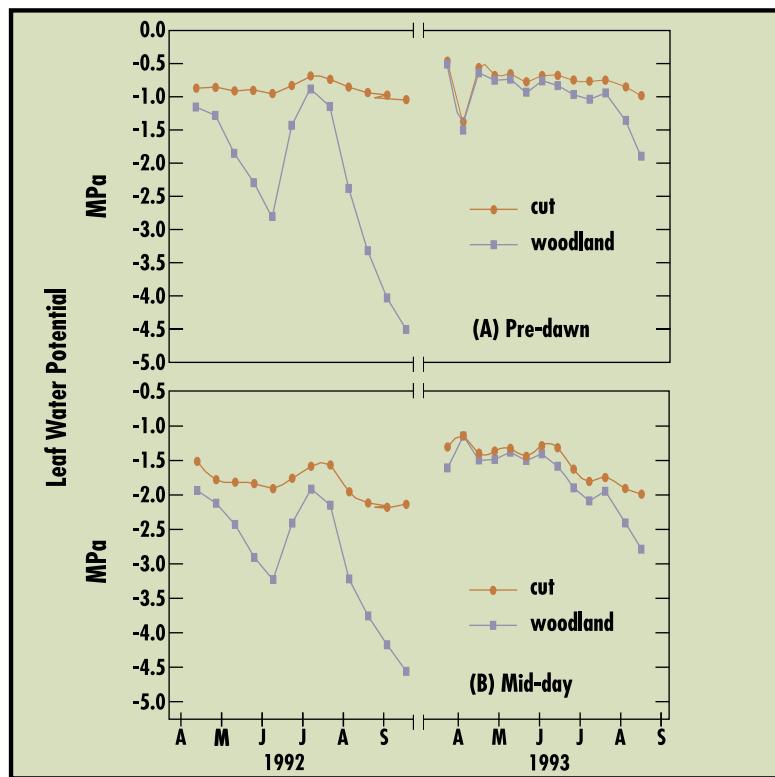
Summary

A major advantage of cutting is the high degree of control in the treatment application. Managers can select specific types of trees to cut or not cut (e.g., old-growth trees should be left on site for wildlife habitat). Unlike prescribed fire, treatment boundaries are predictable and potential liability is thus reduced. Cutting can be conducted almost year-round as long as access is not constrained by weather, road conditions, special land designations (e.g., wilderness study areas) and extreme fire conditions restricting chainsaw use. Cutting is not limited by terrain roughness as are heavy machinery applications. The only area of concern is that leaving cut trees on site can present a fuel load problem for several years following treatment. This is of particular concern in the urban interface, in woodlands that are adjacent to forested plant communities such as ponderosa pine forest where juniper tree densities are high, and where the understory has a significant cheatgrass or medusahead component. Primary disadvantages of cutting compared to fire are costs, limited size of area treated, and minimal control on small trees, which often require a follow-up treatment.

Cutting and leaving trees currently (2004) costs $36–80/acre. The higher cost reflects additional expense incurred when working where terrain is steep and not readily accessible from roads. However, costs increase to as much as $250/acre if cut trees are limbed and slash is scattered. Because of high costs, scattering slash should only be considered on sites with high erosion potential and where broadcast seeding is the only option for replanting. Scattering of slash may be partially achieved without cost by making western juniper cuts open to the public for firewood cutting.

The results indicate that restoration of woodland sites requires patience. Studies in western juniper and in piñon-juniper woodlands have shown delays of one to several years before the understory fully responds to removal of tree interference, especially

Figure 34. Leaf water potential at pre-dawn (least water stressed) and mid-day (most water stressed) for single trees remaining in the cut plots and uncut western juniper woodland on a basin big sagebrush/Thurber needlegrass plant association on Steens Mountain, Oregon. Leaf water potential is an index of soil water availability in the rooting zone (Bates et al. 2000).

when growing conditions are not favorable (Barney and Frischknecht 1976, Tausch and Tueller 1977, Vaitkus and Eddleman 1987, Bates et al. 2000). Even in wet years, a lack of significant understory response the first year after cutting may be expected. Existing plants require time to grow new roots and tillers and new plants need time to establish (Bates et al. 2000).

Heavy machinery

Heavy machinery used to control juniper species in the Intermountain West has included: bulldozers to push trees over; bulldozers pulling anchor chain or steel cable through stands to uproot trees; and various mechanical cutting and grinding devices mounted on dedicated logging equipment, such as feller-bunchers, excavators, front-end loaders, and farm-forestry tractors. Disturbance of the soil surface can vary from minimal to high depending on soil conditions, which include soil moisture content, soil texture, frozen or unfrozen ground, and tight turns by heavy equipment. Controlled research designs assessing plant response to western juniper removal by heavy machinery are limited to two studies.

Young et al. (1985) compared wood harvesting to mechanical clearing using a bulldozer on a relatively dry site in northeastern California. Without weed control the main herbaceous response was composed mostly of exotic annual grasses. Weed control using atrazine and drill seeding of perennials following mechanical clearing was the most successful treatment. Perennial production steadily increased and exceeded annual grass production by the fourth growing season after treatment. Wood harvest successfully removed trees but generated high levels of litter, which reduced the effectiveness of weed control measures and prevented establishment of an adequate perennial component.

Leckenby and Toweill (1983) seeded several chained western juniper communities in southcentral Oregon near Silver Lake with mixed results. They did not provide information on the success of the chaining in removing western juniper competition, however the area was primarily composed of large trees that were successfully removed (R.F. Miller, personal observation). Chained areas had successful establishment of introduced crested and Siberian wheatgrasses *(Agropyron cristatum* and *A. sibericum)* but other species contained in the seed mix rarely established. Seeded species did not establish on untreated woodlands.

Additional results describing mechanical treatment in western juniper woodlands can be found in workshop reports (Appendix 3). These results cover chaining projects in the Klamath Falls region and in central Oregon in the 1960–1980's. Formal research designs were not applied and follow-up monitoring was not adequate to assess long term treatment affects. Chaining has not occurred on public lands since the 1970's in Oregon or on private lands since the early 1980's.

Chaining has been used extensively in the past to control piñon-juniper woodlands throughout the Intermountain West (Stevens and Monsen, 2004). Currently Utah's Division of Wildlife Resources is the only agency continuing to use chaining to control piñon-juniper. Their main treatment goal is to improve big game habitat by increasing shrub and herbaceous production. Chaining practices in Utah are typically combined with seeding of native and/or exotic perennials. Research studies of chaining in Utah and Arizona indicate that two-way[16] chaining is highly successful in reducing piñon-juniper competition and encouraging increased productivity and cover of shrub and herbaceous species (seeded and unseeded treatments). Success of removing trees depends on age and size structure of the stand. Trees greater than 60 years and/or with stem diameters exceeding 2 inches are most easily controlled. Damage to the shrub and herbaceous layer is usually light to moderate. Follow-up treatment is necessary to remove saplings. Although chaining and seeding have generally proven successful when properly applied, this method is expensive with costs ranging from $60 to $200/acre (Chadwick et al. 1999). Herbaceous seed is usually broadcast prior to single chaining or between chainings (two-way chaining), which allows seed to be covered. This increases success on sites where broadcasting seed on bare ground usually is unsuccessful.

The use of equipment to mechanically shear or cut western juniper is common in the southwestern portion of its range, especially in northeastern California (Lassen and Modoc counties) and parts of eastern Oregon. Major users of this type of equipment in northeastern California and south-central Oregon (Lake County) are subcontractors who supply juniper chips to a biomass power plant in Wendell, California. The equipment is also used for land clearing for residential and commercial developments, especially in central Oregon, and rangeland habitat improvement in other locations.

[16] Two-way chaining is the practice of pulling an anchor chain twice over the landscape, with the second chaining pulled perpendicular to the first chaining.

Summary

Depending on the type of machinery used, mechanical treatment can be selective (single tree treatments) to landscape in size. Operators can control where and when to treat sites and liability is low compared to fire prescriptions. Heavy equipment use may be limited by rough terrain and tends to be expensive. The primary concern associated with heavy machinery use for western juniper control is the disturbance to soils and existing understory vegetation, particularly if post treatment recovery is dependent on species present on a site. Disturbance of the soil surface can vary from minimal to high depending on soil conditions (e.g., soil moisture, frozen or unfrozen ground, tight turns by heavy equipment, etc.). Higher levels of surface disturbance can increase opportunities for weed establishment and temporarily increase erosion potential. However, disturbance of the soil may also be beneficial if applied properly when seeding is required (Bruce Roundy, Professor of Range Ecology, Brigham Young University, Provo Utah, communication). In Utah, surface soil disturbance by chaining has been shown to increase establishment success of seeded species. Additional research and monitoring is necessary to properly assess the impacts that heavy machinery have on soil compaction and erosion, and damage to understory species. Soil type, plant composition, time of year, and weather conditions will influence the effect of heavy machinery on a particular site.

Fire

The use of prescribed fire to control western juniper has increased since 1990. A few controlled studies have evaluated post-fire succession or quantified fuel load characteristics required to conduct successful burns in developing western juniper woodlands. The primary factors that will influence post-burn response are plant composition and seed pools prior to treatment, ecological site (site potential), fire severity and extent, pre-and post-fire climate conditions, and post-treatment management.

Plant composition and seed pools

Important biotic factors driving post-fire succession are plant composition prior to treatment, individual species' response to fire, and existing seed pools. The abundance of desirable and undesirable species and their ability to tolerate fire will largely determine post-fire succession. Summaries of plant species' responses to fire are provided by Wright et al. (1979), Bunting (1984), and Miller and Eddleman (2001). Climate conditions and fire severity also influence recovery rates and become more

important as communities approach thresholds. In the preventative stage (Fig. 21, Phases I and II, page 25) successful site recovery and predictability of response is high when plant communities contain an abundance of native species prior to burning. The initial response to fire of plant communities in relatively good condition is typically increased cover, density, and biomass of perennial grasses and perennial and annual forbs. However, this initial response is accompanied by a decrease of litter and woody plant cover, resulting in more bare ground (Quinsey 1984; Koniak 1985; EOARC, unpublished data). As developing woodland communities approach the transition between Phase II and III, or deep-rooted perennial grasses decline to fewer than 1– 2/10ft^2, it is more difficult to predict post-fire succession. The risk of failure also increases if weedy exotic species are present (see section on weeds). Increases of introduced annuals are usually not significant unless the site is in poor condition and there are few native species present to respond (Quinsey 1984, Koniak 1985). Once a community shifts to Phase III and/or native perennial grasses are no longer present in the understory, return to the pre-invasion community is unlikely without major and usually costly restoration inputs. In fully developed woodlands, desirable native species and seed pools become depleted, which potentially limits recovery after fire (Erdman 1970, Koniak and Everett 1982, Miller et al. 2000). As communities shift into the restoration stage (Figs. 21, 22c, pages 27–28) and abundance of understory vegetation and fuels become limiting, prescribed fire alone is no longer a viable management tool.

Two prescribed burns were conducted in productive mountain sagebrush sites during the late 1990's in the central Oregon pumice soil zone and in northeastern California. Western juniper woodland succession varied between Phase I and II across both sites. In the central Oregon burn, cover of bluebunch wheatgrass increased 274 percent and Idaho fescue increased 22 percent above preburn levels by the third year after fire (EOARC, unpublished data). Perennial forb cover, which is typically low on these pumice soils, did not respond significantly after fire. In the northeastern California burn, cover of Idaho fescue initially decreased by 25 percent the first year following fire and by the third year post-burn was 40 percent greater than preburn levels (Appendix 4)(EOARC, unpublished data).

Quinsey (1984) compared response of burned woodlands among dry (basin big sagebrush/Thurber needlegrass) and moist (basin big sagebrush/Idaho fescue-bluebunch) plant communities on the Crooked River National Grasslands, Oregon. Differences in grazing history, time after fire, and preburn vegetation characteristics (degree of stand closure and understory composition) among the burns discounted developing any cause and effect

relationships. However, several patterns in plant response were observed. First, dry warm sites are prone to increases in cheatgrass cover, and it appears that shrub and tree reestablishment take longer than on moist sites. Second, on wetter cooler sites, recovery of shrubs and perennial herbaceous plants occurred more rapidly and annual grasses did not compete well.

Western juniper cutting and prescribed fire combinations

As woodland development enters Phase III, fine fuels and ladder fuels are not sufficient to carry a fire with the necessary intensity to kill western juniper trees. Recently BLM managers have experimented with partial cutting of western juniper (usually one-fourth to one-half of the western juniper trees on a site) to develop sufficient fuel loads to carry fires. These methods have thus far proven successful in mountain sagebrush grassland, aspen, and riparian communities. The number of trees cut should be the minimum required to create ladder fuels to carry the fire. Increasing the number of trees cut increases costs and the potential to sterilize the soils, kill native bunchgrasses, and provide open sites for weed encroachment.

Burning in aspen for juniper control

Fire is an effective tool for restoring aspen communities that have been encroached upon by western juniper. However, the wet nature of aspen communities can create problems in applying prescribed fire. In northeastern California, western junipers greater than 13 ft in height were cut in the fall and left in an aspen stand. Fire was applied to the stand the following fall. The fallen western juniper trees carried fire through the stand, killing all western junipers and aspen trees (EOARC, unpublished data). In the third year post-fire aspen sucker density (3–10 ft tall) was 6,000 stems/acre compared to the pre-burn densities of 300 stems/acre (most of which were 18 inches tall with browsed terminal stems). The burn was rested from livestock use the first 3 years after treatment and there was little elk or deer use.

Partial cutting of western juniper to develop fuel loads to carry fire was used in aspen stands in Kiger Canyon on Steens Mountain, Oregon, in 2001 (Bates et al. 2004). Partial cutting involved felling one-third to one-half of the mature western juniper trees in decadent aspen stands. Tree canopy cover was more than 60 percent, shrub cover less than 1 percent, and perennial herbaceous cover was less than 15 percent. Prescribed fire included fall burning (October 2001) using helicopter-dropped "ping-pongs" and spring burning (April 2002) using drip torches. Partial cutting and fall burning was effective at removing remaining live western juniper trees (Bates et al. 2004). Aspen resprouting varied depending on the condition of the stand prior to

treatment. Resprouting on cut and fall-burned plots ranged from 50 stems/acre to over 5,000 stems/acre by the second year post treatment. Cutting may not necessarily be required when fall burning is applied. In several cases, fires were ignited in untreated stands where sufficient fuel provided by dead-fallen aspen trees was present to kill 80–100 percent of the encroaching western juniper. Resprouting in fall-burned plots ranged from 100 stems/acre to over 4,900 stems/acre. Fires were hot in both cut and uncut burned treatments and herbaceous and shrub layers were negatively affected. Understory response has primarily been limited to annual and perennial forbs and resprouting shrubs, included western snowberry and wax currant. Discounting cover provided by aspen, bare ground has exceeded 95 percent in the two growing seasons after fire was applied. However, the importance of fire in stimulating some species was evidenced by emergence in all heavily burned plots of long-sepaled globemallow (*Iliamna longisepala*), which is considered a rare species in the area.

Partial cutting (May 2001) followed by spring burning (April 2002) was less successful than was fall burning at removing remaining western juniper trees and seedlings. Aspen resprouting ranged from 240 stems/acre to over 2,400 stems/acre. Spring burns were cool and caused little damage to understory species. Soils were partially frozen, saturated, and there was still snow cover in shaded areas at the time of burn application. Shrub and understory composition remained largely unchanged after fire and there was no increase in bare ground.

Post-treatment management and disturbance

Post-treatment management should be part of the planning process. Introduction of livestock after burning in western juniper woodlands has not received adequate scrutiny but is one of the most important decisions resource managers and livestock owners must make. Grazing can be considered a form of disturbance that affects the rate and trajectory of plant community recovery following fire. Typically 2 years of grazing rest is prescribed following fire. This requirement has never been tested experimentally. Decisions regarding livestock reintroduction should be made based upon the response of vegetation following treatment. With slow community recovery, rest may be required beyond the standard 2-year time frame. Reintroduction of livestock within the first 2 years post-fire should not be rejected if recovery proceeds rapidly. Sites that respond rapidly may not be negatively affected by deferring grazing until after the growing season (August or later) within the first 2 years post-fire. Considerations for livestock grazing after western juniper treatment are discussed in a later section of this chapter.

Summary

There are several conclusions that can be drawn from fire research in western juniper woodlands.

- A positive plant community response after fire increases with site condition, site potential, and when the community is still in the early stages of western juniper woodland development.
- Predicting response is lowest when a site is approaching the threshold between Phases II and III.
- Risk of weed invasion and treatment failure after fire will increase with site aridity and warmer temperature regimes. Prediction of outcomes and achievement of desirable responses generally increase along a gradient of increasing moisture and decreasing temperatures in sagebrush plant communities.
- Post-fire climate conditions can influence recovery rates of plant communities and successional dynamics (e.g., plant composition).
- Climatic conditions prior to fire can affect recovery by influencing seed pools.
- The complexity (patchiness and shape) and size of a fire will also influence the potential seed rain from within and outside the fire boundary.
- Fire severity will influence plant response and seed pools within the boundary of the fire.

Chemical

Tebuthiuron and picloram are the herbicides that have received the most attention for western juniper control. Aerial application of tebuthiuron pellets applied at rates of 2.2 and 4.4 lb/acre, was unsuccessful in controlling western juniper in eastern Oregon (Britton and Sneva 1981). Although western juniper was not eliminated, the understory was significantly reduced. Thus aerial application is not a recommended practice. On an adjacent site, individual tree treatments with tebuthiuron were effective in killing trees less than 6.5 ft tall. The lack of effective western juniper control was likely due to site selection. The plant community was a low sagebrush type with heavy clay soils; clays in the surface horizon bind up the chemical, decreasing its infiltration through the soil profile and decreasing its availability for root uptake by western juniper. This study is a good example of the necessity of picking sites that will respond to treatment. Even if western juniper had been removed by some other method, the herbaceous response would be minimal as it was a low sagebrush community with low site potential.

Picloram applied to individual trees around canopy driplines was highly effective at controlling western juniper in northeastern California on a basin big sagebrush site (Young et al. 1985). Perennial grass response was limited primarily because of the increased dominance of cheatgrass

and medusahead. Attempts were made to treat annual grasses with atrazine and then seed with a perennial mix. Weed control was generally unsuccessful because litter accumulations and physical constraints imposed by standing dead trees limited herbicide application and drill seeding of perennial species.

Elsewhere, herbicide applications had mixed results at controlling juniper and piñon species. Redberry (*Juniperus pinchotii*) and alligator juniper (*J. depeanna*), both Southwestern species, have been successfully controlled using dicamba, tebuthiuron, or picloram. However, others have found little control of redberry or alligator juniper using picloram or tebuthiuron particularly when trees exceed 3–10 ft in height. Tebuthiuron application was effective at killing Utah juniper and piñon (single leaf and Rocky Mountain) in old chainings in Utah and across several sites in Arizona and New Mexico when pellets was applied at the base of the tree.

Livestock Grazing Following Western Juniper Treatment

Grazing management following western juniper control requires thorough consideration of when to reintroduce livestock after treatment. Stocking rate, duration, season of use, and how the treatment may influence livestock distribution must be considered when developing follow-up management plans. There are no set prescriptions for reintroducing grazing after western juniper control, and rightly so. Variability in site characteristics (plant association, woodland successional stage, understory composition, soils, and topography), weather, and type and intensity of control method means that no single prescription can be applied with the expectation of successful site restoration. Grazing management must remain flexible, be adaptive to changing conditions, and requires constant reassessment to achieve restoration goals. The primary goal when grazing treated areas is to permit rehabilitation of the sites' ecological functions (particularly hydrologic function and energy and resource capture). In shrub-steppe communities, this is usually best achieved by restoring the system to one dominated by perennial grasses and shrubs. Additional considerations are restoring the structural characteristics of the site, and enhancing resource capture, and improving wildlife habitat.

After western juniper competition has been removed, herbaceous plants will take time to respond. Grazing must be structured to permit short- and long-term successional response. In the short term this necessitates permitting existing plants on site to grow and produce viable seed. Significant seed production tends not to occur until the second or third year after western juniper control. Long-term considerations require

that site management permits germination and establishment of new desired individuals. Following western juniper control, some level of grazing rest or deferment will usually be required to achieve restoration goals. The amount of time required for deferment will largely depend upon conditions of the understory prior to treatment, resilience[17] of the site, and climate conditions.

An ongoing study has evaluated herbaceous plant recovery subjected to grazed and ungrazed prescriptions over four growing seasons (1999–2002) after western juniper was chainsaw cut on Steens Mountain, Oregon (EOARC, unpublished data). The study consisted of four treatments: cut grazed, cut ungrazed, woodland grazed and woodland ungrazed. Plots were short-duration grazing by cow-calf pairs for 4–5 days in early spring the first two growing seasons after cutting. Livestock were removed prior to boot stage to minimize grazing impacts to perennial grasses. Plots were not grazed in 2001 and 2002 in order to assess biomass and reproductive responses. Western juniper cutting removed overstory interference and produced significant increases in herbaceous cover, biomass, and seed production when compared to adjacent woodlands. Herbaceous response did not differ between cut-grazed and cut-ungrazed treatments as measured by cover, biomass, and density. However, grazing the cut areas did reduce perennial grass seed production when compared to the cut ungrazed treatment. This site requires rest or deferment the first several growing seasons to provide plants the opportunity to maximize seed crops and enhance opportunities for seedling establishment when environmental conditions are favorable.

Deferring grazing (after the growing season) following a fire is generally a good management practice. Western juniper control by fire will remove most of the existing nonsprouting shrubs and can potentially kill some of the herbaceous component. The level of herbaceous mortality will depend on fire intensity, fuel moisture, and amount of litter buildup. Deferment of grazing to the fall period during the first several growing seasons is probably a minimum requirement if natural recruitment is prescribed, especially in areas with a severely depleted understory. Perennial grass seed production in most cases will not be significant until the second year post-fire. Plants must be allowed time to maximize seed crop and permit seedling establishment on sites where densities of desirable plants have been depleted. Even short grazing prescriptions in early spring are detrimental

to perennial grass seed production (EOARC, unpublished data). Burned areas should probably be treated as a new seeding, requiring a minimum 2 years of rest during the growing season and possible deferment in later years.

Chainsaw cutting and proper chemical application will minimally affect understory vegetation. In the case of cutting, treatments typically occupy relatively small areas (a few acres to several hundred acres) located in large pastures, which may be several thousand acres in size. Resting entire pastures until these areas recover may be warranted biologically, but may not be practical from management and forage need perspectives. At the same time, introducing livestock too quickly after western juniper treatments may inhibit understory recovery, particularly on sites with a diminished perennial bunchgrass component and may permit dominance by weedy annuals. Cutting small areas may also result in excessive trailing by livestock in the interspace, resulting in severe utilization on the unprotected plants (Jim Buchanan, Burns, Oregon, BLM, personal communication). Western juniper cutting on this type of area should attempt to coincide with regular pasture rotations so cut areas are rested or deferred in years immediately following western juniper treatment. Grazing in late summer and fall may be permissible as plants are largely dormant during this period.

Heavy machinery will produce varying degrees of disturbance to soil surfaces but grazing management after control will be similar to cutting projects. An additional factor that must be considered in management decisions are grazing and browsing impacts by wild ungulates following western juniper control.

Economics

Little information is available to determine the economic impact of the increasing woodlands throughout the Intermountain West. Several studies have evaluated the response of forage production and big game. However, responses are usually variable depending on site condition, climate, and soils, which makes it difficult to evaluate economic return. It is also difficult to place an economic value on restoring sagebrush grassland ecosystems to proper functioning conditions. And, little has been done to determine the economic values of the possibility of reducing catastrophic fire events. Most economic uses on shrub-steppe grasslands being invaded by western juniper will be marginal at best to justify the costs of juniper removal, and will likely need to be subsidized. The greatest justification for subsidizing woodland control is the restoration of intermountain plant communities to proper functioning condition.

[17]Resilience is the ability of a site to recover to potential native vegetation, which is largely dependent upon site characteristics and climate.

Weeds

Ecology

There has been limited research on the relationships between western juniper expansion and weeds. The primary effect western juniper encroachment has on weeds is modification of the existing plant community. As trees become dominant on a site, shrubs and native herbaceous species in the understory decline, soil nutrient resources become less available, and microclimate is modified. The highest risk levels for weed invasion into pre- and post-settlement western juniper communities are in the warmer (mesic temperature regime) lower elevation sites. The risk for weeds to dominate the understory decreases at higher elevations and is low above 5,000 ft. Larger portions of woodlands are at high risk of weed invasion in the relatively warmer Mazama and John Day ecological provinces than in the High Desert and Humboldt ecological provinces.

As western juniper begins to dominate a plant community, the understory species decline (Bates et al. 2000, Miller et al. 2000) and soil resources become less available (Bates et al. 2002). The opportunity for weed establishment increases on sites with a depleted shrub and herb layer. However, reduced soil resources on a western juniper-dominated site (Phase III) may limit the abundance of introduced weeds. Weeds can dominate open stands of western juniper. During the early phase of weed invasion, species such as cheatgrass are usually most abundant beneath the south and west side of the tree canopy. On clay soils, however, medusahead can be a dominant understory layer in closed woodlands. In undisturbed closed woodlands weed abundance will fluctuate with climate but will usually have minimal influence on the site; however, the removal of trees will release soil resources (Bates et al. 2002) and result in a release of weedy species (Tausch 1999). Following fire, soil water and available nutrients generally increase, at least for short periods of time (Blank et al. 1994). Increases in nutrients, especially nitrogen, enhance the growth of cheatgrass and increases the period of dominance (McLendon and Redente 1991, Young et al. 1999). Once established, cheatgrass responds rapidly to woodland fires and shifts the seasonality of fire to the more active growing period of native perennials (Whisenant 1990). Repeated fires can simplify vegetation into a homogenous landscape dominated by exotic annuals (Young and Evans 1973, Young 1991). The availability of soil resources following a reduction in tree density can be a predictor of community invasibility (Burke and Grime 1996).

The increase in exotic annuals in piñon and juniper woodlands in Nevada resulted in dramatic increases in fire size and frequency (Young and Evans 1973, Whisenant 1990, Swetnam et al. 1999, Tausch 1999). Recent crown fires in dense piñon-juniper in the southern Great Basin (Tausch 1999, West 1999) have opened up many woodland areas, often causing them to shift from woodland to annual grassland. As western juniper canopies continue to close the potential for high-intensity crown fires will increase.

Weed response following treatment

Barney and Frischknecht (1976) identified a weedy annual stage that peaked within 3 to 4 years after a fire, followed by several stages with differing mixes of perennial grasses, forbs, and shrubs. In Utah, the change in cheatgrass cover was dramatic, ranging from 12.6 percent in 3-year-old burns to less than 1.0 percent in burns older than 22 years. A similar pattern was identified in piñon-juniper woodlands in southwestern Colorado (Erdman 1970). The pattern may be similar with chaining. Working in central Utah, Davis and Harper (1990) measured a high density of both cheatgrass and burr buttercup *(Ranunculus testiculatus)* immediately after chaining on a piñon-juniper site. By the third year after chaining, the density of both species had declined by 85 percent or more compared to the first year post-treatment values. In this study, the density of seeded perennials increased over the 3-year period. In central Oregon, cheatgrass biomass increased 4 to 6 fold (200 lbs/acre) in the first 2 years following western juniper removal by cutting (Vaitkus and Eddleman 1987). However, after 15 years, tree removal resulted in large increases in perennial grasses and a decline in cheatgrass to less than 10 lb/acre (Eddleman 2002d). Similarly a decline in cheatgrass and increase in native perennials 9 years following tree cutting was measured on Steens Mountain in eastern Oregon (EOARC, unpublished data). Cheatgrass accounted for less than 5 percent of the herbaceous biomass following fire on a north aspect at 4,000-ft elevation in the Mazama Ecological Province on Horse Ridge (EOARC, unpublished data). However, on pumice soils at elevations below 3,500 ft with minimal slope, cheatgrass readily invaded stands where western juniper had been cut or burned. In a depleted western juniper woodland in northeastern California, Evans and Young (1985) measured a dramatic increase in cheatgrass (from near 0 to 1,500 lb/acre) after controlling western juniper with picloram pellets. Cheatgrass frequency declined in the treated areas over a 7-year period, but there was a continual increase in frequency of medusahead.

However, medusahead invasion following herbicide control was not a problem where there was good cover of perennial grasses (Young and Evans 1971).

Research shows that weedy annuals, cheatgrass in particular, will usually increase immediately after trees are killed, whether it is by fire, chaining, cutting, or herbicides. Much of the research indicates that this response will be transient, or that it may not even occur. For example, Barney and Frischknecht (1976) pointed out that the annual stage may be by-passed in areas with good cover of perennial herbaceous species prior to burning. In central Oregon, Quinsey (1984) stratified the fire response of western juniper woodlands into dry and moist sites. Dry sites contained cheatgrass prior to burning, and the increase of cheatgrass following treatment persisted for 20 years in some cases. On the moist sites, perennial grasses dominated the unburned vegetation with little cheatgrass present. The moist sites did not have a fire-induced increase in cheatgrass. On the Lava Beds National Monument in northern California, cheatgrass abundance following fire directly related to site and preburn composition (EOARC, unpublished data). Cheatgrass is typically common or dominates sites below 4,500 ft in elevation but is usually less abundant above 4,500 ft, especially on north to northeast aspects.

Species of concern

Cheatgrass, although not yet abundant, had a broad distribution by the late 1800's in the Intermountain West (Stewart and Hull 1949). By the 1920's it represented an important forage resource in Nevada (Young and Evans 1989). In the 1930's the increases in fire frequency that followed cheatgrass became apparent in southern Idaho (Stewart and Hull 1949). Currently cheatgrass has become widespread at the lower elevation woodlands throughout the Great Basin.

Although cheatgrass is the weed species mentioned most frequently in the literature, it certainly is not the only weed of concern in western juniper woodlands. Another species of concern, as mentioned previously, is medusahead. Western juniper woodlands at greatest risk of medusahead invasion are primarily on clay soils. However, there are examples of infestations on medium-textured soils, so it would be a mistake to assume that only clay soils are at risk. There is presently an ongoing invasion of diffuse and spotted knapweeds (*Centaurea diffusa* and *C. maculosa*, respectively) in upland sites and Russian knapweed (*C. repens*) in the moister sites. It is also likely there are other weedy species that are a potential threat but have not yet been recognized.

Summary

Past work suggests weed response following woodland conversion projects will be site-specific and will depend heavily on the initial floristics of each plant community (Everett and Ward 1984, Koniak 1985). The presence of desirable plants is important in reducing the threat of weed invasion. The ecological site (especially where it fits along the gradient of warm-dry to cool-moist), initial floristics, and the stage of woodland development are very important factors that will influence the response of a site following thinning or total removal of trees.

Guidelines for Management

A Framework for Selecting the Most Appropriate Management Action

Asking the Right Questions

Selecting the most effective management action (including no action) should be based on the current condition of the landscape unit, taking into account the response of soil, water, flora, and fauna. Addressing the questions below will allow managers to effectively identify and set priorities, greatly increase the probability of success, and increase the ability to predict the probable outcome.

To develop a strategy for restoring communities to proper functioning condition the following questions need to be addressed.

Setting goals and objectives

1. What are the desired future conditions or how should the site look in 5, 10, 20 years?

    (Example: maximize the abundance of shrubs, grasses, and forbs suitable for the site.)

2. What vegetation changes need to occur to meet functional goals and/or habitat needs?

Clearly define the perceived problems

3. What is (are) the factor(s) affecting proper ecological function?

    (Examples: western juniper density is increasing, resulting in shrub die-off, low cover and density of desirable native grasses and herbs; relatively high proportion of bare ground, resulting in rill erosion, etc.)

Identifying (inventory) current state of the site

4. What is the stage of woodland transition (i.e., Phase I, II, or III)?

5. What is the understory composition?

6. What are the fuel characteristics?

7. What are the soil characteristics?

8. How is the site functioning with respect to:

    a. hydrologic function—erosion and infiltration;

    b. recruitment of desirable and undesirable plants;

    c. plant succession?

9. How does the site connect to the surrounding landscape?

10. Are seed sources available for desirable understory species?

11. Is seeding required? If perennial grass density is above $2/10ft^2$, probably not.

12. Is restoration feasible or practical?

13. What site components need to be restored?

What are the landscape spatial characteristics of the area to be treated with respect to:

14. Patch size;

15. Amount of edge;

16. Connectivity to other patches;

17. Distance to similar patches;

18. Landscape patch composition;

19. Current use and management activities?
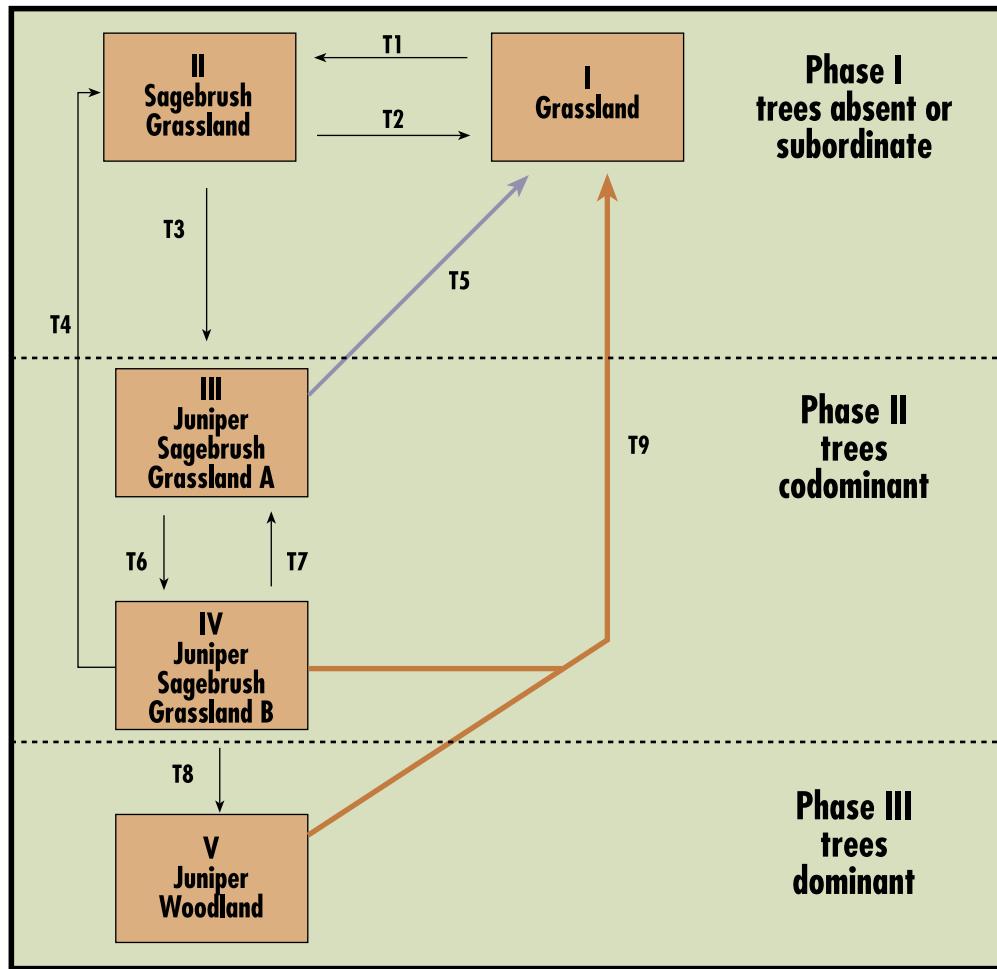
Selecting the best management action and treatment

Woodland succession within and across woodland successional phases will be determined by the type, frequency, intensity, and/or lack of disturbance (Fig. 35, 36). The best management actions will be determined by the composition of all vegetation layers of the woodland (see Appendix 4).

### Predicting the outcome of management action and treatment

20. How will populations of undesirable and desirable plants respond?

21. How will capture, storage, and runoff of water change in response to treatment?

22. Will soil erosion increase or decrease?

23. How will fauna respond?

Figure 35. Successional trajectories in the mountain big sagebrush alliance, where the potential for weed encroachment is minimal, are determined by the type, intensity, and frequency of disturbance.

Definitions for Transitions

Transition 1 (T1): *(Grassland to Sagebrush Grassland).* **Lack of fire** results in succession from an herbaceous-dominated system to co-dominance of shrubs and grass. Fire return intervals of less than 20 years will result in a herbaceous dominated community. Transitional period from grassland to shrub-steppe will vary, especially after fire, depending on the shrub seed bank, weather conditions, and site potential. Most stands will return to 20–25 percent sagebrush cover within 20–35 years (in some cases 15–60 years)(Ziegenhagen 2003).

Transition 2 (T2): *(Sagebrush Grassland to Grassland).* **Natural or prescribed fire** (spray or brush beat could be included) removes shrubs and results in grassland dominance.

Transition 3 (T3): *(Sagebrush Grassland to Juniper-Sagebrush Grassland A).* **Lack of fire.** Rate of tree encroachment varies with site, seed source, and establishment (see Fig. 24).

Transition 4 (T4): *(Juniper-Sagebrush Grassland A to Sagebrush Grassland; Juniper-Sagebrush Grassland B to Sagebrush Grassland).* **Cutting** (chainsaws, feller bunchers) of trees. Life of treatment will depend on the density of seedling junipers and saplings missed during treatment; can be as short as 15–25 years.

Transition 5 (T5): *(Juniper-Sagebrush Grassland A to Grassland)* Fuels are sufficient to carry *fire* and remove trees and shrubs. Results in return to grassland.
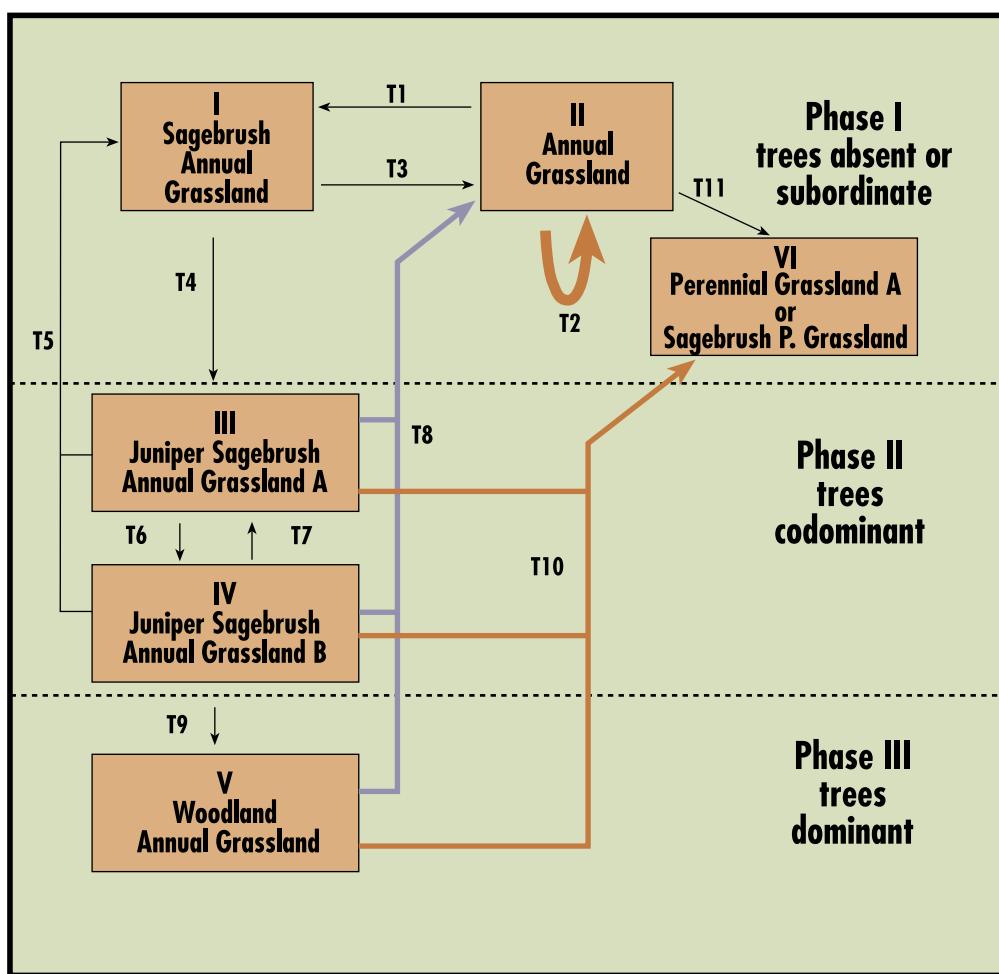
Transition 6 (T6): *(Juniper-Sagebrush Grassland A to Juniper-Sagebrush Grassland B).* **Lack of fire** results in expansion of the juniper canopy and increased tree density. Shrubs decline in cover and density. Control options may be limited to mechanical or combinations of mechanical and prescribed fire (see T9).

Transition 7 (T7): *(Juniper-Sagebrush Grassland B to Juniper-Sagebrush Grass A)* **Thinning operations** to retain mix of trees-shrubs and herbaceous layer. Often lasts for 15–25 years before re-treatment is necessary as seedling junipers and saplings are usually missed. If most trees are removed the community can return to sagebrush grassland if adequate seed source of herbaceous and shrubs are present.

Transition 8 (T8): *(Juniper-Sagebrush Grassland B to Juniper Woodland)* Community **conversion to woodland.** Shrubs lacking or few in the understory. Herbaceous layer may or may not be affected depending on depth to the restrictive layer below the soil surface. Main method of treatment remaining will be mechanical control.

Transition 9 (T9): *(Juniper Sagebrush Grassland B or Juniper Woodland to Grassland).* Lack of fuels eliminates fire as the sole method of tree removal. Removing all trees by **cutting** or **cutting plus fire** (creating a fuels base by cutting a portion of the trees to carry fire) to control remaining live trees is a management option. Treatment results in conversion to early succession community dominated by forbs and grasses. If deep-rooted perennials are fewer than $2/10\text{ft}^2$ seeding will be required.

Figure 36. Successional trajectories in the mountain big sagebrush alliance, where the presence or potential for weeds are high, are determined by the type, intensity, and frequency of disturbance. Successful treatment will require control of both trees and annual grass and seeding deep-rooted perennial grasses. Weeds are a primarily concern in Klamath and John Day provinces and below 4,500 ft and south aspects in High Desert and Humboldt provinces.

Definitions for Transitions

Transition 1 (T1): *(Annual Grassland to Sagebrush Annual Grassland)* ***Lack of fire*** may allow reestablishment of shrubs, resulting in the succession from annual-dominated system to codominance of shrubs and annual grass.

Transition 2 (T2): *(Annual Grassland)* ***Reoccurring fire*** (5 to 15 year cycle).

Transition 3 (T3): *(Sagebrush Annual Grassland to Annual Grassland)* ***Natural or prescribed fire*** (or herbicide, or mechanical) removes shrubs and results in annual grassland dominance without additional treatment for weed control.

Transition 4 (T4): *(Sagebrush Annual Grassland to Juniper Sagebrush Annual Grassland A)* ***Lack of fire*** results in juniper invasion and co-dominance of trees, shrubs, and annuals.

Transition 5 (T5): *(Juniper Sagebrush Annual A and B to Sagebrush Annual)* ***Cutting*** (chainsaws, feller bunchers). Life of treatment will depend on the density of seedling junipers and saplings missed during treatment; can be as short as 15–25 years. Will increase the production of annual grasses and increase the risk of fire.

Transition 6 (T6): *(Juniper Sagebrush Annual A to Juniper Sagebrush Annual B)* ***Lack of fire.*** Juniper continues to increase in density and shrub abundance declines.

Transition 7 (T7): *(Juniper Sagebrush Annual B to Juniper Sagebrush Annual Grassland A)* ***Thinning*** will probably increase annual weeds production resulting in an increased risk of fire. Life of treatment will depend on the density of seedling junipers and saplings missed during treatment; can be as short as 15–25 years.

Transition 8 (T8): *(Juniper Sagebrush Annual A & B and Juniper Woodland Annual to Annual Grassland)* Stand replacement ***Fire.***

Transition 9 (T9): *(Juniper Sagebrush Annual B to Juniper Woodland)* ***Lack of fire or cutting.***

Transition 10 (T10): *(Juniper Sagebrush Annual A and B, and Juniper Woodland to Perennial Grassland or Sagebrush Perennial Grassland)* ***Cutting, fire, control of weeds, and seeding*** of perennial grasses and forbs.

Transition 11 (T11): *(Annual Grassland to Perennial Grassland or Sagebrush Perennial grassland)* ***Control of weeds and reseeding*** of perennial grasses and forbs.

Management Actions

Fire

Advantages: most economical; natural process; vegetation can respond positively under the right conditions; can treat large areas; some control over intensity of fire; and usually results in the longest time period before juniper returns to the site.

Disadvantages: risk; liability; weed threat in some locations; reduction of shrubs (e.g., sagebrush, bitterbrush, mountain mahogany); tree selectivity limited; must have adequate fuels; potential nutrient losses with high intensity fires; limited climatic conditions under which prescribed fire can be used; smoke issues; urban interface.

Site Factors

Desirable

Fuels: adequate fine fuels (grasses and forbs, estimates of more than 500 lbs/acre) to carry the fire and ladder fuels (shrubs and small trees less than 3 ft tall) to kill trees over 5 ft tall.

Understory: desirable understory species are present in adequate abundance that will allow these species to quickly respond during the early post-season fire years.

Introduced annuals: absent or if present, only in small amounts relative to desirable herbs.

Stage of woodland development: South aspects Phase I and early II; North aspects Phase I and mid-II.

Undesirable

Fuels: inadequate to carry a fire under moderate conditions.

Understory: limited abundance of desirable native herbs, especially deep-rooted tussock grasses; potential seed source from undesirable species.

Soil surface: total bare ground in the tree interspace more than 50 percent, indicators of accelerated soil erosion (rills, etc).

Stage of woodland development: South aspects Phases mid II and III; North aspects late Phases late II and III.

Mechanical: chainsaws

Advantages: selective (trees removed); control the area that is treated; broad time period when treatment can be applied; minimal liability; friendly near urban interface, which may negate high costs; maintains shrubs with proper planning; little soil disturbance; not fuel limited; slash may be beneficial in restoring the site; broadcast seed beneath slash.

Disadvantages: high cost/acre; limited amount of area treated; large amounts of woody debris remains following treatment in dense woodlands; potential liability in fire protection zones adjacent to pine forests.

Mechanical: heavy machinery

Advantages: control the area that is treated; broad time period when treatment can be applied; minimal liability; friendly near urban interface, which negate high costs; maintains shrubs with proper planning; not fuel limited; slash may be beneficial in restoring the site; broadcast seed beneath slash; soil surface disturbance may enhance germination of seed broadcast prior to treatment.

Disadvantages: high cost/acre; limited amount of area treated; some mechanical equipment are limited by steepness of slope and rockiness; large amounts of woody debris remain following treatment in dense woodlands; soil disturbance or compaction.

Chemical

Advantages: Can treat areas quickly; not limited by topography; effective on trees less than 6 ft in height.

Disadvantages: Use is highly restricted on Federal lands, at least in Oregon; effectiveness of control often limited; few effective products are currently labeled for this use.

Seeding

If the density of desirable deep rooted grasses is less than 2 /10ft^2 on relatively dry western juniper sites or fewer than 1/10ft^2 on wet sites, seeding should be incorporated into the treatment. Broadcast seeding usually results in limited establishment. However, broadcasting beneath slash or just prior to disturbance of the soil surface (e.g., chaining) may increase success of establishment. Whether exotics or natives, the seed source and species selected must be adapted to the site and should be certified weed free.

Weeds

Questions to address prior to treatment:
- How will weedy annuals respond to the treatment?
- Which ecological sites are to be treated?
- What is the plant composition and present weed population or source of weed seed?
- Why have weeds invaded the site?

Uses

Ethnobotany

The Northern Paiute Indians used western juniper for food, medicine, and shelter. The leaves, small branches, and berries were boiled and the pitch skimmed off the surface to treat colds, sore throats, venereal disease, kidneys, and boils (Couture 1978). The berries are bitter but nutritious and were eaten during food shortages (Olsen 1967, D'Azevedo 1986). The berries were pounded and boiled to remove some of the bitter resinous taste. The white inner bark was stripped and pounded into meal and eaten to hold off starvation. Western juniper pitch was used on baskets to make watertight containers. The pitch was put on a flat stick, held over the fire, and then rapidly applied to the interior (Kelly 1932). Utah juniper was the favored wood used for sinew-backed bows by Indians in western Nevada and the Owens Valley of California (Wilke 1988). Limited evidence suggests that western juniper may have been used for bow staves in northern California and possibly in southern Oregon. However, western juniper was used to make bows for boys, rarely used by the men, who preferred yew or mountain mahogany. Several wickiups built out of western juniper also were found in central Oregon (Polk 1979). The wickiups consisted of western juniper branches used as poles leaning against the branch of a large western juniper tree. It appears that bark was used for wall covering. Evidence also suggests winter houses in the Great Basin were located in dense cedar or piñon thickets on sandy, gently sloping hillsides with a southern exposure, protected from the winter elements.

Current Uses

Early settlers used western juniper for fenceposts and firewood. In the 1920's, wood products industries in eastern Oregon and northeastern California began testing the use of western juniper wood for products including pencil stock. Formal western juniper wood products research began at Oregon State University in the late 1940's with trials to evaluate the service life of treated and untreated posts (Miller 1986). Later studies in the 1950's researched western juniper extractive oils, use of juniper wood for composites, and kiln drying the wood (Swan and Connolly 1998). The wood can be successfully kiln-dried for use in a variety of wood products (Swan 1997, Swan and Connolly 1998). There was a resurgence of interest in western juniper wood products research and trials in the 1990's, resulting in part from the drastic reduction in Federal timber sales in the early 1990's. Basic physical and mechanical testing is being completed by the University of Montana's Wood Products Laboratory. To date, products include firewood, chips for particle-flake board and animal bedding, decking, interior paneling, doors, cabinetry, rustic furniture, picture frame molding, small gifts, Christmas decorations, and the female cones are used as flavoring for gin (Swan 1997). Another use is commercial firewood where wood is hauled to metropolitan areas and sold for over $200/cord. Western juniper biomass is also used in some areas for energy fuel (by biomass combustion). However, its use for energy is limited by terrain, proximity to good roads that permit access to wood chippers and semi-trucks to haul the chips, and distance to the energy plant. The largest commercial user of western juniper between 2000 and 2004 was Honey Lake Power in Wendell, California, which uses a number of different subcontractors to harvest, chip, and transport chipped material to the plant. It is estimated that well over 100,000 tons (dry weight) have been harvested and chipped for biomass between 2000 and 2003 from at least 10,000 acres. A substantial amount of western juniper boughs and berries, estimated at minimum in the hundreds of tons, is also harvested seasonally for the holiday wreath industry (L. Swan, US Forest Service, Klamath Falls, Oregon, personal communication).

Commercial uses of western juniper will probably not make significant contributions to plant community restoration in the western juniper ecosystem on a landscape basis without financial incentives to harvest. In most cases there are cheaper wood fiber substitutes readily available. However, as the biomass energy plant in California demonstrates, regulatory changes, such as the California Renewable Portfolio Standard, which mandates a higher percent of purchased power to come from renewable sources (e.g., biomass, wind, solar, and geothermal), combined with incentives such as tax breaks could help increase commercial use of juniper and number of acres treated. In addition, many small businesses have demonstrated their interest in using this species, and on a subregional basis have proven to be valuable partners for landowners and land managers as they may help to increase the number of acres treated and at the same time make positive contributions to rural economic conditions.

Gedney et al. (1999) estimated wood volume in western juniper woodlands to be 418 million ft^3 in Oregon. A more recent inventory of western juniper in eastern Oregon indicates Gedney's estimates are low (personal communication USDA Pacific Northwest Forest Service, Portland, Oregon) An additional 49 million ft^3 are estimated to occur in eastern Oregon mixed conifer forests, which according to the wood products industry has the best commercial potential because of its form and access (Swan 1997). This does not include western juniper woodlands in California or Idaho. Cubic-foot volume equations and tables have been developed for western juniper (Chitter and MacLean 1984). Harvest methods are generally done with conventional logging equipment—chainsaws and rubber-tired skidders (Swan 1997, Swan and Connolly 1998). Chipping operations have also used tractor-mounted shears.

The biggest barrier to the harvesting of juniper for wood products is cost. Trees have numerous large limbs, average volume/acre is low, terrain is often rocky, and access limited. Most trees will not make saw logs except on some productive sites. Juniper trees also frequently have bark pockets, large trunk-swell at the base, and stem rot in older trees. Waste needs to be sold for chips or hog fuel to help make up expensive harvest costs. Reach Corporation in Klamath Falls has been successful in producing products (animal bedding, particle board) from otherwise waste material generated from western juniper harvesting for lumber and from restoration projects.

Desirable characteristics of western juniper wood are:

- Richly colored
- Aromatic
- Surfaces well during milling
- More stable in shrink/swell than Douglas-fir and ponderosa pine
- Glues and finishes well
- Heartwood more durable than any other northwestern species for fence posts

Combined with restoration efforts, harvesting western juniper for wood products may yet prove profitable. However, it is doubtful that the amount of harvested western juniper for wood products will approach the levels that are required for plant community restoration in the western juniper ecosystem. Several websites describe wood products (including www.westernjuniper.org).

Knowledge Gaps

A great deal has been learned about the ecology, biology, history, and management of western juniper over the past several decades. However, not all questions have been answered in some areas somewhat limiting our ability to manage western juniper on an ecosystem basis. Gaps in our knowledge are:

Biology and Ecology

- Environmental factors that influence western juniper cone production and annual seedling germination and establishment
- Insects, parasites, and diseases that affect western juniper
- Natural mortality rates
- Quantitative indicators identifying abiotic or biotic thresholds that are crossed during juniper woodland development
- Effects of western juniper on subsurface water flow and soil moisture storage

- Long-term impacts of woodland succession and treatment on soil nutrients and development
- Factors that effect thresholds when seeding is required across different ecological sites
- Quantification of thresholds when seeding is required across different ecological sites

Management

- Inventories and mapping of old-growth western juniper
- Limited information on the interactions of site potential, ecological condition, and harvest methods on influencing succession, hydrology, mineral cycling, erosion, soil crusts, soil fertility, and wildlife populations
- The amount of nutrients lost through burning or removal of western juniper
- The effects and interactions of environmental factors with tree removal that may affect stream and spring flows
- Elements for successful seedings

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Appendixes

Appendix 1. Western Juniper Old-growth Cover Type

Species: *Juniperus occidentalis*—western juniper

Description:

Western juniper is usually the only tree species, with the exception of scattered trees growing with ponderosa pine (*Pinus ponderosa*). Stands exhibit considerable diversity in structure and composition, varying from open-shrub tree savannas to nearly closed-canopy woodlands. However, tree canopy cover in the majority of stands is usually less than 20 percent. Ages are usually mixed with little to no recruitment in closed stands. Very old stands usually contain standing and downed dead trees that can persist for several hundred years, especially on dry sites and where down trees do not come into contact with soil. As trees mature (usually over 150 years), their inverted cone shaped canopy becomes increasingly nonsymmetrical in appearance with rounded tops and spreading canopies that may become sparse and contain dead limbs or spike tops. The bark on the trunk becomes deeply furrowed, fibrous (compared to scaly in younger trees), and can turn reddish in color. Lower branches may be very large (more common in open stands), and branches are covered with bright green arboreal fruticose lichens (*Letharia columbiana* and *L. vulpia*). The cambium layer may also die around portions of the tree trunk, leaving only a narrow strip connected to a single live branch.

Understory composition and stand structure are highly dependent on the ecological site, especially soils. In low sagebrush communities stand structure is a tree shrub savanna with tree and shrub canopies usually less than 20 percent. The diagnostic grass species on these shallow soil sites is Sandberg bluegrass (*Poa sandbergii*). These stands can vary from small scattered communities surrounded by mixed conifers in the Blue Mountains to expansive communities located on low sagebrush tablelands in the High Desert, Klamath, and Humboldt ecological provinces. On sedimentary soils understory vegetation is often sparse. Less common are stands that occupy deeper soils (i.e., Juniper Mountain, Oregon). On these sites tree canopies will generally vary from 30 to 40 percent on south and west aspects and exceed 50 percent on north aspects. Shrubs such as mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) are usually sparse and the abundance of herbaceous vegetation is determined by soil depth.

Historically humans have had minimal effects on these stands. Grazing has generally had little effects in these communities due to limited water (Mazama Province) and sparse forage. In the early 1900's and 1930's, limited number of trees were cut by homesteaders for firewood and posts; some areas were harvested for fence posts by the Civil Conservation Corps. More recently, firewood cutting (i.e., Modoc Plateau, California), selective cutting for high-quality furniture, cutting for urbanization, and landscaping decorations, and off-road vehicle use have impacted these stands.

The most extensive stands occupy pumice soils in the Mazama Province. In adjacent ecological provinces, stands occupy soils that are usually shallow, rocky, and high in clay content. Soil temperature regimes are usually mesic and frigid. Old-growth western juniper can be found throughout this species' range. In the southern portion of its range it hybridizes with Utah juniper (*Juniperus osteosperma*) (i.e., Jackson Mountains, Nevada, and along the California border south of Adel, Oregon).

Appendix 2. Summary Results from Research Investigating Response of Western Juniper to Various Treatments.

Treatment method	Study period	Plant community	Location	Results	Citation
Cutting	1982–2000	ARTRT/AGSP-FEID, ARAR/ AGSP-FEID,	Prineville, OR vicinity	**Cut:** large increases in cover and density of shrubs and perennial plants. **Uncut woodland:** little change in herbaceous layer but declines in shrub cover and density on the two lower sites.	Eddleman 2002d
Cutting & slash scatter, seeding	1987–2000	ARTRV/FEID-AGSP		**Seeding:** selected species established more successfully under juniper slash cover in average and above-average precipitation years, but not in dry years.	Eddleman 2002b
Cutting, rollerpunch slash scatter and seeding	1991–2000	ARTRT/STTH ARTRV/AGSP-STTH		**Wet year seeding:** roller punching and slash covering 0–25% adequate for seedling establishment. **Dry year seeding:** roller punching and slash covering more than 50% was beneficial for seedling establishment.	Eddleman 2002 a, c
Cutting	1982–1984	ARTRT/AGSP-FEID, ARAR/ AGSP-FEID, ARTRV/FEID-AGSP	Prineville, OR vicinity	Above-ground biomass increase by about 100% on deep soil sites and by nearly 300% on shallow soil sites.	Vaitkus and Eddleman 1987
Cutting	1984–1986	PIPO/FEID	Prineville, OR vicinity	**Pine growth:** Basal growth of ponderosa pine was not affected by juniper removal. This 2 year study probably not sufficient to assess response of pine to juniper removal. **Understory:** increased productivity by 50 % after 2 years.	Rose and Eddleman 1994
Cutting	1992–1995	ARTRV/AGSP ARTRV/FEID	Page Ranch, Grant County, OR	Increases in cover and density of squirreltail and perennial forbs. Response of Idaho fescue and bluebunch wheatgrass was minimal indicating their response will take longer.	Eddleman and Miller 1999

AGSP—bluebunch wheatgrass; ARAR—low sagebrush; ARTRT—basin big sagebrush; ARTRV—mountain big sagebrush; ARTRW—Wyoming big sagebrush; FEID—Idaho fescue; PIPO—ponderosa pine; POSA—Sandberg bluegrass; POTR—quaking aspen; STOC—western needlegrass; STTH—Thurber needlegrass; SYOR—snowberry.

Appendix 2. Summary Results from Research Investigating Response of Western Juniper to Various Treatments, continued.

Treatment method	Study period	Plant community	Location	Results	Citation
Cutting	1991–1993	ARTR/STTH-AGSP	Steens Mt, OR	**Biomass:** 900% increase after 2 years. **Herbaceous cover:** 350 % greater in cut versus woodland controls. **Diversity:** twice as great in cut compared to woodland plots. **Soil water:** significantly greater in cut treatment versus woodland controls. **Soil N:** nitrogen availability mainly controlled by inter-annual weather patterns and less affected by treatment. In first year after cutting, a dry year, N more available on cut areas. Juniper debris tended to suppress N mineralization in soils. **Soil organic carbon:** increased in soils under juniper debris.	Bates et al. 1999, 2000, 2002
Thinning with fall & spring burning	2000–2002	POTR/SYOR/STOC-FEID	Steens Mt, OR	**Herbaceous cover:** severely depleted in fall burn plots on spring burn plots little measurable loss of plants **Aspen:** response has varied depending on condition of stand prior to treatment. Monitoring will continue. **Shrubs:** shrub species capable of resprouting were not negatively affected by fire. 100% loss of sagebrush in fall burns. **Rare plants:** globemallow responded favorably to hot fall burns.	Bates and Miller (ongoing study)

AGSP—bluebunch wheatgrass; ARAR—low sagebrush; ARTRT—basin big sagebrush; ARTRV—mountain big sagebrush; ARTRW—Wyoming big sagebrush; FEID—Idaho fescue; PIPO—ponderosa pine; POSA—Sandberg bluegrass; POTR—quaking aspen; STOC—western needlegrass; STTH—Thurber needlegrass; SYOR—snowberry.

Appendix 2. Summary Results from Research Investigating Response of Western Juniper to Various Treatments, continued.

Treatment	Year	Community	Location	Results	Source
Cutting	1991–1997	ARTR/STTH-AGSP	Steens Mt, OR	**Biomass:** 900% greater in cut versus uncut woodland plots after 6 years. **Herbaceous cover:** 350% increase **Perennial grass density:** Increased 233–300% from pre-cut conditions. **Annual grass:** large increase under juniper debris and around tree stumps. To reduce annual grass component would necessitate removal of juniper debris.	Bates et al. 1998, EOARC, unpublished data
Cutting and winter slash burning;		ARTR/STTH-AGSP	Steens Mt, OR	**Cover:** after initial decline, herbaceous cover under debris has not differed from unburned intercanopy but has been greater than under unburned debris.	
1. Burned with wet-frozen soil	1997–2002			**Density:** little loss of perennial grasses first year after burn. Density has increased in subsequent years.	EOARC, unpublished data
2. Burned with dry soils	1999–2002			Under burned debris there was large initial decline in cover and density of perennial grasses compared to unburned debris and intercanopy zone.	EOARC, unpublished data
Cutting	1994–1999	ARTRV/FEID-AGSP	Steens Mt, OR	**Cover:** perennial grass cover in the late successional woodlands doubled 5 years after treatment. In the closed canopy woodland and mid-successional woodland there was little change in perennial grass and other understory cover. Shrub cover increased significantly and bareground declined. The varied understory response may also have been a product of differential livestock grazing, with some areas receiving greater use than others.	Miller and Bates 2001
Fire	various	ARTRT/STTH (dry site) ARTRV/FEID-AGSP (wet site)	Crooked River National Grassland, OR	Dry sites are prone to increases in cheatgrass cover and it appears that shrub and tree re-establishment take longer than wet sites. Wet sites—recovery of shrubs and perennial herbaceous plants occurs more rapidly and annual grasses are present.	Quinsey 1984

AGSP—bluebunch wheatgrass; ARAR—low sagebrush; ARTRT—basin big sagebrush; ARTRV—mountain big sagebrush; ARTRW—Wyoming big sagebrush; FEID—Idaho fescue; PIPO—ponderosa pine; POSA—Sandberg bluegrass; POTR—quaking aspen; STOC—western needlegrass; STTH—Thurber needlegrass; SYOR—snowberry.

Appendix 2. Summary Results from Research Investigating Response of Western Juniper to Various Treatments, continued.

Treatment method	Study period	Plant community	Location	Results	Citation
Fire	1995–1998	ARTRV/FEID-AGSP	Cedar Creek, CA	Over 80% of the trees were killed, Idaho fescue decreased in the first year but increased from 8 to 12% cover in the second growing season following fire. Perennial forbs significantly increased in the first year, and were not different in the second growing season.	EOARC, unpublished data
Chaining and seeding	1960's	ARTRV/AGSP	Silver Lake, OR	Chained areas had successful seedings of crested and Siberian wheatgrass. Seedings were unsuccessful in untreated woodlands	Lechenby and Toweill 1983
Chemical— tebuthiuron	1975–1978	ARAR/FEID-Posa	Harney County, OR	Aerial application did not effectively control juniper but resulted in severe depletion of the understory. Individual tree control was effective at controlling juniper <2m tall.	Britton and Sneva 1981
Chemical—picloram	1975–1982	ARTRW/STTH-AGSP	Lassen County, CA	Hand application was effective at killing trees. Understory response was primarily increases in cheatgrass and medusahead. Additional chemical control and seeding of perennial grasses was necessary to reduce annual grass presence.	Evans and Young 1985, Young et al. 1985

AGSP—bluebunch wheatgrass; ARAR—low sagebrush; ARTRT—basin big sagebrush; ARTRV—mountain big sagebrush; ARTRW—Wyoming big sagebrush; FEID—Idaho fescue; PIPO—ponderosa pine; POSA—Sandberg bluegrass; POTR—quaking aspen; STOC—western needlegrass; STTH—Thurber needlegrass; SYOR—snowberry.

Appendix 3. Benefits and Comments Extracted from BLM Reports on Western Juniper Treatments Conducted in Oregon.

Site, county, date	Juniper control method	Acres	Benefits and comments	Citation
Camp Creek, Crook, 1964	Chaining (single)	3,000	Forage production increased from 22 to 7 acres/AUM. Young trees survived and <50 % of large trees removed.	Winegar and Elmore 1977
Willow Valley 1969	Chaining (double)	1,400	Increased forage production 219 lb/acre. Shrub seeding not successful. Good control of trees except small ones.	Winegar and Elmore 1977
Ninemile 1969	Chaining (double)	1,100	Shrub seeding not successful. Response of understory good.	Winegar and Elmore 1977
North Harpold	Chaining (double)	180	Wildlife focus. Increased herbaceous production and deer use. Required further hand cutting to remove small trees	Winegar and Elmore 1977
Spring Creek	Chaining (double)	300	Wildlife focus. Increased herbaceous production and deer use. Required further hand cutting to remove small trees.	Winegar and Elmore 1977
Sheep Mtn, Crook, 1971	Chaining (double)	300	Increased herbaceous production 646 lb/acre. Shrub seeding not successful. Numerous small trees survived.	Winegar and Elmore 1977
Wand Lake, Lake, 1972	Chaining (double)	500	Increased forage production 275–440 lb/acre.	Winegar and Elmore 1977
Bear Creek, Crook, 1975–1978	Chainsaw	12,000	Increased herbaceous production, reduced bare ground from 61 to 28 %, minimal site disturbance, and increased understory structure resulting in improved small mammal habitat.	Elmore 1977

Appendix 4: Pre- and Post-treatment Results for Two Different Ecological Sites.

Site Description

Plant association:
Mountain big sagebrush/Thurber needlegrass

Pretreatment community:
Western juniper/Sandberg bluegrass

Phase: III

Slope: 10%

Aspect: West

Elevation: 5,200 ft

Soils: Typic argixeroll,
16–24 inches to cemented ash layer

Location: Cucamonga, Steens Mt

Treatment

Cut (chainsaw) and drop fall of 1991

Pre-treatment composition

Juniper = 20% cover, 80 trees/ac

Sagebrush = <1% cover

Perennial herbaceous cover = 2%

Deep-rooted perennial grasses = 2 plants/10ft^2

Post-treatment composition
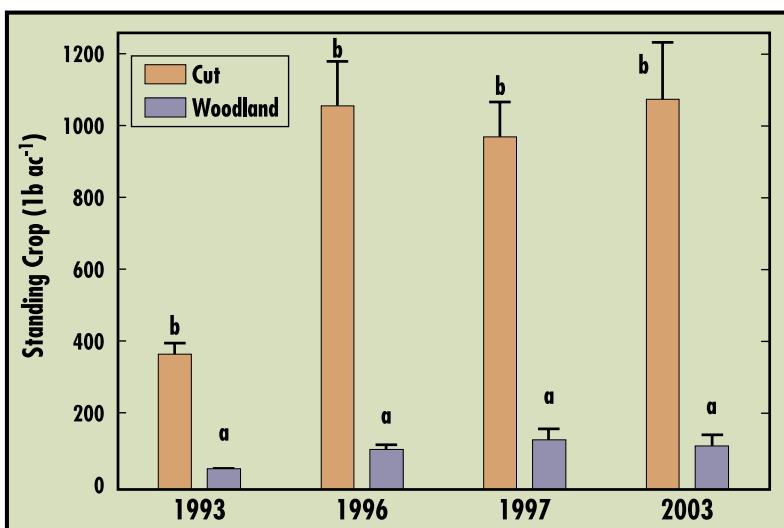
1993

Juniper = 0% cover

Sagebrush = <1% cover

Perennial herbaceous cover = 11%

Deep-rooted perennial grasses = 5/10ft^2

Appendix 4: Pre- and Post-treatment Results for Two Different Ecological Sites, continued.

Site Description

Plant association:
Mountain big sagebrush/Idaho fescue

Pre-treatment community:
Western juniper/mountain big sagebrush/Idaho fescue

Phase: II
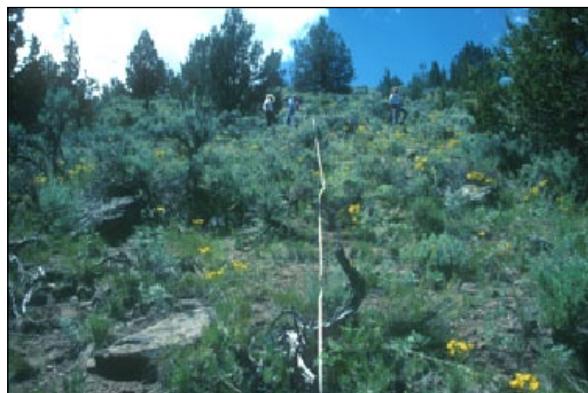
Slope: 20%

Aspect: north

Elevation: 5700 ft

Soils: loamy-skeletal, mixed frigid, Typic haploxerolls, 20 to 40 inches deep

Location: Cedar Creek, Alturas, CA

Treatment

Fall burn 1995

Pre-treatment composition

Juniper = 12% cover, 115 trees/ac

Sagebrush = 11% cover

Perennial Grasses = 9.1% cover

Perennial Forbs = 8.5% cover

Annual Grasses = 0.1% cover

Post-treatment composition

1998
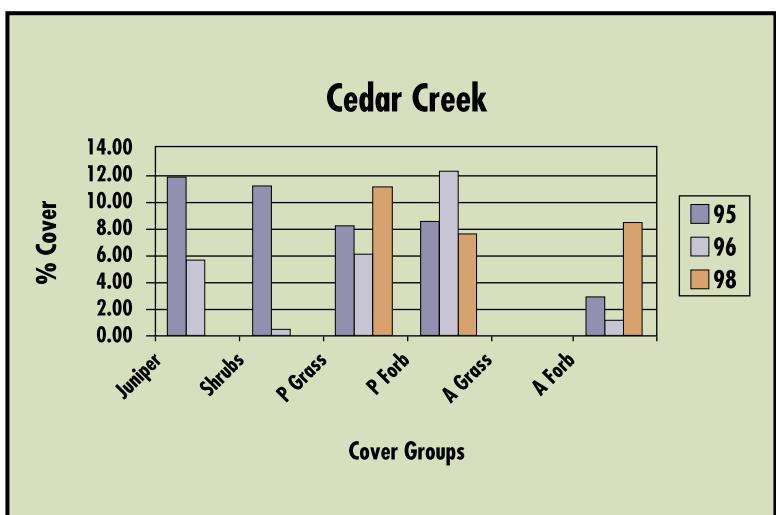
Juniper = 0% cover

Sagebrush = 0.1% cover

Perennial grasses = 11.5% cover

Perennial forbs = 7.7% cover (12.2% in 1996)

Annual grasses = 0.1% cover

Cedar Creek

Dave Pacioretty standing in an old-growth stand on top of Juniper Mountain in Harney and Lake counties, Oregon.

Western juniper has occupied its current range for several thousand years.

BIOLOGY, ECOLOGY, AND MANAGEMENT OF
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