The Ecology, History, Ecohydrology, and Management of Pinyon and Juniper Woodlands in the Great Basin and Northern Colorado Plateau of the Western United States

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Abstract

This synthesis reviews current knowledge of pinyon and juniper ecosystems, in both
persistent and newly expanded woodlands, for managers, researchers, and the interested
public. We draw from a large volume of research papers to centralize information on these
semiarid woodlands. The first section includes a general description of both the Great Basin
and northern Colorado Plateau. The ecology section covers woodland and species life
histories, biology, and ecology and includes a detailed discussion of climate and the potential
consequences of climate change specific to the Great Basin and Colorado Plateau. The history
section discusses 20,000 years of woodland dynamics and geographic differences among
woodland disturbance regimes and resilience. The ecohydrology section discusses hydrologic
processes in woodlands that influence soil conservation and loss; water capture, storage, and
release; and the effect that woodland structure and composition have on these processes. The
final section, restoration and management, covers the history of woodland management, the
different methods used, the advantages and disadvantages of different vegetation treatments,
and posttreatment vegetation responses. We also discuss successes and failures and key
components that determine project outcomes important for consideration when restoring
ecosystem function, integrity, and resilience.

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Cover image—Singleleaf pinyon arrived in the very north end of Owens Valley at the base of
the White Mountains near the California and Nevada border around 9,000 years ago. It slowly
moved north along the east slope of the Sierras arriving to its northwestern most boundary
near Pyramid Lake 300 years ago. (Photo from the Benton Range looking towards the
woodland belt on the north end of the White Mountains; by Rick Miller.)

This publication reports research involving pesticides. It does not contain recommendations
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pesticides must be registered by appropriate State and/or Federal agencies before they can be
recommended.

CAUTION: Pesticides can be injurious to humans, domestic animals, desirable plants, and fish
or other wildlife—if they are not handled or applied properly. Use all pesticides selectively and
carefully. Follow recommended practices for the disposal of surplus pesticides and pesticide
containers.

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Acknowledgments

Table of Contents

INTRODUCTION

In 1879, John Muir traveled across the rugged young State of Nevada and wrote, "The lower ranges and the foothills and slopes of the higher [mountains] are roughened with small scrubby junipers and nut pines ... Nearly every mountain in the State is planted with it from near the base to a height of from 8 thousand to 9 thousand feet above the sea." The woodlands of scrubby juniper and pine that Muir wrote about are a major component of the Great Basin and Colorado Plateau, shifting in abundance and distribution over thousands of years (fig. 1-1).

Semiarid woodlands, composed of pinyon and/or juniper trees, cover many square miles of the Great Basin and northern Colorado Plateau. Extending across 9 level III ecoregions (fig. 1-2; USDA Forest Service 2018) and 15 Major Land Resource Areas (MLRAs) (fig. 1-3; table 1-1), this vast area is characterized by a complex set of landscapes where climate, geology, soils, and topography vary at multiple scales—resulting in continual changes in potential vegetation, disturbance regimes, resilience, and resistance to invasive annuals (see Glossary for definitions). These semiarid woodlands are typically dominated by a single pinyon or juniper species—or a combination of both—with species composition determined largely by the abundance and distribution of precipitation (winter versus summer), temperature, and soils.

Figure 1-1—When John Muir traveled across Nevada in 1879, he observed scrubby junipers and nut pines occupying the lower slopes and foothills of higher mountains. Since Muir's travels, there has been considerable debate as to the extent of infill and expansion of these woodlands across the Great Basin and Colorado Plateau—a topic thoroughly discussed in this synthesis. Northern Great Basin. (Photo by Rick Miller, Oregon State University.)
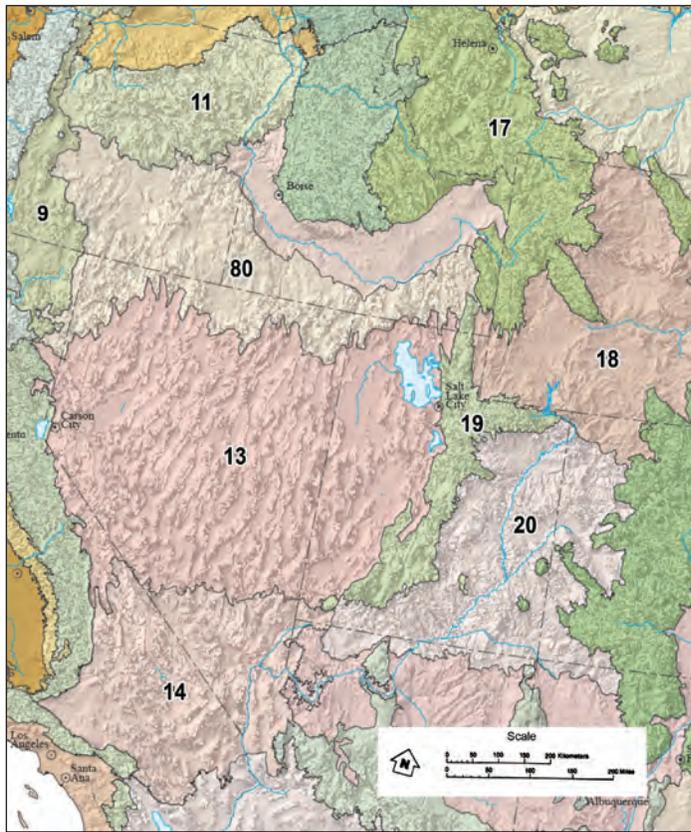
Figure 1-2—This synthesis covers pinyon and juniper woodlands across 9 level III ecoregions, which included parts or all of the Eastern Cascade slopes (#9), Blue Mountains (#11), Central Basin and Range (#13), mountains of the Mojave Basin and Range (#14), southwest end of the Middle Rockies (#17), mountains of the Wyoming Basin (#18), Wasatch and Uinta Mountains (#19), Colorado Plateau (#20), and Northern Basin and Range (#80) (USDA Forest Service 2018).
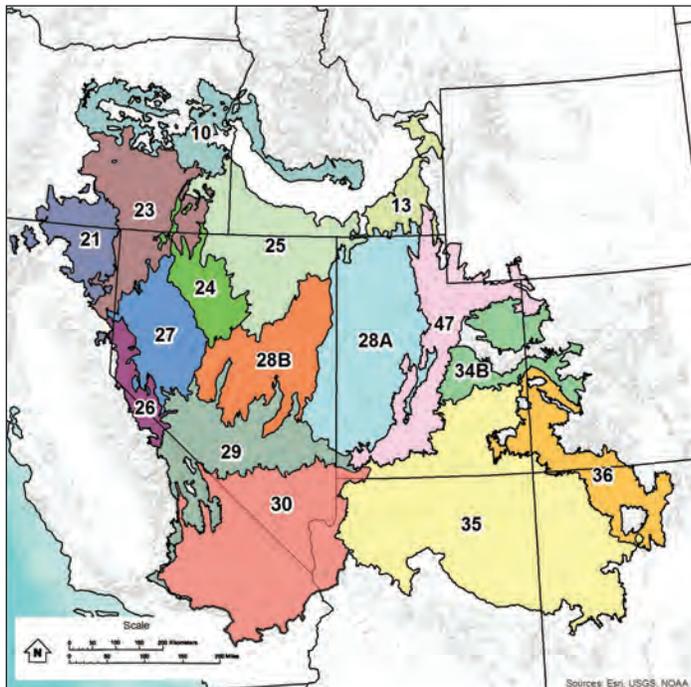
Figure 1-3—Major Land Resource Areas (MLRAs) within the geographic boundaries of this synthesis extend across the ranges of both western juniper and Utah juniper: Central Rocky and Blue Mountain Foothills (#10), Eastern Idaho Plateaus (#13), Klamath and Shasta Valleys and Basins (#21), Malheur High Plateau (#23), Humboldt Area (#24), Owyhee High Plateau (#25), Carson Basin and Mountains (#26), Fallon-Lovelock Area (#27), Great Salt Lake Area (#28A), Central Nevada Basin and Range (#28B), Southern Nevada Basin and Range (#29), Mojave Desert (#30), Warm Central Desert Basins and Plateaus (#34B), Colorado Plateau (#35), the northern Southwestern Plateaus, Mesas, and Foothills (#36), and Wasatch and Uinta Mountains (#47). (Map by Dave Nagel, USDA NRCS 2006).

Table 1-1—Major Land Resource Area (MLRA) names and identification numbers (see map fig. 1-3) included in this synthesis. For each MLRA the size of land area, States listed in descending order of proportion of area it covers, most common elevation range (extreme), geology, common soil orders, range of average annual precipitation (extremes), range in temperatures, and soil temperature regime are described. The synthesis only covers the sagebrush, and pinyon and juniper communities, within these MLRAs. MLRAs 8, 10, and 11 are located in the Northwestern Wheat and Range Region. The remaining MLRAs are located in the Western Range and Irrigated Region. (Descriptions derived from USDA NRCS 2011; and Shane Green, Randy Lewis, and Sara Quitzberg, Soil Scientists, USDA NRCS, Salt Lake City, Utah, personal communication, 2015.)

MLRA (id #)	Area (mi²)	States	Elevation ft	Geology	Soils	PPT inches	Temp °F	Soil moisture	Soil temp
Blue Mt Foothills (10)	17,515	OR ID	1,300–6,600	Basalt alluvium sedimentary	Mollisols Aridisols	8–16 (41)	36–53	Aridic-xeric	Mesic frigid cryic
Eastern Idaho Plateaus (13)	7,270	ID	4,500–6,600 plateaus/foothills 6,600–9,500 mts	Loess underlain by sedimentary & some basalt	Mollisols	12–25 (48)	36–48	Xeric	Frigid Cryic
Klamath Valleys (21)	11,495	CA OR	2,600–4,600 (> 7,000)	Basalt rhyolite andesite	Mollisols	12–30 (9; 30–58)	39–52	Xeric	Mesic frigid (cryic)
Malheur High Plateau (23)	22,896	OR NV CA	3,900–6,900 (> 9,000)	Basalt & andesite	Aridisols & Mollisols	6–12 (> 50)	39–52	Aridic-xeric	Mesic frigid, cryic
Humboldt Area (24)	12,680	NV OR	3,950–5,900 (> 8,850)	Alluvium (some andesite & basalt)	Entisols Inceptisols Mollisols	6–12 (40)	38–53	Aridic	Mesic (frigid)
Owyhee High Plateau (25)	28,930	NV ID OR UT	3,000–7,550 (> 9,800)	Andesite basalt rhyolite	Aridisols Mollisols	7-16 (> 50)	35–53	Aridic-xeric	Mesic frigid
Carson Basin & Mts (26)	6,520	NV CA	3,900–6,550 (13,100)	Granitic andesite basalt	Aridisols Mollisols	5–36	37–54	Aridic-xeric	Mesic (frigid & cryic)
Fallon-Lovelock (27)	12,565	NV CA	3,300–5,900 (<7,800)	Alluvium andesite basalt	Aridisols Entisols	5–10 (19)	43–54	Aridic	Mesic
Great Salt Lake (28A)	36,775	UT NV ID	4,900–6,550 basin 6,550–11,900 mts	Carbonate (north) andesite basalt (south) Alluvium & lakebed deposits	Aridisols Entisols Mollisols	5–12 (basins) 8–36 (mountains)	34–52	Aridic-xeric (ustic)	Mesic Frigid (cryic)
Central NV Basin & Range (28B)	23,555	NV	3,950–6,560 basin 6,560–11,150 mts	Playa lakebed deposits, old-sedimentary	Aridisols Entisols Mollisols	5–12 (49)	39–53	Aridic-xeric	Mesic frigid, cryic
Southern Nevada Basin and Range (29)	26,295	NV CA UT	1,950–5,600 valleys 9,400 mts	Andesite basalt alluvium	Aridisols Entisols Mollisols mt	3–12 (12–29)	28–72	Aridic-xeric	Mesic
Mojave Desert (30)		CA NV AZ UT	-282–3,950	Alluvium granitic igneous	Aridisols Entisols	2–8 (> 37)	43–76	Aridic	Thermic
Warm Central Desert Basins (34B)	12,850	UT CO	4,100–7,500	Shale sandstone Residual basin floor materials	Aridisols Entisols (Mollisols)	6–10	41–54	Aridic	Mesic
Colorado Plateau (35)	71,735	AZ UT NM CO	4,250–8,000 (> 10,000)	Shale sandstone limestone dolomite volcanic	Alfisols Aridisols Entisols Mollisols	6–18 (< 5–30)	36–66	Aridic or ustic	Mesic frigid
Southwestern Plateaus, Mesas, and Foothills (36)	23,885	CO UT (excluding NM)	4,600–8,500 (> 9,000)	Sedimentary shale sandstone	Alfisols, Inceptisols, Mollisols, Entisols, Aridisols	12–20 (8–31)	37–56	Ustic-aridic	Mesic frigid
Wasatch and Uinta Mts (47)	23,825	UT WY CO ID	4,900–13,500	Volcanic	Aridisols Entisols Inceptisols Mollisols	15–30 (6–10 basins, 10–15 foothills)	30–58	Aridic-xeric-ustic	Frigid (mesic & cryic)

In addition to spatial variation, woodland complexity also varies with time. Significant fluctuations in woodland expansion, contraction, and infill, in addition to changes in structure and composition, have been occurring for tens of thousands of years. However, recent changes attributed to the interaction of natural factors—such as climate with more recent anthropogenic variables—can be difficult to sort out. Twentieth century increases in tree densities (infill) and expansion have resulted in both land management concerns and considerable debate as to the extent of increase, and the primary factors causing these changes. There is also considerable concern about the recent regional die-offs of pinyon and juniper and the consequences of climate change and invasive weedy species to the future of these ecosystems. Managers, private landowners, and scientists struggle with concerns over fire severity and frequency of occurrence, reductions in understory vegetation and forage, changes in wildlife habitat (especially for sagebrush obligates), reduced resistance to invasive species, altered ecohydrology processes, and the reduction in resilience and ecosystem function. As a result, private landowners and public agencies have treated large areas across the Interior West since World War II by removing trees with prescribed fire and/or mechanical and chemical methods.

But tree removal is not always successful for solving the above concerns. Successional trajectories following tree-removal projects have ranged from progression toward native shrub-steppe or shrubland sagebrush communities to large increases in invasive annuals (Miller et al. 2013). And restoration efforts through tree removal have not always separated young postsettlement woodlands from persistent old-growth woodlands. Successful management of pinyon and juniper woodlands requires an ecosystems approach with careful evaluation of restoring ecosystem integrity, function, and resilience (Benson 2012; Boyd et al. 2014; Miller et al. 2017). This necessitates the careful consideration of key ecosystem components in both persistent and newly expanded woodlands that influence ecological function, resilience, and resistance to invasive plant species.

Many dedicated scientists and managers have worked to understand these woodlands in an attempt to provide ecosystem services (including clean air and water, forage, wildlife habitat, and recreation) and to restore ecosystem function. Over the past 75 years we have learned a considerable amount about these woodlands, with many successes and failures. In this synthesis, we have collected and summarized the literature on the ecology, history, and management of these semiarid woodlands in an effort to help managers quickly reference the current state of our knowledge.

Synthesis Purpose and Structure

The primary purpose of this synthesis is to: (1) review the current knowledge of pinyon and juniper ecosystems, in both persistent and newly expanded woodlands; (2) address the issues and concerns regarding pinyon and juniper expansion and treatment effects on ecosystem function and resilience and ecosystem services; and (3) make the scientific information for understanding and managing pinyon and juniper woodlands available to managers, researchers, and an interested public. This document provides a resource of information that draws from a large volume of research papers and reports on these semiarid woodlands. In the synthesis, we have reviewed and cited approximately 1,000 papers (of approximately 2,000) related to pinyon and juniper woodlands in the American West.

The synthesis is divided into five sections. The first section, *General Physical Setting,* includes descriptions of both the Great Basin and northern Colorado Plateau. The second section, *Ecology, Life History, and Biology,* covers woodland and species life histories, biology, and ecology. This section also includes a detailed discussion of climate and

the potential consequences of climate change specific to the Great Basin and Colorado Plateau, and to the possible future of woodland ecosystems. The third section, ***20,000 Years of Woodland History***, discusses 20,000 years of woodland dynamics, beginning at Glacial Maximum. This section discusses the magnitude of changes of woodland distribution and structure and the primary factors attributed to prehistoric woodland dynamics. The end of this section focuses on recent changes (including the past 200-300 years) related to the interactions between climate and anthropogenic disturbance, the extent of the change, and the geographic differences among woodland disturbance regimes and resilience. Section 4, ***Ecohydrology of Pinyon and Juniper Woodlands,*** discusses hydrologic processes in woodlands that influence soil conservation and loss; water capture, storage, and release; and the effects woodland structure and composition have on these processes. Section 5, ***Restoration and Management,*** covers the history of woodland management, the different methods used, the advantages and disadvantages of different vegetation treatments, and posttreatment vegetation responses. In the ***Conclusion*** we discuss successes and failures and the key components that determine project outcomes important for consideration when restoring ecosystem function, integrity, and resilience.

Geographic and Ecological Boundaries of This Synthesis

Given the large amount of variation that characterizes pinyon and juniper woodlands in the American West, we focus this synthesis on four juniper and two pinyon species that occur in the Great Basin and northern portion of the Colorado Plateau (northern Arizona, eastern Utah, and western Colorado, level III ecoregion 20) (fig. 1-2). Tree species are western juniper (*Juniperus occidentalis* Hook.), Utah juniper (*J. osteosperma* (Torr.) Little), Rocky Mountain juniper (*J. scopulorum* Sarg.), Sierra juniper (*J. grandis* R.P. Adams), singleleaf pinyon (*Pinus monophylla* Torr. & Frém.), and twoneedle pinyon (*P. edulis* Engelm.) where it is primarily associated with Utah juniper. There is considerably less information on Rocky Mountain and Sierra junipers compared to the other four tree species. This synthesis does not extensively cover tree species on the outer perimeter of this region, which include oneseed juniper (*J. monosperma* (Engelm.) Sarg.), alligator juniper (*J. deppeana* Steud.), or California juniper (*J. californica* Carriére). However, some of the research conducted on these species was included to strengthen the discussion on process and function.

Most of this region typically receives less than 35 percent of its total precipitation during the summer. Cool season grasses dominate the majority of the area with warm season grasses becoming codominate in the southern part as summer precipitation increases (Romme et al. 2009; Shane Green, Soil Scientist, USDA NRCS, Salt Lake City, Utah, personal communication, 2017). Although the focus is on the Great Basin and Colorado Plateau, there are portions of several adjacent areas that were included to cover the distribution range of western and Utah juniper (fig. 1-3). The geographic boundaries we include extend north to eastern Oregon and southern Idaho, west along the east slopes of the Cascades and Sierra Nevada Mountains in California, east to the Rocky Mountains in western Colorado and central Wyoming, and south into northern Arizona and the northwestern corner of New Mexico. Ecoregions included are the Eastern Cascade Slopes (9), Blue Mountain (11), Central Basin and Range (13), high elevation mountains of the Mojave (14), southwest corner of the Middle Rocky Mountains (17), Wyoming Basin (18), Wasatch Uinta Mountains (19), northern Colorado Plateau (20), and Northern Basin and Range (80) (table 1-1; fig. 1-2). Major Land Resource Areas included are shown and listed in figure 1-3 and table 1-1.

SECTION 1: GENERAL PHYSICAL SETTING: GEOLOGY, TOPOGRAPHY, AND SOILS

The Great Basin is characterized by basins, mountains, and plateaus that range in elevation from 1,300 feet to more than 10,000 feet—the majority lying between 2,500 and 7,500 feet above sea level (USDA NRCS 2011). Mountain ranges within the Great Basin are commonly 50–75 miles long and 6–15 miles wide (Lustig 1969). Basins, which are located at the lowest elevations, are hot and dry (mesic/aridic, desert shrub) while the higher elevations are cool to cold and moist (frigid to cryic/xeric, mountain shrub or subalpine). Temperature increases approximately 3 °F for every thousand-foot increase in elevation (Oosting 1956). Pinyon and juniper often intermingle with sagebrush on the midslopes between desert and mountain shrub or forest. Alluvium and playa lakebed deposits typically fill the basins (USDA NRCS 2011). Foothills and mountains are commonly composed of volcanic rock (basalt, andesite, and rhyolite) with occasional granitics. Also important are carbonate soils in the central Nevada Basin and Range and sedimentary and carbonate soils in the Great Salt Lake Region formed from Paleozoic sedimentary rocks (table 1-1).

The most common soil orders are Mollisols, Aridisols, and Entisols. The three most common soil temperature regimes are mesic (warm), frigid (cool), and cryic (cold), commonly mapped by elevation and indicator species. Elevations at which these regimes are mapped change with latitude and thus vary with the geographic location of each MLRA. The dominant soil moisture regimes in the Great Basin are dry (aridic, less than 12 inch precipitation zone—"PZ") and moist (xeric, equal to or more than 12 inches PZ), with aridic or xeric bordering on ustic (available summer soil moisture) in the southern end of the Great Salt Lake MLRA (#28A), and Wasatch Mountains MLRA (#47), which form the boundary between the Great Basin and the Colorado Plateau (table 1-1, fig. 1-3). Soils mapped with aridic-xeric (dry-moist) usually fall within the 10– to 12 –inch PZ.

In Utah, the Wasatch Mountains form the boundaries between the Great Basin and the Colorado Plateau. The Colorado Plateau Ecoregion (Level III Ecoregion 20) is characterized by highly dissected intermountain plateaus extending across the eastern half of Utah, the western edge of Colorado, the northern fringe of Arizona, and the northwestern corner of New Mexico. It includes the area drained by the Colorado River and its tributaries—the Green, San Juan, and Little Colorado rivers. Average elevation is 5,000 feet with peaks over 12,000 feet. Common parent materials are shale, sandstone, limestone, dolomite, and volcanic rock outcrop. The most widespread soil orders are Alfisols, Aridisols, Entisols, and Mollisols (USDA NRCS 2011; table 1-1). Soil moisture regimes are commonly ustic, ustic-bordering-on-aridic, or xeric.

Climate

Most of the Great Basin is characterized by a semiarid temperate climate with cold-wet winters, wet springs, and warm and dry summers. Amounts of summer precipitation are low (typically less than 25 percent of total annual precipitation), increasing along a gradient moving south and east from eastern Oregon (less than 5 percent) to the west slopes of the Wasatch Mountains in southcentral Utah (less than 25 percent). Annual precipitation across the region ranges from 6–12 inches at the lower- to mid-elevations and 12–16 inches at the mid- to upper-elevations, with extremes of less than 6 inches in some of the arid basins and more than 20 inches in the higher mountain elevations.

Average annual precipitation across most of the Colorado Plateau is from 6–18 inches, dropping to less than 5 inches in a few desert basins and reaching the highest levels of precipitation of 30 inches in the isolated mountains of southern Utah (USDA

NRCS 2011). Summer precipitation, July through September, accounts for 25–55 percent of total annual precipitation, increasing from the Wasatch Mountains to southeast New Mexico (Romme et al. 2009). April, May, and June are the driest months (USDA NRCS 2011). The Wasatch Mountains, forming the northwest boundary of the Colorado Plateau, receive considerable winter Pacific moisture but weaken the moisture flow to the east resulting in considerably lower winter precipitation and a higher ratio of summer to winter moisture in the basins to the east (Gray et al. 2004).

The modern climate of western North America is influenced predominantly by juxtaposition of eastern Pacific subtropical high, and the southwestern monsoonal circulation for the Gulf of California and Gulf of Mexico, each with spatially and seasonally varying impacts on the region (Bryson and Hare 1974; Mitchell 1976). Monsoons of western North America occur July through September centered over the Sierra Madre Occidental, Sinaloa Durango, Sonora, and Chihuahua (northern and western Mexico). These rains are a result of a shift in wind patterns in the summer as Mexico and the southwest United States warm under intense solar heating. These winds flow from ocean areas into arid and semiarid regions of the Southwest. Arizona, Utah, and southern Nevada and California are primarily influenced by pulses of moisture from the Gulf of California and the Pacific. The seasonal moisture gradient within the area of interest of this synthesis ranges from less than 15 percent to approximately 35 percent of total annual precipitation received during the summer months. This approximates the distributions of Utah and western junipers.

Locally, temperature and the abundance and seasonal distribution of precipitation are strongly influenced by elevation and latitude across both the Great Basin and Colorado Plateau. For every thousand feet increase in elevation, temperature declines approximately 3 °F (Oosting 1956). In central Utah, precipitation increases approximately 5 inches for each thousand-foot increase in elevation (Lull and Ellison 1950). For more detail on climate and climate change, see Section 2 on climate.

Floristics Divisions and Vegetation Zones

The strong latitudinal gradient in Pacific versus Monsoonal moisture resulting in the amounts of precipitation received during the summer months is a major influence in the geographic distribution of plant species (Blaisdell 1958; Davis 2004; Krebs 1972; Oosting 1956). At the regional and local levels, topography and soils are important modifiers of macro climate and moisture availability that determine vegetation composition and structure.

Floristic Divisions

Cronquist et al. (1972) described four floristic divisions of the Intermountain West: (1) Great Basin, (2) Wasatch Mountains, (3) Colorado Plateau, and (4) Uinta Mountains. Floristic divisions were further divided into 16 floristic provinces. Division boundaries are partially influenced by local regions of endemic species and community dominance plus climatic differences, physical differences, and/or soil differences. At this scale, climatic differences are largely influenced by elevation and latitude.

Vegetation Zones

Vegetation zones are an area in which a particular type of vegetation is expected to dominate over time, commonly referred to as the climax or potential vegetation. Vegetation zones within the Intermountain West are closely associated with elevation, which changes with latitude and aspect. Much of the paleobotany literature refers to the vegetation zones listed below, which include characteristic plant species (Appendix A; Thompson 1990).

Common vegetation zones and their diagnostic species (see Appendix A for a complete list of common and scientific plant names) in the Great Basin and Colorado Plateau associated with pinyon and juniper woodlands in the Intermountain Region include:

1a. Desert shrub—less than 6 inches PZ on nonsodic or nonsalty soils. Spiny hopsage, winterfat, bud sagebrush, shadscale, Nuttall's saltbush, Indian ricegrass, James' galleta, and bottlebrush squirreltail.

1b. Desert shrub (salt-desert shrub)—less than 8 inches PZ on salty or sodic soils (typically in the bottoms of ancient pluvial lakes). Greasewood, salt rabbitbrush, shadscale, basin wildrye, saltgrass, alkali cordgrass, and alkali sacaton.

2. Sagebrush—semi-desert, 8–12 inches PZ. Wyoming big sagebrush, basin big sagebrush, low sagebrush (shallow or clayey sub-soils), black sagebrush (shallow to an indurated duripan), rubber (gray) and green rabbitbrush, bitterbrush, bluebunch wheatgrass (only on north aspects in the southern portion of the region), Sandberg bluegrass, muttongrass, Thurber's needlegrass, needle and thread grass, Indian ricegrass, and blue grama and galleta on aridic soils bordering on ustic.

3. Upland sagebrush and pinyon and juniper—12–16 inches PZ. Mountain big sagebrush, low sagebrush, bitterbrush, western juniper, Utah juniper, singleleaf and twoneedle pinyon, Idaho fescue, bluebunch wheatgrass, Thurber's needlegrass, Columbia needlegrass, Lettermen's needlegrass, western needlegrass, pine needlegrass, Junegrass, Sandberg bluegrass, and low elevation aspen (north-facing snow drift pockets).

4. Mountain brush—more than 16 inches PZ. Mountain big sagebrush, bitterbrush, snowberry, serviceberry, curlleaf mountain mahogany, Idaho fescue, red fescue, spike fescue, Columbia needlegrass, Lettermen's needlegrass, western needlegrass, Lemmon's needlegrass, mountain brome, June grass and mutton grass; on the Wasatch and Colorado Plateau, Gambel oak and bigtooth maple frequently occur in place of the sagebrush-mountain brush zone.

5. Mixed conifer forest—more than 20 inches PZ. Douglas-fir, whitebark pine, limber pine, ponderosa pine, lodgepole pine, white fir, grand fir, aspen, Sierra juniper, and bristlecone pine.

6. Subalpine woodlands—usually composed of bristlecone, limber or whitebark pine, and occasionally Engelmann spruce located just below the limit of tree growth (timberline) and above the foothill or montane or mixed conifer zone.

SECTION 2: ECOLOGY, LIFE HISTORY, AND BIOLOGY

Summary

Pinyon and juniper woodlands occupy over 70,000 square miles of the Great Basin and Colorado Plateau, extending across a significant climatic gradient from eastern Oregon to the Four Corners of Utah, Colorado, Arizona, and New Mexico (West 1984). These semiarid woodlands occupy precipitation zones between 8 and 20 inches, elevations of less than 1,000 to over 8,000 feet, and a wide variety of soils and parent materials. Pinyon and juniper woodlands often project the illusion of being homogeneous, but they vary in age, structure, and composition, and they often intermingle with other plant communities. The broad heterogeneity of these woodlands and the sites they occupy result in large spatial and temporal variations in ecohydrologic process, disturbance regimes, resilience to disturbance, response to vegetation

management, and resistance to invasive species (Stringham et al. 2015; West 1999; West et al. 1978b). The most common semiarid conifer species in this region of the western United States are Utah juniper (*Juniperus osteosperma* (Torr.) Little), western juniper (*J. occidentalis* Hook.), singleleaf pinyon (*Pinus monophylla* var. *monophylla* Torr. & Frém), and twoneedle pinyon (*Pinus edulis* Engelm.). Common but less abundant within this region are Rocky Mountain (*Juniperus scopulorum* Sargent, Gard. & Forest) and Sierra junipers (*Juniperus grandis* R.P. Adams).

Temperature and amount and seasonal patterns of moisture are the primary variables that determine the distribution of these semiarid conifers and their varieties. Rapid warming periods interrupted by cold periods during late winter and early spring in the Northwest limit the northwestern distributions of Utah and singleleaf pinyon. Of the four species, Utah juniper is the most adapted to drought (West 1984). Western juniper typically occurs in cooler, and somewhat wetter environments than Utah juniper. And, the two pinyons are separated by the amounts of summer precipitation. Pinyon and juniper woodlands occupy a wide variety of soils, but there are some general differences in soil characteristics that occur between postsettlement and persistent woodlands.

Soils occupied by persistent woodlands are most commonly associated with shallow to restrictive layers including claypans, fractured basalt, and calcareous horizons and extremely cobbly, or very coarse-textured with gravelly surfaces, often resulting in shallow and transient soil moisture storage (Campbell 2015; Leonard et al. 1987). Woodlands found on deeper and more productive soils (often with higher levels of organic matter) are mostly relatively young and were previously occupied by shrubland or grassland communities.

Annual cone and seed development for the four conifer species is highly variable both temporally and spatially, and it requires at least two growing seasons to produce mature seed (Chambers et al. 1999a). Pinyon pines have short-lived seeds and junipers relatively long, affecting the longevity and abundance of seed in the seedbank. In singleleaf pinyon, a typical mean seed crop during a year of heavy cone production ranges from 2,000 to 8,000 filled seeds per tree, but values for individual trees vary tremendously. Pinyon and juniper seeds are well adapted for dispersal by both birds and small mammals, with distances commonly ranging from several feet to several miles. Germination and establishment are most likely to occur when favorable growing season conditions follow a mast (cone producing) year. Placement of seed beneath nurse plants (shrubs) or covered by soil such as a seed cache also increases the chances of successful establishment. Competition from forbs and grasses can reduce seedling emergence, but once established, competition appears to have little effect on survival.

Pinyon and juniper species are hosts to a large number of insects, disease, and fungi (Bunderson et al. 1986; Shaw et al. 2005). All three can cause reductions in seed crops and increase susceptibility to other pests and mortality. Insects are the largest cause of mortality, with ips (bark beetle) considered the most important insect mortality agent in the Colorado Plateau. Drought is closely linked to pest outbreaks and can significantly increase mortality, especially in the Colorado Plateau.

Expansion of woodlands into sagebrush ecosystems is characterized by an inverse relationship of an increasing tree overstory and decreasing shrub and herbaceous understory (Margolis 2014; Miller et al. 2008; Roundy et al. 2014a). The time it takes woodlands to go from the very early to late successional stages (Phase I to Phase III) is largely determined by the key components of the ecological site and disturbance history. The shift from sagebrush-dominated ecosystems to woodland influences water, energy, and nutrient cycles; C and N pools; and disturbance regimes, wildlife habitat, resilience, and resistance to invasive species.

The earth's climate has been continually changing, long before the formation of the first pinyon and juniper woodland, and has had significant impact on woodland distribution, migration, expansion, contraction, infill, composition, structure, resilience, and resistance to invasive species. But recent concerns over natural climatic variation linked to anthropogenic effects on climate have raised considerable apprehension related to future impacts on ecosystems in the West including pinyon and juniper woodlands (Eddy and Bradley 1991). Increasing temperatures and changes in the amounts and seasonal distribution of precipitation will have significant impacts on persistence and migration of woodlands, insect and disease outbreaks, wildfire, and the expansion of invasive annual grasses. These impacts will likely vary regionally and at multiple scales across the Great Basin and Colorado Plateau.

Woodland Distribution and Ecological Site Characteristics

Pinyon and juniper woodlands occupy 33 to 100 million acres in the American West (West 1984). The inconsistency in area reported is largely a result of the method used to estimate area, the geographic area or species included, and the criteria used to define persistent pinyon and juniper woodlands, wooded shrublands, newly expanding woodlands, and pinyon and juniper savannas (see Glossary for definitions). Kuchler's (1970) potential vegetation map estimates 43 million acres of pinyon and juniper woodlands across Nevada, Utah, Colorado, Arizona, New Mexico, and a portion of southeastern California. West (1984) reported 60 million acres, which includes Kuchler's estimates in addition to adding the Trans-Pecos region of west Texas and western juniper in the northwest. Within the boundaries of this synthesis—the Great Basin and northern Colorado Plateau—there is an estimated 48 million acres of woodland (table 2-1; figs. 2-1, 2-2).

Table 2-1—Estimated area occupied by Utah and western junipers and singleleaf and twoneedle pinyon pines in the Intermountain West (from Miller and Tausch 2001).

State	Species	Area (acres)	Reference
Arizona	JUOS-Pinyon	11,641,999	Springfield 1976
California	JUOC	1,283,999	Bolsinger 1989
	JUOC savanna	796,999	Bolsinger 1989
	JUOS-JUCA	1,089,000	Bolsinger 1989
Colorado	JUOS-PIED	5,937,880	estimated from Powell et al. 1994
Idaho	JUOC	617,500	Chojnacky 1995
	JUOS	137,001	Tueller et al. 1979
New Mexico	JUOS-PIED	3,081,999	Springfield 1976
Nevada	JUOS-PIMO	7,155,970	O'Brien and Woudenberg 1999
	JUOS	1,683,566	O'Brien and Woudenberg 1999
	JUOC	100,000	estimated Miller et al. 2005
Oregon	JUOC	3,339,000	Azuma et al. 2005
	JUOC savanna	3,227,000	Azuma et al. 2005
Utah	JUOS-PIED (PIMO)	7,766,307	O'Brien and Woudenberg 1999
	JUOS	148,400	O'Brien and Woudenberg 1999
Wyoming	JUOS-PIED	202,999	Powell et al. 1994
Total		**48,209,619**	

JUOS = *Juniperus osteosperma;* JUOC = *J. occidentalis* var. o*ccidentalis,* JUCA = *J. californica,* PIMO = *Pinus monophylla,* PIED= *P. edulis*
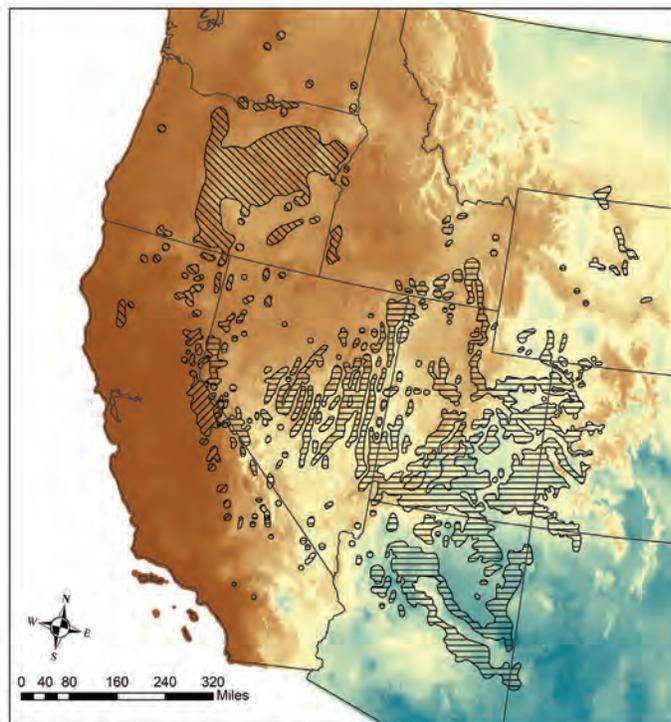
Figure 2-1—Distribution of Utah (*Juniperus osteosperma*), western (*J. occidentalis*), and Sierra (*J. grandis*) junipers across a monsoonal precipitation gradient in the Great Basin and Colorado Plateau. Map derived by David Board, Ecologist/Data Analyst, USDA Forest Service, Rocky Mountain Research Station, Reno, NV.
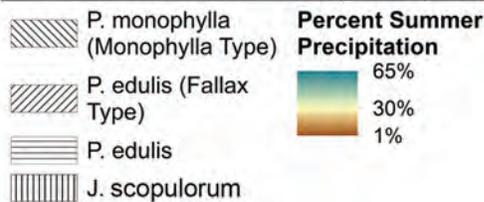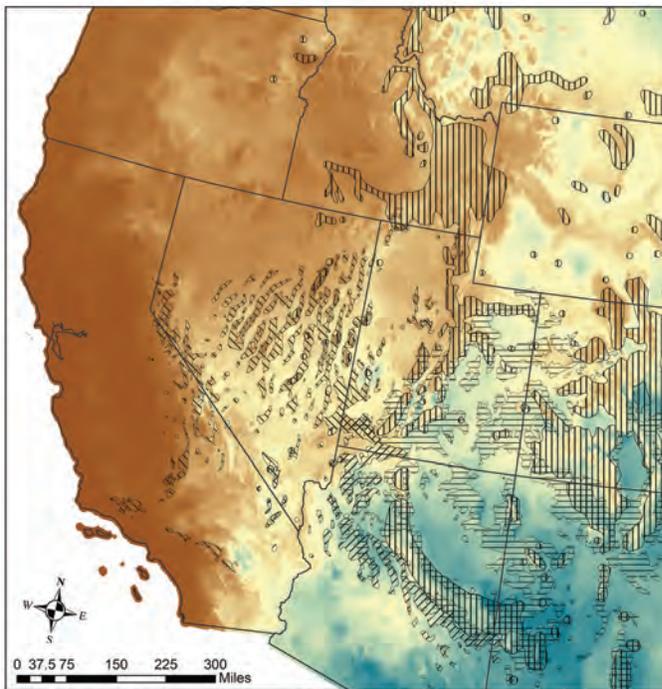
Figure 2-2—Distribution of singleleaf (*Pinus monophylla*) pinyon, twoneedle (*P. edulis*) pinyon, variety P. edulis var. fallax, and Rocky Mountain juniper (*J. scopulorum*) across a monsoonal precipitation gradient in the Great Basin and Colorado Plateau. Map derived by David Board, Ecologist/Data Analyst, USDA Forest Service, Rocky Mountain Research Station, Reno, NV.

Figure 2-3—Pinyon and juniper woodlands may appear as homogenous stands within a contiguous elevation belt. But in reality, they vary in stand structure and composition and intermingle with other plant communities across continually varying topography, soils, and microclimates. Schell Creek Mountains, Nevada. (Photo by Rick Miller, Oregon State University.)

Although pinyon and juniper woodlands are sometimes reported as occurring within a specific elevation belt in the Great Basin, they often intermingle with other plant communities (Tueller et al. 1979). The transition between pinyon and juniper woodlands to other vegetation zones or plant communities is usually associated with changes in elevation, aspect, microtopography, and soils, which influences temperatures, abundance of available moisture, and disturbance regimes (fig. 2-3). Latitude also influences woodland elevation boundaries and dominance on north versus south aspects (Tueller et al. 1979). Pinyon and juniper woodlands typically occupy elevations well above the desert basins in the more than 8 inches PZ and just above the 8 -12 inches PZ occupied by the sagebrush semi-desert. Utah juniper often dominates the lower woodland elevations (typically less than 7,000 feet). Woodlands composed of both pinyon and juniper frequently intermingle with upland sagebrush (10-16 inches PZ) (Stringham 2015a,b; West 1999). Mountain brush and mixed conifer forests frequently occupy elevations just above or intermingle with the upper pinyon and juniper woodland boundary. In the northwest, western juniper is the sole semiarid conifer, which most commonly occurs in the 10–15-inch PZ (Gedney et al. 1999) and is associated with Wyoming big sagebrush at the lower elevations, mountain big sagebrush at the mid- to upper- elevations, and on shallow to very shallow soils low sagebrush (Miller et al. 2005).

Climate

The majority of woodlands receive 10–16 inches of total annual precipitation across the Intermountain Region (Stringham et al. 2015; West et al. 1978a; West 1988), with extremes ranging from 8 to 20 inches (Stringham et al. 2015). There is a large range in the distribution of seasonal precipitation along a gradient from eastern Oregon to southeast Utah and southwest Colorado, where summer precipitation ranges from less

than 5 percent to more than 30 percent of the total annual precipitation (figs. 2-1, 2-2). This northwest to southeast shift in the seasonal precipitation is a result of the importance of Pacific and Monsoonal storm systems.

The quantity and seasonal availability of soil moisture, especially in the summer, are major factors influencing the geographic distribution of plant species across the Intermountain Region (Blaisdell 1958; Daubenmire 1974; Oosting 1956). Juniper is generally more tolerant to dry and cold conditions (especially fluctuations of cold and warm in early spring) than pinyon, resulting in juniper often dominating the more arid lower elevations of the woodland zone and often the more dominant tree in the northern latitudes (Tueller and Clark 1975). The relative abundance of pinyon to juniper often increases with elevation, and pinyon typically becomes dominant on the midslopes, where temperature inversions create thermal belts that have less extreme temperatures than the basins below (Billings 1954; Tueller et al. 1979; West et al. 1978).

Western juniper and singleleaf pinyon grow in areas where most precipitation falls between October and May (Cole et al. 2008a; Gibson 2011; Miller et al. 2005), while twoneedle pinyon grows in areas where more than 30 percent of precipitation falls between July and September (fig. 4-4 in Section 4; Cole et al. 2008a; West et al. 2007a). Utah juniper grows in both winter-dominant and summer-dominant precipitation regimes, although it is more widely distributed where most precipitation falls in winter (Gibson 2011). Seasonal ranges in temperature between these four species are quite similar, but temperature events not captured in monthly climate data can exert control on species distributions.

Singleleaf pinyon and Utah juniper typically grow in thermal belts above the level of freezing fogs in valleys and are susceptible to damage from freezing in late winter and early spring during rapid shifts between warm periods and Pacific, or polar, front cold periods (Billings 1954; Nowak et al. 1994). The northwestern limits of singleleaf pinyon and Utah juniper appear to correspond to the energy limits of cold Pacific fronts resulting in rapid cycling of warming periods interrupted by cold periods during late winter and early spring brought about by these Pacific frontal storms (Nowak et al. 1994a; West et al. 1978a). These warm periods encourage pinyon to break dormancy early, making them susceptible to frost damage. Western juniper is limited at upper elevations by extreme winter temperatures and may be limited by spring freezes at lower elevations (Miller and Rose 1995; Miller et al. 2005). Twoneedle pinyon is sensitive to soil temperature in the upper soil layer during summer, especially during drought (West et al. 2007a; Williams and Ehleringer 2000).

Of the four species, Utah juniper is the most adapted to drought (Munson et al. 2011). Western juniper typically occurs in cooler, and somewhat wetter environments than Utah juniper (fig. 2-4; Miller et al. 2005; Nowak et al. 1994a) and is more sensitive to drought in sites with high evapotranspiration demand, low soil water storage capacity, or both, such as lower elevations, steep slopes, and rocky or sandy soils (Knutson and Pyke 2008). Utah juniper appears to be much less sensitive to changing climate than both pinyon species and western juniper, based on the paleoecological record (Greenwood and Weisberg 2008; Miller et al. 2005; Nowak et al. 1994a). Twoneedle pinyon appears to be the most sensitive to drought of these four species, particularly to drought in summer when temperatures are typically high (Lloret and Kitzberger 2018; Peterman et al. 2013; West et al. 2007a; West et al. 2007b). Singleleaf vulnerability to drought appears to be low in the Great Basin (Biondi and Bradley 2013) compared to high vulnerability of twoneedle pinyon in the Colorado Plateau (Breshears et al. 2005; Mueller et al. 2005a) (discussed in further detail in Sections 2 and 3).
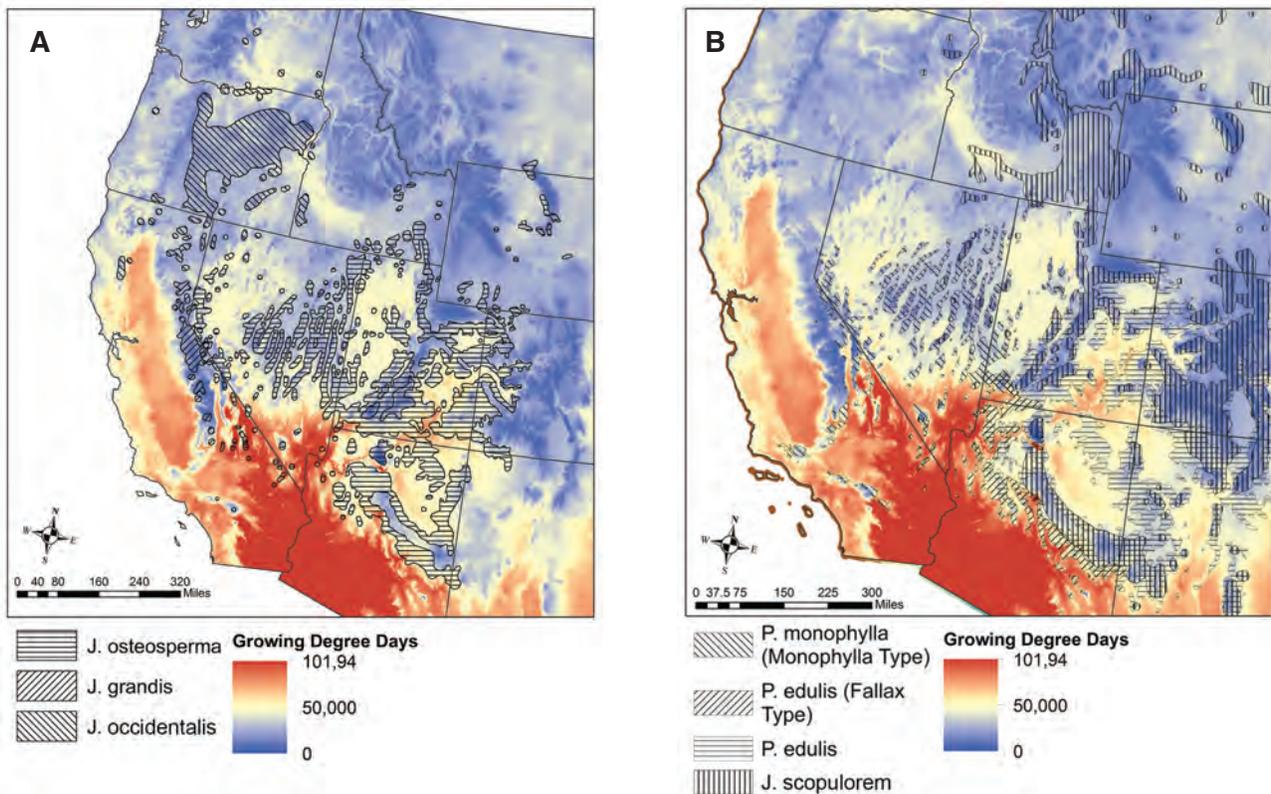
Figure 2-4—Distribution of (A) Utah (*Juniperus osteosperma*), western juniper (*J. occidentalis*), and Sierra juniper (*J. grandis*); and (B) twoneedle pinyon (*Pinus edulis*), type Fallax, singleleaf pinyon (*P. monophylla*), and Rocky Mountain juniper (*Juniperus scopulorum*), across temperature gradients based on Growing Degree Days (GDDs). GDDs are based on 30-year averages with a lower limit of 32 ˚F and an upper limit of 110 ˚F. (Map derived by David Board, Ecologist/Data Analyst, USDA Forest Service, Rocky Mountain Research Station, Reno, NV.)

Soils

> *"The best stands* [of pinyon pine] *are found on coarse gravel, gravelly loam, or on coarse sand, of 1.5 m* [5 feet] *in depth, on which humus and ground cover are almost entirely lacking"* (Phillips 1909).

Pinyon and juniper are capable of growing on a wide range of soils and parent materials across the Great Basin and Colorado Plateau (Harper and Davis 1999; Leonard et al. 1987; Pearson 1931; Springfield 1976; Stringham et al. 2015; West et al. 1998; table 1-1). Trees occupy soils from parent materials originating from metamorphic, sedimentary, and igneous sources including tuff, welded tuff, pumice, volcanic ash, rhyolite, andesite, granite, basalt, limestone, sandstone, eolian soils, and colluvial or alluvial mixtures of the above. Woodlands most commonly occur on Mollisols, Aridisols, Entisols, and to a lesser degree Inceptisols. The highest tree densities are most frequently found on Mollisols, which are typically the most productive among the above soil orders (Nettleton and Mays 2007), and have the highest potential for encroachment (Campbell 2015). Pinyon and juniper woodlands are also found on soils with nearly all particle sizes from fine to coarse—with the exception of silty and sandy skeletal (Leonard et al. 1987). Trees occupy soil depths ranging from very shallow to deep (Barney and Frischknecht 1974; Campbell 2015; Leonard et al. 1987). On very shallow soils, tree roots often penetrate the deep fractures and cracks of the restrictive layers.

The wide variety of soils occupied by pinyon and juniper result in different levels of potential productivity, fuel loads, size and density of trees, and understory composition (Leonard et al. 1987; Miller et al. 2005; Ramsey 2003; Stringham et al. 2015). This

results in changes of woodland structure and disturbance regimes, which can occur at relatively small scales (Miller and Heyerdahl 2008). In northern Arizona, tree densities on limestone- and basalt-derived soils were similar but double that of sandstone soils (Landis and Bailey 2005). On heavy clay or shallow claypan soils, western juniper densities and growth rates are significantly less than on moderately deep clay loams and loams (Miller and Rose 1995; Young and Evans 1981).

Tree mortality can also vary across different soils. Greater dieback in singleleaf pinyon and Utah juniper occurred in Arizona on soils derived from volcanic cinder compared to basalt or sedimentary (Koepke et al. 2010). Geographic information data across the Colorado Plateau demonstrated that 84 percent of the pinyon and juniper mortality during the 2003 and 2004 drought occurred on soils with relatively low available water holding capacity or less than 3.9 inches (Peterman et al. 2013). These are soils that typically have high sand and low organic matter content. Low soil water availability decreases the sapwood/unit leaf area, which increases the sensitivity of trees to bark beetle mortality.

There are some general differences in soil characteristics that occur between postsettlement and persistent woodlands, although there is considerable overlap. Persistent woodlands are often associated with rock outcrops, knolls, ridges, and/or soils that are shallow, coarse, rocky, and often high in clay or sand (fig. 2-5; Barney and Frischknecht 1974; Bauer and Weisberg 2009; Cottam and Stewart 1940; Emerson 1932; Holmes et al. 1986; Miller and Rose 1999; Nicol 1937; Stringham et al. 2015; Woodbury 1947). Characteristics reported in soil surveys most frequently associated with persistent woodlands are shallow to restrictive layers including claypans, fractured basalt, and calcareous horizons and extremely cobbly or very coarse-textured with gravelly surfaces (Leonard et al. 1987; ; Stringham et al. 2015; USDA NRCS 1997; Randy Lewis, Soil Scientist USDA NRCS, Tremonton, Utah, personal communication, 2015).

Figure 2-5—Old-growth woodlands frequently occur on soils of low productivity associated with rock outcrops, knolls, ridges, and/or soils that are shallow to very shallow, coarse, rocky, and often high in clay or sand. But they can also occur on mollic soils. Central Nevada. (Photo by Rick Miller, Oregon State University.)

Romme et al. (2009) reported that persistent woodlands tended to occur where soil moisture storage is shallow and transient. Early work by Emerson (1932) reported pinyons and junipers typically grow on rocky ridges and dissected edges of mesas, whereas grasslands were on the more level areas at the base or on the tops of these mesas. If persistent stands of pinyons and junipers are found on moderately deep to deep soils, the soils are typically skeletal and/or coarse-sandy (Harris et al. 2003; Jacobs et al. 2008; Leonard et al. 1987; Miller and Heyerdahl 2008; Waichler et al. 2001). In relatively dense persistent woodlands, soils typically lack mollic horizons, are poorly developed (Blackburn et al. 2015), and are of low productivity (Barney and Frischknecht 1974; Howell 1941; Leonard et al. 1987; Miller and Heyerdahl 2008; Springfield 1976; Weisberg et al. 2008). However, presettlement woodlands have been reported on relatively productive, moderately deep to deep soils (Floyd et al. 2004). Mollic horizons can also be found in persistent pinyon and juniper woodlands with relatively open canopies supporting a dominant understory of perennial herbaceous vegetation on moderately deep to deep soils with adequate available moisture—e.g., more than 12 inches PZ (fig. 2-6). However, the majority of woodlands found on the deeper and more productive soils are often composed of postsettlement trees (Bauer and Weisberg 2009; Campbell 2015; Emerson 1932; Jacobs et al. 2008; Miller et al. 2005; Miller and Heyerdahl 2008; Nicol 1937; Woodbury 1947). Relatively young woodlands expanding into Mollisols can potentially have the highest tree densities (Nettleton and Mays 2007).

Soil Moisture and Temperature Regimes

Soil moisture regimes that commonly support pinyon and juniper woodlands within the Great Basin and the northern portion of the Colorado Plateau are xeric or intergrades of aridic/xeric (e.g., aridic bordering on xeric, see soil moisture regime in Glossary and

Figure 2-6—This western juniper woodland—with 20 to 25 percent tree canopy cover—grows on moderately deep sandy (pumice) skeletal loam soils with an understory of bluebunch wheatgrass, Idaho fescue, and mountain big sagebrush. The majority of trees are 200–400 years old and 20th century infill has been minimal to moderate. Horse Ridge, Oregon. (Photo by Rick Miller, Oregon State University.)

sidebar 1) and aridic/ustic (Leonard et al. 1987). Soil temperature regimes are typically mesic and frigid. But Rocky Mountain juniper most commonly occurs on frigid/ustic soils. Pinyon and juniper species on thermic, mesic/aridic, and cryic/udic soils are less common (Campbell 2015; USDA NRCS data). Over 50 percent of twoneedle pinyon occupies mesic (warm) and aridic-ustic intergrades, whereas singleleaf pinyon typically grows on mesic and frigid soils that are xeric or aridic-xeric intergrades (table 2-2). Twoneedle and singleleaf pinyons, and Utah and western junipers, typically do not grow on cryic soils, but Sierra juniper commonly grows on cryic soils. Oneseed and alligator junipers, which grow on the southern and southeastern boundaries of Utah juniper, occur on ustic soils where summer precipitation accounts for more than 35 percent of the annual total (figs. 2-1, 2-2).

Potential woodland expansion into sagebrush communities varies across soils, moisture, and temperature regimes (Campbell 2015). Encroachment potential is moderate to high on mesic and frigid soils with xeric soil moisture regimes or xeric and aridic intergrades. The potential for encroachment declines on aridic soils, especially where mean annual precipitation is less than 10 inches, and very low in cryic and thermic temperature regimes, and soils that are poorly drained, alkali, saline, and sodic.

Topography: Elevation and Landform

The majority of Utah juniper and pinyon woodlands are located between 4,500–8,000 feet (Menlove et al. 2016; Springfield 1976; Woodbury 1947; Wright et al. 1979) and western juniper between 1,500– 6,000 feet (Gedney et al. 1999; Miller and Rose 1995). But elevations occupied by pinyon and juniper woodlands vary with latitude and, at local scales, with aspect. In the White Mountains just east of Bishop, California, woodlands grow between 6,500–9,500 feet (Jennings 1995; St. Andre et al. 1965). At 9,500 feet, they are replaced by limber and bristlecone pines. In the northern range of western juniper along the Columbia River, trees are found at 600 feet (Miller et al. 2005). Another exception is Sierra juniper, which typically grows in open scattered stands with other conifers near the treeline on the east slopes of the Sierra Nevada Mountains (Miller et al. 2005) and at high elevations in the mountains of central and western Nevada above 9,000 feet (Charlet 1996).

Limited moisture is a primary factor restricting woodland movement into lower elevations (Daubenmire 1943; Pearson 1920). But spring temperature inversion layers that form cold-air layers in the valley basins also appear to be related to the lower boundary of pinyon and juniper woodland, with shrubland forming along the base of the slopes and the valley floors (Billings 1954). At the upper elevations, cold temperatures and/or competition from other conifer and mountain shrub species better adapted to the environmental conditions limit pinyon and juniper. Size of the mountain mass can also influence woodland elevation. Woodlands often occur at higher elevations on smaller mountain ranges as they tend to intercept less moisture than larger mountain masses (Tueller et al. 1979; West et al. 1978a).

Woodlands also grow across a broad range of landforms (Romme et al. 2009; Stringham et al. 2015) including ridges, hill and mountain slopes, terraces, tablelands, plateaus, alluvial fans, broad basins, and valley floors. Across the Colorado Plateau it is common to observe woodlands dominating the convex slopes and shrublands in the adjacent concave topography (fig. 2-7).

Sidebar 1

Soil Term Reference
(see Glossary for expanded definitions)
Aridic – dry
Xeric – moist
Ustic – summer moisture regime
Mesic – warm
Frigid – cool
Cryic – cold

Table 2-2—Juniper and pinyon species in the American West, their general range, and soil moisture regimes they commonly occupy within the Great Basin and northern Colorado Plateau (from Adams 2014; Bailey 1987; Cole et al. 2008; Critchfield and Little 1966; Eckenwalder 2009; Leonard et al. 1987; Vasek 1966; West 1999; Willson et al. 2008).

Species		Soil temperature - moisture regimes	General range and elevation
Juniperus			
J. arizonica[1]	Redberry juniper		Central and southeastern Arizona to southwestern New Mexico and northern edge of Mexico; 3,200–5,250 (7,200) ft
J. californica	California juniper		Central California, western Arizona and south into Baja; (900) 2,450–3,900 (5,250) ft
J. ashei	Ashe's juniper		Central Texas, Oklahoma, Arkansas
J. coahuilensis[2]	Redberry juniper		Southwestern Texas and northern Mexico 3,940–6,560 ft
J. communis	Common juniper	Frigid-cryic/xeric	var. *depressa* – Alaska, Canada, northeastern U.S., outlying populations central U.S.; var. *kelleyi* Alaska, British Columbia, northwestern U.S.
J. deppeana	Alligator juniper	Mesic-ustic	Southwest and Mexico 4,900–9,500 ft
J. flaccida	Drooping juniper		Mexico; 6,000–8,000 ft
J. grandis[3]	Sierra juniper	Frigid, cryic/xeric, xeric-aridic	Sierras and the San Gabriel, San Bernardino and San Jacinto Mountains, high mts of central Nevada; 3,280–9,850 ft
J. horizontalis	Creeping juniper		Northern U.S.; 0–3,300 ft
J. monosperma	Oneseed juniper	Mesic/udic, ustic, and aridic-ustic	Northern Arizona, New Mexico, southeastern Colorado northern and western Texas; 3,300–7,550 ft
J. occidentalis	Western juniper	Mesic, frigid/xeric, xeric-aridic	Northwest; 600–7,000 ft
J. osteosperma	Utah juniper	Mesic, frigid/xeric, aridic-xeric, aridic-ustic,	Great Basin and Colorado Plateau; 4,250–8,225 ft
J. pinchotii	Pinchot's juniper		western and central Texas, western Oklahoma
J. virginiana	Eastern redcedar		Midwest and eastern U.S.
J. scopulorum	Rocky Mountain juniper	Frigid-ustic (aridic/ustic)	Interior Mountain Ranges from British Columbia to northern Arizona and New Mexico; 5,000–8,500 ft
Pinus (cembroides)			
P. cembroides	Mexican pinyon		Southwestern Texas and Mexico
P. discolor[4]	Border pinyon		Southern Arizona and New Mexico
P. edulis	Twoneedle	Mesic, (frigid)/aridic-ustic	Colorado Plateau, central Colorado, New Mexico; var. Fallax; 4,500–7,500 (8,500) ft
P. monophylla	Singleleaf	Mesic, frigid/xeric, aridic-xeric	Great Basin Nevada, southern California into Baja; var. *monophylla, californiarum;* 4,000–8,000 (10,000 White Mts) ft

[1] Syn: *J. coahuilensis,* which occurs in Mexico and southwest Texas and *J. erythrocarpa.*
[2] Syn with *J. erythrocarpa*
[3] Syn: *J. occidentalis* var. *australis*
[4] Disputed species; some consider to be *P. cembroides* var *bicolor*

Taxonomy

There are 15 species of juniper and 4 species of pinyon in the American West and northern Mexico (table 2-2). However, singleleaf and twoneedle pinyons, and Utah and western junipers, are the primary species occurring within the Great Basin and northern Colorado Plateau. Two additional species are typically found growing in cooler temperatures than western and Utah junipers (fig. 2-4a,b). Rocky Mountain juniper is found primarily in the Rocky Mountains and Mogollon Rim, and Sierra juniper is found

Figure 2-7—Across the Colorado Plateau, it is common to observe woodlands dominating the convex slopes and shrublands in the adjacent concave topography—where soils are usually deeper and surface fuels greater. Northern Arizona. (Photo by Rick Miller, Oregon State University.)

on the upper east slopes of the Sierras and high mountain ranges in central Nevada. Pinyon and Utah juniper species grow separately or in mixed stands, whereas western juniper is not associated with pinyon pines. The abundance and seasonal distribution of precipitation and temperature are primary factors that determine the distribution and separation of these tree species (figs. 2-1, 2-2, 2-4a,b; Springfield 1976).

Utah Juniper (Juniperus osteosperma (Torr.) Little)

Utah juniper is the most common tree in the Great Basin (Lanner 1984) and one of the most abundant trees in the Colorado Plateau. Within the Great Basin, Utah juniper, often associated with singleleaf pinyon, occupies 17 million acres, of which nearly two-thirds occur in Nevada (Tueller et al. 1979). This species is most commonly found from 4,500–8,500 feet in elevation and ranges from western New Mexico and central Wyoming to the east slopes of the Sierra Nevada Mountains (fig. 2-1; tables 2-1, 2-2; Cronquist et al. 1972; Vasek 1966). Its southern boundary occurs in the high mountains of the Mojave and Sonoran deserts in southern California and along the Mogollon Rim in Arizona where it is associated with blackbrush (fig. 2-8). It occurs as far north as southern Montana in the Pryor Mountains (fig. 2-9). The majority of its range is characterized by hot, dry summers and wet, cold winters with precipitation ranging from 10 to 18 inches and extremes as low as 8 and as high as 20 inches. The species extends into north central Arizona and western New Mexico where summer precipitation approaches 40 percent of the total annual precipitation (fig. 2-1). But the majority occurs where summer moisture accounts for less than 30 percent of total annual precipitation. It grows on a wide range of soils, but most commonly on gravelly loams and gravelly clay loams within a pH range of 7.4 to 8.0. It most often occupies soils with aridic and xeric soil moisture regimes, along with mesic and frigid temperature regimes. But soil moisture regimes often border on ustic in the southeastern portion of its range.

Figure 2-8—The Utah juniper southern boundary occurs in the high mountains of the Mojave and Sonoran deserts in southern California and Arizona, where it is associated with blackbrush. Southwest corner of Utah. (Photo by Rick Miller, Oregon State University.)

Figure 2-9—The northern range of Utah juniper occurs in Wyoming crossing into the very southern portion of Montana in the Pryor Mountains. Wind River Canyon of central Wyoming. (Photo by Rick Miller, Oregon State University.)

Utah juniper is a small monoecious (sometimes dioecious) tree 10 to 20 feet tall (Cronquist et al. 1972) but it can exceed heights of 40 feet. It often has one to three trunks or multiple main upright branches nearly the same size as the trunk. The scale-like leaves are 0.08–0.12 inches (2-3 mm) long with serrated margins (Willson et al. 2008) and are mostly opposite in twos or (rarely) in threes. Resin glands on the leaf scales are deeply imbedded in the mesophyll and often difficult to see (fig. 2-10). Unlike western juniper, they do not exude resin. Female cones are usually blueish-brown or reddish-brown beneath the glaucous, contain one to two seeds, and are 0.31–0.35 inches in diameter, with extremes 0.24–0.47 inches (8–9 mm, extremes 6–12 mm) in diameter (fig. 2-11).

Figure 2-10—Young female cones developing in the leaf axials of the leaf scales in Utah juniper. Note the lack of resin glands occurring on the leaf scale surface, which occur in western juniper. (Photo by Rick Miller, Oregon State University.)
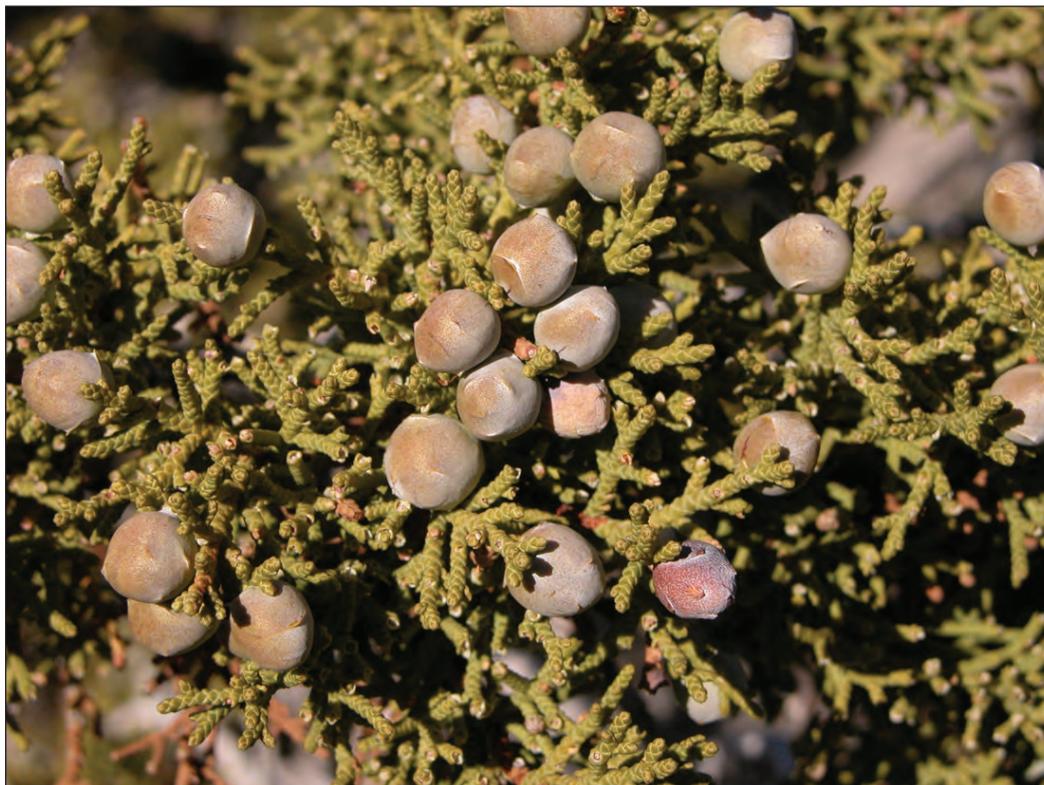
Figure 2-11—Usually blueish-brown and reddish-brown beneath the glaucous, Utah juniper cones contain one to two seeds and take two growing seasons to produce mature fruit. (Photo by Rick Miller, Oregon State University.)

Western Juniper (Juniperus occidentalis Hook.)

Western juniper grows north of singleleaf pinyon and Utah juniper woodlands in central and eastern Oregon, southwestern Idaho, northern and northeastern California, northwestern Nevada, and north into Washington (fig. 2-1; tables 2-1, 2-2). It occupies elevations between 600 and 6,000 feet (Gedney et al. 1999) and (uncommonly) above 7,000 feet (Miller et al. 2005). It grows on a wide variety of soils and parent materials derived from aeolian, sedimentary, and igneous sources (Driscoll 1964; Miller et al. 2005). Soil textures range from coarse, sandy to heavy clays and soil moisture regimes of xeric and xeric-aridic. Most western juniper are found on frigid soil temperature regimes or intergrades between mesic and frigid and rarely on cryic soils. The highest densities of western juniper are found on moderately deep Mollisol soils, of which the majority are young trees (less than 150 years). On shallow to moderately deep soils, it is commonly associated with Wyoming and mountain big sagebrush. When growing on shallow to very shallow soils (underlain by bedrock or claypans including Camborthids, Duragids, and Haplargids), western juniper forms open low-density canopies associated with low sagebrush.

Western juniper is a submonecious tree typically 12 to 35 feet tall, though reaching heights exceeding 60 feet (Adams 2014; Cronquist et al. 1972; Miller et al. 2005). Leaf scales are usually opposite, or in whorls of three, with a serrated leaf margin. Leaves contain conspicuous resin glands, exuding resin that forms a white crust on the leaf scales—a useful characteristic for separating western from Utah juniper (fig. 2-12). In northwestern Nevada and northeastern California, it is common to find western and Utah juniper hybrids, with a single tree having both well-developed exuding and inconspicuous nonexuding resin glands (Terry et al. 2000; Vasek 1966). Male cones are 0.1–0.2 by 0.08–0.12 inches (3–5 by 2–3 mm) in size and form on the ends of the short branchlets. Female cones are approximately 0.28–0.35 inches (7–9 mm) bluish-black at maturity, covered with a resinous pulp, and contain one to two (sometimes three) seeds.

Figure 2-12—Western juniper male cones just prior to pollination in the early spring. Leaf scales are usually opposite or in whorls of three and contain obvious ruptured resin glands on the back of the leaf scales, which are visible but usually not ruptured on Sierra juniper, and not visible on Utah juniper. (Photo by Rick Miller, Oregon State University.)

Sierra juniper, *Juniperus occidentalis* Hook. var. *australis* Vasek, once grouped as a variety of western juniper, has been identified as a separate species, *J. grandis* (Adams 2014). A form of western juniper has also been recently identified as *J. occidentalis* forma *corbetti*, located just east of Bend, Oregon, exhibiting a growth form of a compact shrub (Adams 2014).

Rocky Mountain Juniper (Juniperus scopulorum Sargent, Gard. & Forest*)*

Rocky Mountain juniper has the widest distribution among both pinyon and juniper species in the West (fig. 2-2). It is found at sea level (San Juan Islands) but more commonly between 4,000–9,000 feet in the southern part of its range and 2,000–7,500 feet in the northern portion of its range. Its distribution extends from southcentral New Mexico north into Canada (fig. 2-13; Adams 2014). Most commonly it occurs throughout the Rocky Mountains in both the United States and Canada and along the Mogollon Rim in Arizona. It grows in clay, rocky, and sandy textured soils derived from limestone, basalt, and sandstone—and in typically cooler environments than Utah juniper or western juniper (fig. 2-4a,b). Along its eastern boundary, it hybridizes with eastern red cedar, forming the subspecies *Juniperus virginiana* var. *scopulorum*. It also hybridizes with creeping juniper.

Rocky Mountain juniper is a dioecious, typically single-stemmed tree up to 60 feet tall with a pyramidal to occasionally round top (Adams 2014; Cronquist et al. 1972). Leaf scales are usually opposite, but sometimes in whorls of three, and usually not overlapping or less than 1/5 of the length, and with inconspicuous resin glands. Leaf margins are entire, unlike the serrated leaf scales in Utah and western junipers (Willson et al. 2008). Leaf branches are also considerably more slender than Utah or western junipers. Male cones form on the branch tips and are very small, 0.08–0.12 inches (2–3 mm). Seed cones are 0.24–0.35 inches (6–9 mm) in diameter, light blue with heavy glaucous, but dark blue beneath. There are usually one to three seeds per cone.

Figure 2-13—Rocky Mountain juniper has the most extensive range of junipers that occur in the West, extending north into Canada and south along the Mexico border. Garden of the Gods, east slope of the Rocky Mountains. (Photo by Rick Miller, Oregon State University.)

Sierra Juniper (*Juniperus grandis* R.P. Adams)

Also known as grand juniper and big western juniper, Sierra juniper grows from Lassen County in northeastern California, south along the east slopes of the Sierra Nevada Mountains, to the San Bernardino Mountains (Vasek 1966), typically between 3,300–10,000 feet (fig. 2-14; Adams 2014). It also extends east into the central and northern Nevada mountain ranges where it grows at moderately high elevations well above Utah juniper (Charlet 1996). It occupies cooler sites than western or Utah junipers (fig. 2-4a). In the Sierra Nevada Mountains, it is often associated with subalpine forests and in central Nevada with high sagebrush or mountain brush communities.

Once considered a subspecies of western juniper (*J. occidentalis* subsp. *australis*), Sierra juniper is mostly dioecious and can attain a larger size (up to 100 feet) than western juniper (Vasek 1966). Leaf scales are mostly in threes with visible glands that often do not exude resin (Adams 2014). Development of both male and female cones is similar to western juniper, with one to two and sometimes three seeds per cone.

Singleleaf Pinyon (*Pinus monophylla* var. *monophylla* Torr. & Frém.)

First described by John C. Fremont in 1944, singleleaf pinyon is also known as singleneedle pine, nut pine, one-leaf pine, *piñon*, or pinyon. It grows throughout much of the Great Basin in Nevada, throughout western Utah, and along the east slopes of the Sierras south to southern California and north to southern Idaho (fig. 2-2; tables 2-1, 2-2). Annual precipitation most commonly ranges between 10 and 16 inches, with extremes as low as 8 or as high as 18 inches, of which the majority occurs during the winter and spring and less than 25 percent occurring during the summer months (fig. 2-2). Singleleaf pinyon typically grows between 3,300–9,000 feet in elevation (as high as 9,700 feet in the White Mountains of California) on coarse-textured soils derived from a wide variety of parent materials with surface pH commonly between 6.0–8.0 (Meeuwig et al. 1990). Soil moisture and temperature regimes are most commonly frigid or mesic bordering on

Figure 2-14—Sierra juniper rooted in the granitic walls of the High Sierras in Yosemite National Park at 8,500 ft. In the Sierras this species often grows at timberline. (Photo by Rick Miller, Oregon State University.)

frigid. It is most commonly associated with Utah juniper, but at higher elevations it mixes with the lower boundaries of ponderosa, limber, and bristlecone pines and Douglas-fir.

Trees are typically 15 to 50 feet tall often with one to three main stems. Leaves are 1.2–2 inches (30–50 mm) long and mostly one needle, but a small percentage supporting two needles per fascicle are commonly found in the southeastern portion of its range (Cronquist et al. 1972). See below for pinyon varieties. Male cones are 0.24 inches (6 mm) long. Female cones are broadly ovoid 1.4–2 inches (35–55 mm) but occasionally up to 3.1 inches (80 mm) long and occurring subterminal or lateral on the branchlets (fig. 2-15). Seeds are 0.51–0.67 inches (13–17 mm) long.

Twoneedle Pinyon (Pinus edulis Engelm.)

Twoneedle pinyon, also known as Colorado pinyon, nut pine, twoneedle pinyon, and two-leaf pinyon, occurs along the Wasatch Mountains, the boundary between the Great Basin and where it can mix with singleleaf pinyon and the Colorado Plateau. It extends east and south from the Wasatch and Colorado Plateau well into New Mexico, southern Colorado, and the extreme western edge of Oklahoma and west Texas (fig. 2-2). Its southern-most boundary is southern New Mexico and extends north near the Wyoming, Colorado, and Utah border. The range of twoneedle pinyon generally has milder winter temperatures and greater proportions of summer annual precipitation than the range of singleleaf pinyon. Twoneedle pinyon primarily occupies the rocky plateaus, mesas, foothill terraces, and lower mountain slopes between 4,500– 7,500, up to 8,500 feet, in the Colorado Plateau (Cronquist et al. 1972; Ronco 1990). Total annual precipitation is widely variable, ranging between 10–22 inches with summer precipitation exceeding 3 inches (West et al. 1978a). This species grows in a wide variety of soil depths and textures ranging from very coarse to fine compacted clays (Ronco 1990). Just over 50 percent of this species grows on mesic-aridic/ustic moisture regime. It is frequently associated with Utah juniper, alligator juniper, oneseed juniper, and Rocky Mountain juniper.

Figure 2-15—In pinyon pines, portions of three growing seasons are required to produce mature seeds. The exact timing of the developmental events varies among pinyon species and with the elevation or latitude of the pine stand. Great Basin National Park. (Photo by Rick Miller, Oregon State University.)

Trees can grow up to 50 feet but are usually smaller (Cronquist et al. 1972). Needles are 0.8–1.6 inches (20–40 mm) long and mostly two per fascicle, but occasionally 1 or 3 (see pinyon varieties below). Male cones are 0.24 inches (6 mm) long. Female cones ovoid 0.8–2 inches (20–50 mm) long with seeds 0.39–0.63 inches (10–16 mm) long occurring subterminal or lateral on the branchlets.

Pinyon Varieties

Pinus monophylla var. *monophylla* is the predominant pinyon found throughout the Great Basin. However, two other subspecies are recognized by some taxonomists, which occupy different environments (Bailey 1987; Cole et al. 2008a; Eckenwalder 2009; Little 1968). Variety *Pinus monophylla* var. *californiarum* (Bailey 1987) like var. *monophylla*, grows in areas with wet winters and dry summers, but with warm rather than cold winters. It primarily grows in southern California south into Baja. Both Bailey (1987) and Cole et al. (2008a) suggest *Pinus monophylla* var. *californiarum* should be a separate species, *Pinus californiarum*. Variety *fallax*, is believed to be genetically closer to twoneedle pinyon and is considered a variety of this species by some taxonomists (Cole et al. 2008a; Little 1968). It grows just south of the Mogollon Rim in Arizona in areas with wetter summers than the other singleleaf pinyon varieties, but like var. *californiarum*, warm winters. Variety *fallax* usually grows south and at lower elevations than twoneedle pinyon and in areas with more extreme May and June droughts similar to californiarum, which is associated with stronger winter precipitation (Cole et al. 2008a).

Morphology

Foliage

Juniper leaves are reduced small scales that form an oblique triangle. The edges are serrated in all of the western species with the exception of Rocky Mountain juniper, which are smooth, similar to eastern redcedar (Willson et al. 2008). Leaf margins in adult foliage are slightly recurved, creating a slight cupping, which seals one leaf margin against the other—forming a chain-like cylinder of scales against the stem (figs. 2-10, 2-12; Miller and Shultz 1987). The leaf surface facing toward the stem (adaxial) is not exposed to sunlight or air movement, which significantly reduces potential transpiration. The epidermis is heavily cuticularized, to the extent the cell lamina are nearly obliterated. In western juniper, the majority of stomates are distributed on the protected adaxial surface (fig. 2-16a). On the surface facing away from the stem, stomates are restricted to the base of the leaf that is entirely covered by the adjacent subtending scale leaf (fig. 2-16b).

Leaf morphology of western juniper (and most likely other junipers in the West) allows for increased drought avoidance through low surface-to-volume ratios, thick cuticle layers, and protected stomates. Mean maximum leaf conductance (transpiration passing through the leaf surface to the atmosphere) per unit of leaf area is less in western juniper than values reported for other conifer species (Miller and Shultz 1987). Juniper leaf scales develop slowly and can remain functional on the tree for 6 or more years (Peek et al. 2006; Reich et al. 1992) leading to nearly constant leaf area displayed throughout the year (Grier et al. 1992; Hicks and Dugas 1998; Miller et al. 1992). Juniper foliage has a high cost of development and low assimilation rates (Reich et al. 1992), typical of the foliage of plants adapted to dry, nutrient-poor environments (Reich et al. 1992). In juvenile foliage, the triangular leaf scales in juniper do not compress against the adjacent leaf scale but spread from the stem (fig. 2-17). This growth form is found on very young trees and sometimes at the base of adult trees after top removal. The juvenile foliage allows for increased sunlight and absorption of CO_2 resulting in higher photosynthesis rates compared to adult foliage but has poorer stomatal control resulting in lower water use efficiency than adult foliage (Miller et al. 1992).
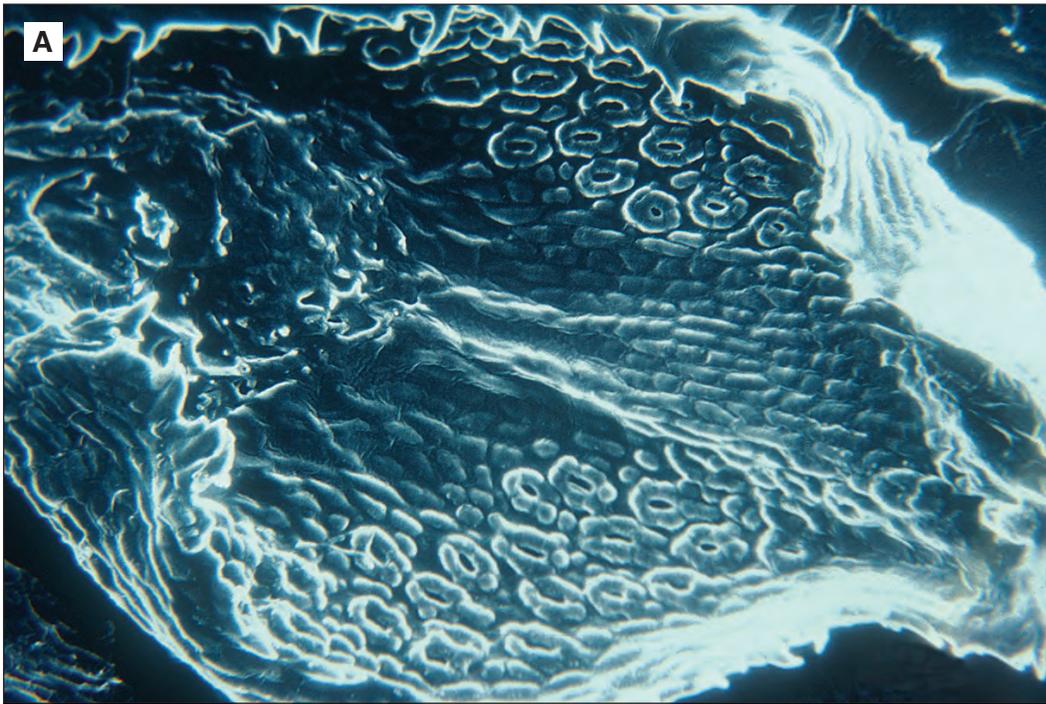
Figure 2-16—(A) The leaf surface of western juniper that faces the stem (adaxial), where the majority of stomates are located, is not exposed to sunlight or air movement—which significantly reduces potential transpiration and increases photosynthetic efficiency per units of water used. (B) On the exposed leaf scales of western juniper facing away from the stem, stomates are restricted to the base of the leaf scale, which are entirely covered by adjacent subtending scale leaf. The epidermis on the outer leaf scale is heavily cuticularized, which further reduces water loss in hot, dry environments. (Photo from Miller and Schultz 1987.)

Figure 2-17—In juvenile juniper foliage, the triangular leaf scales do not compress against the adjacent leaf scale but spread from the stem. The juvenile foliage on juniper allows for increased sunlight and absorption of CO_2 resulting in higher photosynthesis rates compared to adult foliage but has poorer stomatal control resulting in lower water use efficiency than adult foliage. (Photo by Rick Miller, Oregon State University.)

Pinyon pine needles have thick waxy cuticles with the stomates set within small depressions increasing the boundary layer protecting them from arid environments. Singleleaf pinyon needles contain 2–7 resin ducts and 17–30 stomatal lines (Cole et al. 2008a). Twoneedle pinyon needles contain 2–3 resin ducts and 8–15 stomatal lines. The resin ducts are important defense mechanisms against insects and other animals in creating a distasteful substance and disrupting digestion. Singleleaf pinyon needles can stay functional for 8–12 years (Tausch and West 1987).

Roots

Work on juniper and pinyon roots is very limited and is based on a low number of tree excavations, especially for mature trees. Root depth, lateral spread, and shoot-to-root ratio within each species likely vary with substrate, depth to restrictive layer, fracturing of the restrictive layer, and moisture availability. Mature pinyon pine, western juniper, and likely Utah juniper have extensive lateral roots that typically extend one to three times the height of the tree, have a fine root mat directly beneath the tree canopy, and lack or have just a stunted tap root (fig. 2-18; Everett 1984; Russell and Honkala 1990; Rick Miller, Professor Emeritus, Range Ecology, Oregon State University, Corvallis, Oregon, personal observation, 1985. Juniper and pinyon roots are capable of penetrating depths of more than 20 feet, especially in substrates that are fractured, allowing both water and roots to attain considerable depths (fig. 2-19; Cannon 1960; Foxx and Tierney 1987; Rick Miller, personal observation). However, taproots are important for tree seedlings. Taproot growth in twoneedle pinyon is rapid, averaging 7–11 inches in the first year (Harrington 1987). In western juniper, root development is primarily a taproot during the first 10 years of growth (Kramer 1990).

Figure 2-18—Mature pinyon pine, western juniper, and Utah juniper (see fig. 2-5) have extensive lateral roots that typically extend one to three times the height of the tree. They have a fine root mat directly beneath the tree canopy and no—or just a stunted—tap root. Southcentral Utah. (Photo by Rick Miller, Oregon State University.)

Figure 2-19—Juniper and pinyon roots are capable of penetrating depths more than 20 feet, especially in substrates that are fractured, allowing both water and roots to attain considerable depths. Singleleaf pinyon. Southcentral Utah. (Photo by Rick Miller, Oregon State University.)

As trees mature, taproot development declines and often dies when juvenile foliage is replaced by adult foliage (Young et al. 1984). After 10 years, lateral root development increases accounting for 65 percent of the root biomass in trees 30–35 years old (Kramer 1990). Trees less than 0.25 inches and 0.25–0.6 inches in basal diameter had lateral roots extending 90–100 inches and 120–225 inches, respectively (Miller et al. 1990). Seasonal growth and mortality of roots are related to soil moisture and temperature. Fine root growth in Utah juniper followed a seasonal progression from the surface toward the lower soil depths as the growing season progressed. (Peek et al. 2006). As temperatures increased, root loss increased in the upper soil layers. The average life span of shallow and deep fine roots was 5–15 months, respectively.

Seed and Seedling Ecology

The production of pinyon and juniper seeds and their subsequent fates determine the ability of the different species to maintain existing populations and expand into new areas. In this section, we summarize and update earlier publications on the seed and seedling ecology of pinyon and juniper species (Chambers et al. 1999a,b). We begin by reviewing factors that influence seed production, and then discuss the potential pathways and fates of seeds from seed or fruit development to seedling establishment. The seed fates of pinyon and juniper species differ and are illustrated in seed and seedling fate diagrams for pinyon (fig. 2-20) and junipers (fig. 2-21; Chambers et al. 1999a).

Seed Production

In pinyon pines, portions of three growing seasons are required to produce mature seeds (fig. 2-20). The exact timing of the various developmental events varies among pinyon species and with the elevation or latitude of the pine stand. Like all pines, pinyons are monoecious and both male and female cones occur on the same plant (Mirov 1967; Vidakovic 1991). Male and female cones (strobili) of twoneedle pinyon form between August and October and develop the following spring (Jeffers 1994; Little 1938, 1941). Pollination is usually complete by the end of June, but seeds do not mature until September of the following year. Cone production can occur on trees 3–4 feet tall and 10–20 years old, but the largest crops are produced on mature trees, which are usually greater than 20–30 years old (Jeffers 1994).

Many of the seeds produced are unfilled or are eaten by insects before they mature. In twoneedle pinyon, nearly half of the seeds in mature cones are empty (Ligon 1978; Vander Wall and Balda 1977). In singleleaf pinyon in the Pine Nut Range of western Nevada, 18 percent and 21 percent of the seeds were empty during two successive years (Stephen Vander Wall, Professor, Department of Biology, University Nevada, Reno, unpublished data). A variety of insect species feed on the developing cones and seeds of pinyon pines. The two most important types are *Conophthorus* and *Dioryctria* (Christensen and Whitham 1991, 1993; Keen 1958). *Conophthorus* species are cone beetles whose larvae eat through the cone scales and seeds, causing second-year cones to die. In some years, beetle larvae can destroy up to 90 percent of the cone crop (Keen 1958). The larvae of *Dioryctria albovitella*, juvenile moths known as "pine coneworms," attack terminal shoots and developing cones, reducing cone production, viable seed production, and seed mass (Christensen and Whitham 1991, 1993; Mueller et al. 2005b). Twoneedle pinyon trees susceptible to moth attack had 93–95 percent lower cone production—and surviving cones produced 31-37 percent fewer viable seeds—resulting in a 96–97 percent reduction in whole tree viable seed production (Mueller et al. 2005b). Because seed germination is positively associated with seed size, seedling biomass, and seedling height, trees suffering chronic herbivory likely produce lower quality offspring (Mueller et al. 2005b).
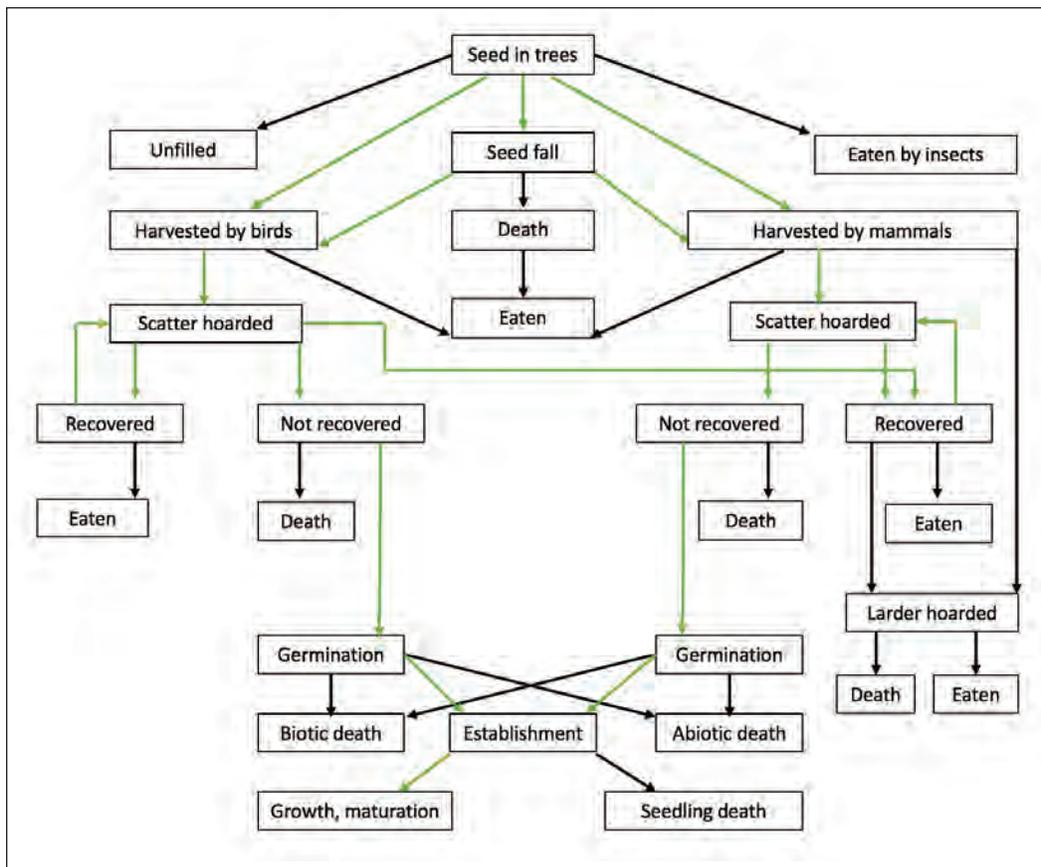
Figure 2-20—Seed fate diagram for pinyon pines from Chambers et al. 1999. Slightly more than 2 years are required to produce mature "seeds in the trees." Many seeds are unfilled because of environmental and developmental constraints. Others are eaten by insects or animals before they fully develop. Seeds that do mature are harvested or eaten by a variety of animals, some of which act as seed predators and others as seed dispersers. Birds (corvids such as jays and nutcrackers) and rodents (chipmunks, mice, and kangaroo rats) harvest and disperse pinyon seeds. Although birds eat some seeds, they scatter-hoard many others in shallow caches in the soil. A portion of cached seeds are recovered and eaten at a later date or recovered and moved to new locations to avoid predation by competitors. Cached seeds not recovered by birds or rodents either die of abiotic or biotic causes or survive to germinate. A portion of rodent harvested seed may end up in large underground larders, where they likely are consumed or die due to unsuitable germination and establishment conditions. Regardless of the disperser, if the seed ends up in a microhabitat suitable for germination and growth, seedling establishment can occur. Seedlings are then susceptible to rodent predation, insects, or pathogens. Seed dispersed to a microhabitat unsuitable for growth and survival will die.

Jeffers (1994) reported mean filled seeds per cone in twoneedle pinyon as 10–20 seeds, while other reports range from about four in twoneedle pinyon (Ligon 1978; Vander Wall and Balda 1977) to about 16 in singleleaf pinyon after accounting for the various forms of predispersal seed losses (Vander Wall 1997). During a year of heavy seed production, large trees can produce more than 1,000 cones, although the average tree produces less. A typical mean seed crop during a year of heavy cone production ranges from 2,000–8,000 filled seeds per tree, but values for individual trees vary tremendously. Seed production in a stand of singleleaf pinyon in the Pine Nut Range of western Nevada was 1,873 filled seeds per tree in a year of moderate cone production and 5,936 filled seeds per tree in a year of relatively heavy cone production (Vander Wall 1997).
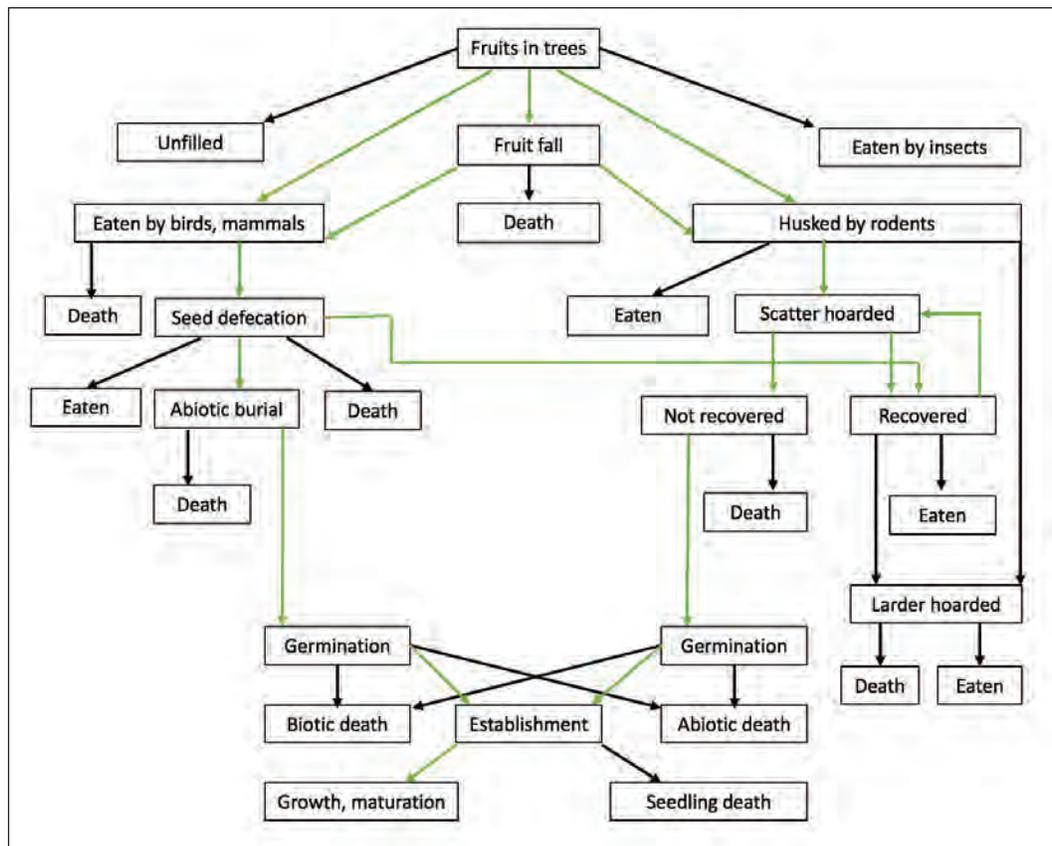
Figure 2-21—Seed fate diagram for juniper from Chambers et al. 1999. In juniper species, seed maturation takes two growing seasons. Many of the seeds produced are unfilled due to environmental and developmental constraints. Others are eaten by insects or destroyed by animals. Seeds that do develop and mature are harvested or eaten by a variety of animals, some that act as seed predators and others that serve as seed dispersers. Juniper seeds are adapted for dispersal by frugivorous birds and mammals that either eat the seeds while they are on the trees or after they fall to the ground. If the fruits are ingested by frugivorous mammals, the pericarps are digested by the animals and seeds defecated onto the ground.

Depending on species of juniper and animal disperser, some seeds are killed. Many survive ingestion only to be eaten later by other animals or die of abiotic or biotic causes. Surviving seeds may remain on the soil surface or be buried abiotically where additional mortality or germination may occur. Seeds can also be scatter-hoarding by rodents such as chipmunks, mice, and kangaroo rats. Many cached seeds are later recovered and eaten, but others are probably recached. Some cached seeds are not recovered by rodents and die of abiotic or biotic causes, while others survive to germinate. Some seeds may be placed in large underground larders, where the majority are either eaten by the rodents or die because of unsuitable conditions for germination and establishment. Regardless of the disperser, if the seed has been dispersed to a microhabitat with environmental conditions suitable for germination and growth, seedling establishment can occur. However, seedlings may still be susceptible to rodent predation, insects, or pathogens. If the seed has been dispersed to a microhabitat unsuitable for growth and survival, mortality by biotic or abiotic causes is inevitable.

Climate warming and associated droughts have the potential to cause significant declines in pinyon cone production. In twoneedle pinyon, seed cone production declined by 40 percent from the 1974 decade (1969–1978) to the 2008 decade (2003–2012) in revisited stands throughout New Mexico and northwestern Oklahoma (Redmond et al. 2013). Seed cone production was negatively correlated with late summer temperatures at the time of cone initiation. Further, declines in seed cone production were greatest among populations that experienced the greatest relative increases in growing season temperatures, which were

the populations located at the cooler, upper elevations. Drought predisposes twoneedle pinyon to bark beetles, which feed on the phloem of the trees (Gaylord et al. 2013). However, less is known about effects on insects that attack cones and seeds.

In Utah juniper and western juniper, it takes two growing seasons following pollination to produce mature "fruits in the trees" (figs. 2-10, 2-11, 2-21; Adams 1993; Fechner 1976). Most juniper species (table 2-2), including oneseed juniper, are dioecious with male and female cones produced on separate trees (Adams et al. 2014; Johnsen 1962). However, western juniper and California juniper may be either dioecious or monoecious, and Utah juniper is generally monoecious (Adams 1993; Johnsen and Alexander 1974; Tueller and Clark 1975). Reproductive phenology varies with species and climate, but in most junipers, male cones mature in late summer to fall and release pollen in spring as female cones are emerging (Fechner 1976; Tueller and Clark 1975; Eugene Schupp, Professor, Wildland Resources/Ecology Center, Utah State University, Logan, Utah, unpublished data). A delay of several months to a year or more between pollination and fertilization is typical with maturation occurring a year or more later.

Some species produce only a single (sometimes two) seed per cone (e.g., Utah juniper and oneseed juniper), others produce one to three seeds per cone (western juniper), while still others mature four to five or more seeds per cone (e.g., alligator juniper, also known as checkerbark juniper) (Adams 1993; Dimitri et al. 2018). Ripe fruits can remain on the tree for 2 or even 3 years in some species. Although seed bearing begins at 10–20 years of age (Johnsen and Alexander 1974), significant fruit production starts at 50–70 years and continues for centuries (Eddleman 1984; Miller and Rose 1995; Noble 1990; Tueller and Clark 1975). As with pinyon pine, there is extensive loss of potential seed production between pollination and seed maturation with many fruits failing to fill or being eaten by insects or other predators (fig. 2-21). Junipers ripen many fruits with fully developed seed coats—but without an embryo or endosperm (Adams et al. 2014; Fuentes and Schupp 1998; Johnsen and Alexander 1974; Noble 1990). The proportion of unfilled seeds is highly variable, both among and within species and over time.

In 1993 in Tintic Valley, western Utah, seed fill of Utah juniper varied from 0 to 17.3 percent among individuals with a mean of 5.6 percent filled (Fuentes and Schupp 1998). In the same year, Utah juniper had less than 1 percent of its seeds filled at U.S. Army Dugway Proving Grounds about 50 km (31 miles) away. Three years later it produced roughly 33 percent filled seeds (Fuentes and Schupp 1998; Eugene Schupp, professor Utah State University, unpublished data). Similarly, in 2010, seed fill in Utah juniper ranged from 0 percent for Big Cottonwood Canyon, Utah, to 79 percent for Sedona, Arizona (Adams et al. 2014). In 2011, the same Big Cottonwood Canyon population had 0.4 percent filled seeds, while the Sedona, Arizona, population had dropped to 7.2 percent. The causes of unfilled seeds are poorly understood in juniper, but it is thought that the proportion of filled seeds varies with the age, structure, density, and community composition of a stand, in addition to physiography and weather during pollination and/or seed development (Noble 1990).

Seed production can be significantly impacted by invertebrate and vertebrate predation on developing seeds. Caterpillars of the moths *Periploca atrata* feed on juniper seeds, consuming the entire embryo and endosperm (Furniss and Carolin 1977; Keen 1958; Powell 1963), and the cecidomyiid midge *Walshomyia juniperina* (fig. 2-22a,b) eats the fruit of Utah and western junipers (Furniss and Carolin 1977). The juniper-berry mites *Trisetacus quadrisetus* and *Eriophyes ramosus* have been known to destroy the entire fruit crop of some trees (Furniss and Carolin 1977; Morgan and Hedlin 1960). Fernandes and Whitham (1989) found that the larvae of an unidentified beetle (*Anobiidae*) infested oneseed juniper near Sedona, Arizona, and that insect attack increased the likelihood of fruit abscission.
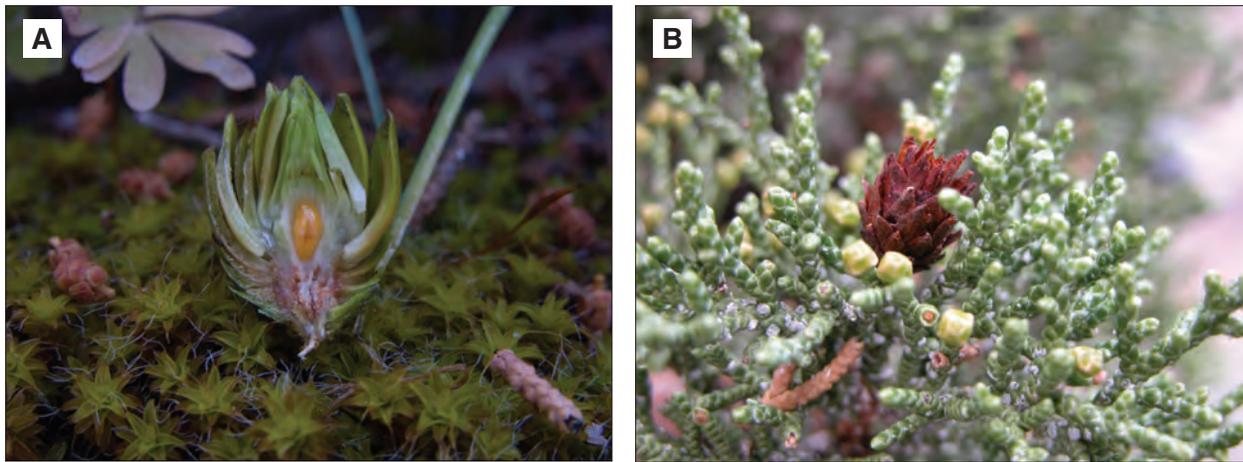
Figure 2-22—Artichoke-like gall (a and b) located on the brachlet of junipers (frequently misidentified as a reproductive structure) is formed by the midge larvae *Walshomyia juniperina.* (Photo by Rick Miller, Oregon State University.)

In an early study of western juniper, the chalcidoid wasp *Eurytoma juniperina* destroyed about 25 percent of the seed crop near Hilt, California, in 1913 (Keen 1958). More recently, a study at two sites in northeast California had over 30 different species of arthropods (insects and arachnids) associated with the fruits or seeds of western juniper (Dimitri et al. 2014, 2017; Tonkel et al. 2014). These included frugivorous and granivorous species (cone- and seed-feeders, respectively), predators of other insects, parasitoids (insects that develop from eggs laid inside other insects, ultimately killing their host), hyperparasitoids (parasitoids of other parasitoids), and inquilines (species that harmlessly cohabitate with other species). The seed-destroying arthropods collected included species of several insect orders, such as beetles, moths, wasps, and flies, as well as a family of mites. Seed damage by all arthropod taxa varied both spatially and temporally, and masting (cone producing) events overwhelmed the responses of seed predators (Dimitri et al. 2017). This research shows that junipers harbor a diverse and complex community of arthropods in their berries, some of which can drastically reduce a tree's seed-yield within a given year.

Seed Dispersal Processes

To understand plant dispersal processes, information on both the types and behaviors of the seed dispersers and the effectiveness of dispersal for plant establishment is necessary. Disperser effectiveness has been defined as "the contribution a disperser makes to the future reproduction of a plant" population (Schupp 1993; Schupp et al. 2010). Effectiveness has a quantitative component (the number of seeds dispersed) and a qualitative component (the likelihood that a dispersed seed will survive to produce a new plant in the population). Here, we examine the available information on both the types and behaviors of the animals that disperse pinyon and juniper, and the effectiveness of those dispersers.

Seed dispersal of pinyon pines by birds has been relatively well studied. Several species of corvids—jays and nutcrackers (for common and scientific animal names see Appendix A)—that store seeds in shallow caches in the soil disperse pinyon pine seeds (Vander Wall 1990; Vander Wall and Balda 1981). Pinyon pine cones and seeds are well adapted for dispersal by birds. The seeds are large and nutritious (Botkin and Shires 1948). The dry mass of the edible portion of twoneedle pinyon seeds weighs 125–225 mg and that of singleleaf pinyon seeds weigh 200–300 mg. Twoneedle pinyon seeds are very high in fats (63 percent), whereas singleleaf pinyons have less fat (26 percent) but are rich in carbohydrates with 60 percent (Botkin and Shires 1948, values adjusted to

dry weight). The cones are weakly constructed so that the seeds can be easily extracted by corvids with long pointed beaks, such as Clark's nutcracker and pinyon jays (Vander Wall and Balda 1981). Seeds are enclosed in deep pockets and held by thin flanges so that they do not fall readily from the cones (fig. 2-15). Seeds of twoneedle pinyon and singleleaf pinyon have thin coats, and even small birds such as mountain chickadees and red-breasted nuthatches are able to open the seeds. Unlike many conifers, cones are primarily pointed to the side and upward, which not only retards seed loss, but increases the visibility and availability of the seeds to avian dispersers.

Birds typically disperse seeds from several feet to 3.1 miles (Vander Wall and Balda 1981). Differences in dispersal distances exist among different bird species with the more solitary scrub jays seldom dispersing seeds more than 0.62 miles, the gregarious pinyon jays carrying seeds slightly farther, and Clark's nutcrackers carrying seeds as far as 13.7 miles (Vander Wall and Balda 1981). While jays typically place one seed in each cache site, nutcrackers cache from 1–10 seeds with a mean of about 4 seeds per cache. While many of the cached seeds are recovered and eaten by the birds, others are not recovered and thus have the potential to germinate and produce seedlings. Although not well documented, some of the recovered seeds are probably re-cached in new locations to avoid predation by competitors in what has been termed "shell games" in animals (e.g., Vander Wall 1995; Vander Wall and Joyner 1998). These re-cached seeds may be recovered once again and then either eaten or moved to yet another location. The total number of seeds cached can be phenomenal. In a good seed crop year, individual Clark's nutcrackers scatter-hoard (creating multiple caches) between 22,000 to 33,000 twoneedle pinyon seeds (Vander Wall and Balda 1977) or 17,900 singleleaf pinyon seeds (Vander Wall 1988). Ligon (1978) estimated that a flock of 250 pinyon jays could cache about 4.5 million pinyon seeds over 5 months.

Seed caching by corvids has important consequences for the fate of a pinyon pine seed. It can be quantitatively effective for pinyon pine as large numbers of seeds are cached and—especially in large seed crop years—many are left unrecovered to germinate and possibly establish. Bird dispersal can also be qualitatively effective because pinyons almost always require burial for establishment, and birds bury the seeds 0.8–1.6 inches (20–40 mm) in the soil (Stephen Vander Wall, Professor, Department of Biology, University of Nevada, Reno, personal observation, 1985). However, pinyon often have a nurse plant and shading requirement and birds tend to place most seeds in interspace environments, not in more favorable microhabitats under trees or shrubs (Stephen Vander Wall, personal observation).

Pinyon pine seeds are also cached by rodents. Rodents prefer pinyon over other native seeds, such as desert peach, antelope bitterbrush, and Utah juniper (Vander Wall et al. 2019). Unlike corvids, most rodents forage for seeds on the ground after the seeds have fallen from the tree. In the Pine Nut Range of Nevada, deer mice, pinyon mice, Great Basin pocket mice, Panamint kangaroo rats, least chipmunk, and white-tailed antelope ground squirrel all scatter-hoard singleleaf pinyon seeds (Hollander and Vander Wall 2004; Vander Wall 1997). In one study, 1,000 labeled singleleaf pinyon seeds were placed under 5 source trees—of which 69 percent were consumed by rodents and 24 percent were scatter-hoarded 0.2–1.2 inches (5–30 mm) deep (Vander Wall 1997). Rodents placed 36 percent of these scattered caches under shrubs, 39 percent in the open, and the rest (25 percent) at the edge of shrub canopies. In a second study, all 6 of the above species harvested over 90 percent of the singleleaf pinyon seeds presented to them and scatter-hoarded from 31 percent (pocket mouse) to 66 percent (chipmunk) of the seeds they harvested (Hollander and Vander Wall 2004). Pinyon mice and deer mice distributed seeds to more cache sites (i.e., put fewer seeds in caches) than the other species and, consequently, created more potential establishment sites.

All species scatter-hoarded seeds at depths suitable for seedling emergence, but pocket mice and pinyon mice cached under shrubs more than the other species, increasing the probability of establishment. Also, in the San Francisco Mountain volcanic fields of northern Arizona, pinyon mice and brush mice placed most (70 percent) twoneedle pinyon seed caches in small-particle soil with rock cover where seeds were subject to less pilfering and had a higher probability of germination and establishment (Pearson and Theimer 2004). In a study conducted in the Pine Nut Mountains in western Nevada, rodents removed artificially cached singleleaf pinyon pine seeds more rapidly than other native seed species that included bitterbrush, desert peach, and Utah juniper (Vander Wall et al. 2019). Although most of the removed pinyon pine seeds were eaten (80 percent) and not re-cached, the new caches found were on average dispersed farther than the other three species across seasons. Pinyon pine had the largest proportion of seeds that were not recovered in both trials (74 percent), likely because seeds were taken outside of the search area.

In all of these studies, rodents were qualitatively effective seed dispersers because they buried the seeds and placed many of them in locations favorable for germination and establishment. However, many of the seeds harvested by rodents are consumed or placed in larders (large caches with high numbers of seeds), where they have little or no chance of establishment (Vander Wall 1990). From a quantitative perspective, rodents may be less important to see dispersal and establishment because, except in heavy seed crop years, most seeds do not fall to the ground before they are harvested by birds.

Less is known about dispersal of juniper species. Most species of juniper have been assumed to be dispersed largely by birds, but mammals can also disperse significant numbers of seeds. Recently it was noted that even harvester ants can carry oneseed juniper berries up to 50 feet to their nests, remove the pulp, and deposit the seeds outside their mounds (Adams and Thornburg 2010). Unlike bird dispersal of pinyon, in which the seeds are scatter-hoarded, bird dispersal of junipers is by frugivory and endozoochory, in which the seeds are ingested and passed through the gut track. At least 12 species of birds feed on fruits and potentially disperse seeds of western juniper (Maser and Gashwiler 1978), 13 species are known to disperse Ashe juniper (Chavez-Ramirez and Slack 1994), and 52 species have been observed feeding on eastern redcedar (Van Dersal 1938). Of the wide diversity of bird species involved, the most important for juniper dispersal are probably members of the highly frugivorous subfamily *Turdinae* (*Muscicapidae*) such as bluebirds, Townsend's solitaire, and American robin—and two members of the family *Bombycillidae*, the Bohemian waxwing and cedar waxwing (Chavez-Ramirez and Slack 1994; Gabrielson and Jewett 1940; Holthuijzen and Sharik 1985; Poddar and Lederer 1982; Salomonson 1978).

The rounded and more-or-less fleshy cones of junipers are well suited for frugivorous dispersal, especially by birds (Salomonson 1978; Salomonson and Balda 1977). Fruits are conspicuously colored blue or reddish and are easily accessible on the outer layers of the foliage. The fleshy portion of a juniper cone is a reasonably rich energy source. Oneseed juniper cones have an energy content of 4.57 cal/mg (Salomonson 1978), while western juniper cones have a nutritional content of 4 percent protein, 16 percent lipid, and 46 percent carbohydrate (Poddar and Lederer 1982). The thick, hard seed coat allows seeds to pass undamaged through the guts of most birds and mammals.

Despite large losses from abortion and predation, substantial fruit production can still occur. Salomonson and Balda (1977) estimated that winter feeding territories of Townsend's solitaires in northern Arizona contained about 27.2 million oneseed juniper cones per hectare in a good year and 1 million cones per hectare in the following poor year. Dispersal distances and patterns vary depending on the bird species and the juniper species (Chavez-Ramirez and Slack 1994; Holthuijzen and Sharik 1985). Birds dispersed

eastern redcedar juniper seeds to perch sites an average of 40 feet away (Holthuijzen et al. 1987), while American robins dispersed Ashe juniper seeds to perches 144 feet away (Chavez-Ramirez and Slack 1994).

The effectiveness of birds as dispersal agents varies among species of juniper. For species such as western juniper, birds appear to disperse the majority of the seeds and, thus, are quantitatively important (Lederer 1977; Poddar and Lederer 1982). Bird dispersal is often qualitatively effective as most birds deposit seeds primarily in more favorable under-shrub or tree microhabitats and only occasionally carry seeds to open microsites (Dimitri and Longland 2017). Also, seeds tend to be deposited singly or in small groups and, thus, may be less likely to die from density-dependent seed predation or competition (Chavez-Ramirez and Slack 1993; Schupp 1993). However, some bird species, such as cedar waxwings, travel in flocks and use the same perches repeatedly, resulting in high seed densities under single trees. Another limitation of bird dispersal is that seeds are deposited on the soil surface and are dependent on other mechanisms of burial (Longland and Dimitri 2016; Johnsen 1962).

Mammals have been considered to be unimportant dispersers of juniper seeds in the past but may be quite important for certain juniper species depending on the type of dispersal. Endozoochorous dispersal occurs when seed is consumed and passed through the gut. Mammals that consume and disperse juniper seeds include woodrats, Virginia opossum, Nuttall's cottontail, desert cottontail, black-tailed jackrabbit, coyote, red fox, gray fox, black bear, ringtail, racoon, mule deer, white-tailed deer, assorted livestock, and many different rodent species (Chavez-Ramirez and Slack 1993; Dimitri et al. 2017; Johnsen 1962; Longland and Dimitri 2016; Martin et al. 1951; Maser and Gashwiler 1978; Miller 1921; Parker 1945; Salomonson 1978; Schupp et al. 1997a,b; Willson 1993). All species that ingest seeds pass at least some intact seeds and, in some cases, seed germination is increased (Johnsen 1962; Miller 1921; Schupp et al. 1997a). Certain mammals that ingest seeds may be quantitatively more important in dispersing some species of juniper than others. Coyotes appear to be one of the few mammalian dispersers of western juniper (Schupp et al. 1997b), but Utah juniper appears to be dispersed more by cottontail rabbits and jackrabbits. In general, mammalian endozoochorous dispersal may not be effective for plant establishment. Seeds are deposited on the soil surface, often in high densities, and usually in the open and not in more favorable shaded environments. However, passage through the gut tract can be relatively slow, resulting in long dispersal distances.

The role of rodents likely varies among juniper species, with rodents directly dispersing junipers with drier cones such as Utah juniper and acting as secondary dispersers, removing seeds from bird and carnivore scat, in juniper species with more fleshy cones such as western juniper (Dimitri et al. 2017). Seed removal experiments of western and Utah juniper berries and seeds conducted in the spring, summer, and fall showed that the majority of seeds and berries of both species were removed in just over 2 weeks (Dimitri et al. 2017). Pinyon mice removed the most cones and seeds from all stations at both sites, so although Utah juniper berries and seeds may not be preferred by some species, such as chipmunks, juniper might be more preferred than previously thought by other rodent species.

Vander Wall et al. (2019) found conflicting evidence of low preference by rodents for Utah juniper seeds. While Utah juniper seeds were removed from artificial caches more slowly than pinyon pine and desert peach in summer and fall, and many seeds were dug up and left, the caches that were made were dispersed farther on average than desert peach and bitterbrush caches. Longer dispersal distances have previously been thought to indicate higher preference. It is likely that the mixed results are due to variation in preference or value among scatter-hoarding rodent species (Vander Wall et al. 2019).

The importance of rodent dispersal of juniper is indicated by clumps of juniper seedlings observed emerging from caches in the spring (Vander Wall 1990; 1997; Eugene Schupp, Professor, Wildland Resources/Ecology Center, Utah State University, Logan, Utah, unpublished data). In west-central Utah, a minimum of 16 to 33 percent of all naturally occurring Utah juniper recruits less than or equal to 6.6 feet tall emerged from rodent caches (Eugene Schupp, Professor, Wildland Resources/Ecology Center, Utah State University, Logan, Utah, unpublished data). Many juniper berries are husked before they are cached by rodents, likely enhancing germination and emergence (Dimitri et al. 2017). Pinyon mice and Panamint kangaroo rats husked Utah juniper seeds from cones at seed removal stations, and whole Utah juniper cones and husked seeds were found in Panamint kangaroo rat burrows (Dimitri et al. 2017).

To quantify the role of rodents in the dispersal of Utah juniper, 500 seeds (husked fruits) labeled with scandium-46 were placed under four different source trees in the Pine Nut Range, Nevada, and their fates were monitored (Stephen Vander Wall, Professor, Department of Biology, University of Nevada, Reno, unpublished data). Slightly less than half of the seeds were taken (41 percent) and of those, 27 percent were found in caches. The study was conducted in mid-summer when other, possibly more desirable, shrub and forb seeds were available.

Over the course of 21 trials using radio-labeled seeds placed under juniper canopies spread out across a western juniper site on Shinn Peak in northeastern California, 1,837 of 2,200 (83.5 percent) seeds were removed, and of those removed, 22.6 percent were cached (Dimitri et al. 2017). Pinyon mice made the most caches, placing them under juniper canopies and shrubs, and caches were generally small and shallow with an average dispersal distance of 22 feet from the source. Over 21 trials across a Utah juniper site in the Pine Nut Mountains in western Nevada, rodents removed 2,061 Utah juniper seeds from below canopies (93.7 percent), and of those removed, 22.3 percent were found in scatter-hoards (Dimitri et al. 2017). Pinyon mice again made the most caches, which were again small and shallow with most caches being found under shrub canopies at an average dispersal distance of 17.5 feet. Although Panamint kangaroo rats mostly larder-hoarded Utah juniper seeds, they made 11 caches that were found generally in open microsites with an average dispersal distance of 27.5 feet and a maximum 90 feet (Dimitri et al. 2017).

To investigate the relative roles of frugivorous birds and rodents in the dispersal of western juniper, bird and rodent removal of intact western juniper berries versus seeds cleaned either manually or by passage through birds was quantified at two northeastern California sites (Longland and Dimitri 2016). Six species of rodents were observed harvesting bird-passed and hand-cleaned western juniper seeds, including yellow-pine chipmunk, California ground squirrel, California kangaroo rat, Great Basin pocket mouse, deer mouse, and pinyon mouse (Longland and Dimitri 2016). Only California ground squirrels and deer mice were photographed taking intact juniper berries as well as seeds. All of the granivorous rodent species are scatter-hoarders, with the possible exception of California ground squirrels. Seedling emergence experiments showed that seed burial was necessary for seedling emergence (Longland and Dimitri 2016). However, emergence was significantly greater for bird-passed than for hand-cleaned seeds, which both produced significantly more seedlings than intact berries.

In general, rodents are primary dispersers of both western and Utah juniper, and because they also harvest seeds defecated by birds, they are secondary dispersers of bird-passed seeds through diplochory (Dimitri et al. 2017; Longland and Dimitri 2016; Vander Wall and Longland 2004). Rodent preference for juniper berries appears to be mixed due to variation in preference or value among scatter-hoarding rodent species. Of the seeds that are harvested, the available data indicate 20 to 25 percent are cached with the depth of

the cache—and thus likelihood of establishment—depending on the rodent species. Birds may increase seed germinability through gut passage, but many rodents husk Utah juniper berries before caching them (Dimitri et al. 2017). As described for pinyon pine, caching of bird-passed or husked juniper seeds by rodents may be highly effective as many of the seeds are placed in favorable environments and have a high potential for establishment (Dimitri et al. 2017). This may be particularly important in semiarid ecosystems.

Seedling Establishment Processes

Seed characteristics, the types of microhabitats in which seeds are placed, and growing season condition are all important in determining seed fates after dispersal. In general, pinyon pines have short-lived seeds with little innate dormancy (Meeuwig and Bassett 1983). Thus, they form only a temporary seed bank with most seeds germinating the spring following dispersal (Chambers 2001). Density of seeds in the seed bank is highly dependent on the current year's cone crop. Pinyon pines exhibit regionwide synchrony in cone production with singleleaf pinyon masting (producing cones) every 2 to 3 years and twoneedle pinyon every 5 to 7 years (Tueller and Clark 1975). The potential for a large temporary seed bank is high during mast (cone producing) years, especially since many seeds probably remain unrecovered by animals. During nonmast years, the seed bank is probably quite sparse. Germination and establishment are most likely to occur when favorable growing season conditions follow a mast year (Chambers et al. 1999a).

In contrast to pinyon pines, junipers often have long-lived seeds. Tests of stored juniper seeds showed that 45-year-old Utah juniper seed still had 17 percent germination, 21-year-old oneseed juniper had 54 percent germination, and 9-year-old alligator juniper seeds had 16 percent germination (Johnsen 1959). The long-lived seeds are often highly dormant, with germination being delayed by impermeable seed coats, immature embryos, embryo dormancy, or the presence of inhibitors (Fisher et al. 1987; Young et al. 1988). A warm stratification period, followed by a cold stratification period, results in the highest germination for Rocky Mountain juniper (Barbour and Carvalho 2009; Johnsen and Alexander 1974) and Utah juniper (Chambers, unpublished data). The frequency of large seed crops is variable among species, populations, and individuals. Despite the suggestion in the early literature that heavy seed production may occur annually in western juniper (Deal 1990), this is not the case in northeast California (Dimitri et al. 2018) or other parts of the species range in the Great Basin (Rick Miller, Professor Emeritus, Range Ecology, Oregon State University, Corvallis, Oregon, personal observation, 2018). Most species tend to produce large cone crops every 2 to 5 years and to mature smaller numbers of fruits in intervening years (Johnsen and Alexander 1974; Noble 1990). Because junipers have highly dormant seeds, and often continuous seed production, they form persistent seed banks with germination of a single seed cohort extending over many years (Chambers et al. 1999a).

Successful establishment of pinyon and juniper seedlings is generally assumed to be higher in association with a nurse plant (fig. 2-23). Most studies reporting on this phenomenon have used an observational approach and simply examined the locations of seedlings in communities with varying tree and shrub cover. Higher numbers of pinyon and juniper seedlings are typically found under shrubs or adult trees than in interspace environments (Burkhardt and Tisdale 1976; Callaway et al. 1996; Eddleman 1987; Everett et al. 1986a; Johnsen 1962). In fully stocked stands of oneseed juniper, singleleaf pinyon, and western juniper, higher number of seedlings occur under trees than in interspace environments (Everett et al. 1986a; Johnsen 1962; Miller and Rose 1995). But in areas where western juniper is expanding into sagebrush communities, higher numbers of seedlings occur under sagebrush (52 to 65 percent) than under trees (17 to 31 percent) (Everett et al. 1986a; Johnsen 1962; Miller and Rose 1995).

Figure 2-23—These singleleaf pinyons have established beneath the canopy of black sagebrush. Successful establishment of pinyon and juniper seedlings is generally assumed to be higher in association with a nurse plant. Eagan Range, Nevada. (Photo by Rick Miller, Oregon State University.)

Microhabitats under shrubs and trees have several characteristics favorable for seedling establishment. These microhabitats often have higher concentrations of limiting nutrients, higher organic matter and total nitrogen, lower bulk densities, higher infiltration and soil water holding capacities, and higher rates of nutrient cycling (Chambers 2001; Doescher et al. 1987; Everett et al. 1986b; Klopatek 1987a; Roberts and Jones 2000; Stubbs and Pyke 2005). Lower irradiance and soil temperatures also occur under shrubs and trees (Chambers 2001; Stark 1994). And although these areas receive less effective precipitation than interspace areas, they experience higher relative humidity and delayed dry down relative to open areas and grasslands (Johnsen 1962; Stark 1994; Vetaas 1992).

The few studies that have examined establishment processes indicate that nurse plants facilitate tree-seedling establishment through shading and temperature modification. Although singleleaf pinyon juveniles growing in full sun exhibit higher growth rates than those beneath shrubs (Callaway et al. 1996), growth rates in western juniper were greater beneath the shrub canopy than the interspace (Miller and Rose 1995) and Utah juniper seedlings exhibited higher survival in artificial shade (Meagher 1943). Singleleaf pinyon seedlings show higher survival under shrubs (Callaway et al. 1996; Chambers 2001), while twoneedle pinyon juveniles 1.6–39 inches tall show higher survival under shrubs in high stress environments (Sthultz et al. 2007).

High water use efficiencies and conservative growth strategies of tree seedlings may be the primary mechanisms that allow the seedlings to survive in the shaded environment under adult trees and shrubs (Nowak et al. 1999). Seedlings of western juniper have higher leaf conductance and transpiration but greater total CO_2 assimilation per unit of leaf weight than adult trees (Miller et al. 1992). Seedlings of singleleaf pinyon have higher water use efficiencies than big sagebrush nurse plants (Drivas and Everett 1988) and when growing in association with the shrub than with the shrub removed (Callaway et al. 1996).

Differences exist in the nurse plant requirement both among and within species.

Singleleaf pinyon seedlings (fig. 2-23) very rarely establish in interspaces or open environments (Callaway et al. 1996; Chambers 2001; Everett et al. 1986a). One of the few mechanistic studies of seedling establishment in pinyon and junipers showed that establishment of singleleaf pinyon was dependent upon a complex set of interacting factors including microhabitat characteristics, growing season conditions, and animal species burying the seed. Timing and rate of seedling emergence for singleleaf pinyon were temperature dependent with the order of emergence paralleling mean growing season temperatures: tree and shrub interspace was greater than under shrub, under shrub was greater than under Utah juniper, and under Utah juniper was greater than under singleleaf pinyon (Chambers 2001). Seed burial was required for rooting and the highest emergence occurred from seed burial depths of 0.4–1.2 inches (10–30 mm), which are similar to burial depths by birds and rodents.

All seedlings, including those that emerged from seeds and transplants, died within the first year in interspace microhabitats. Survival in under-tree or under-shrub microhabitats depended on soil water availability and corresponded closely to soil water contents over the three-year study. Under-shrub microhabitats had more favorable soil and microenvironmental characteristics than under-tree microhabitats and had the highest seedling life spans for the first-year seedling cohort. Predation of pinyon seedlings by rodents was a significant cause of mortality with caged transplants exhibiting life spans that were 74 percent longer overall than uncaged transplants. These results are similar to Callaway et al. 1996, except that one to two singleleaf pinyon seedlings were alive in interspaces when the study ended 3 months after seedling emergence.

The available data indicate that juniper seedlings are capable of establishing over a wider variety of microenvironmental conditions than pinyon seedlings. First-year survival of Utah juniper seedlings in the Pine Nut Range in interspace microhabitats was less than in under-tree sites but was as high, or higher, than in under sagebrush sites (Jeanne Chambers, Research Ecologist, USDA Forest Service, Rocky Mountain Research Station, Reno, Nevada, unpublished data). In Tintic Valley, Utah, emergence of Utah juniper seedlings differed among open, shrub, and tree microhabitats and also among years with no clear patterns (Eugene Schupp, Professor, Wildland Resources/Ecology Center, Utah State University, Logan, Utah and José Gomez unpublished data).

In expanding western juniper populations, 18–47 percent of established seedlings occurred in interspaces (Burkhardt and Tisdale 1976; Miller and Rose 1995). For Utah juniper on stabilized Lake Bonneville sand dunes in Utah, most of the few natural juveniles occurred in interspaces (Eugene Schupp unpublished data). Also, in the southwestern grasslands and shrublands, oneseed juniper seedlings appear to readily establish in open environments (Johnsen 1962; Salomonson 1978). Differences between pinyon and juniper in the nurse plant requirement may be related to their physiological characteristics. Juniper species have greater drought tolerance and a higher capacity to obtain water resources from interspace microhabitats and shallow soils (Nowak et al. 1999; West et al. 2008). This may enable seedlings to establish in unshaded interspaces with higher soil temperatures.

Although nurse plants facilitate seedling establishment, they also compete for available resources. A shrub manipulation experiment at Sunset Crater National Monument, northeast of Flagstaff, Arizona, found that growth and survival of juvenile pinyon trees from less than 1.6 to 39 inches (40 mm to 1 m high) growing in association with Apache plume differed in high versus low stress environments (Sthultz et al. 2007). Average mortality of juvenile twoneedle pinyon in 2002 was higher in interspaces than under shrubs in a high stress environment, but lower in interspaces than under shrubs in a low stress environment. Stem growth showed the opposite trend. Removing or tying back the shrub had a negative effect on twoneedle pinyon in a high stress environment but a

positive effect in a low stress environment. These results are consistent with Bertness and Callaway's (1994) hypothesis that as environmental stress increases, the importance of facilitation increases while the importance of competition decreases.

Higher establishment of twoneedle pinyon and juniper species in interspace environments and under higher precipitation in the Southwest is likely due to precipitation patterns. While much of the Great Basin and more northern areas receive most precipitation during winter, the Southwestern grasslands and shrublands receive a higher percentage of summer precipitation (fig. 2-1; Romme et al. 2009). Summer precipitation in the Southwest may offset the beneficial microenvironmental effects of nurse plants for seedling establishment observed in the Great Basin. This effect may be most pronounced in less stressful, higher precipitation areas.

Effects of competition from grasses and other herbaceous vegetation on pinyon and juniper seedling establishment are not clear because of the lack of experimental data. It appears that competition from annual forbs and grasses can reduce the seedling survival of Utah juniper and singleleaf pinyon during the first year after emergence (Jeanne Chambers, Research Ecologist, USDA Forest Service, Rocky Mountain Research Station, Reno, Nevada, personal observation). Also, competition from established grasses reduces the initial establishment of oneseed juniper (Salomonson 1978). However, once seedlings are established (i.e., more than 1 or 2 years old), competition appears to have little effect on subsequent survival. In western Oregon, western juniper seedlings were capable of establishing into the community regardless of grass cover or ecological condition (Miller et al. 1994).

Insects, Nonvascular Plants, and Disease Associated With Pinyon and Juniper

Although mortality of pinyon and juniper trees are often attributed to drought, mortality is usually associated with multiple factors related to drought (Shaw et al. 2005) including fire, insects, and disease. Mortality in persistent woodlands is typically low (Landis and Bailey 2005; Waichler et al. 2001), but episodic events linked to drought can result in significant die-offs of young and old pinyons and sometime junipers at local and sometimes regional levels. There are many kinds of insects, nonvascular plants, and diseases associated with pinyon and juniper (table 2-3) that can have minor to significant impacts on growth, cone and seed production, and mortality. However, the magnitude of impact is often closely related to other factors influencing tree vigor including drought, ecological site characteristics, and tree density (Furniss and Carolin 1977; Gaylord et al. 2013; Greenwood and Weisberg 2008; Negrón and Wilson 2008; Shaw et al. 2005). The extent of impact also varies at local and geographical scales (Biondi and Bradley 2013; Shaw et al. 2005). Agents of particular importance include pinyon ips or bark beetle, twig beetles, pitch moths (families *Pyralidae*, and especially *Dioryctria* spp. and *Sesiidae*), black stain root disease, and pinyon dwarf mistletoe (Hagle et al. 2003; Rogers 1993; Shaw et al. 2005).

Insects

A forest inventory in Nevada reported insects to be the largest cause of tree mortality, followed by fire (Menlove et al. 2016). They can result in significant reductions in leaf area and seed crops (Mueller et al. 2005b; see section on seed production). However, insect-caused mortality is often linked to other factors causing stress in trees, especially drought, which can result in large areas of tree mortality (Shaw et al. 2005), and pinyon cone and seed crops can be reduced by more than 90 percent (Keen 1958; Mueller et al. 2005b) (see section on seed reproduction). Juniper seed production can also be

Table 2-3—Some of the more common insects, fungi, mistletoe, and mosses associated with pinyon and/or juniper (from Bunderson et al. 1986; Burns et al. 1990; Geils et al. 2002; Hreha and Weber 1979; Jacobi and Cranshaw 2014; Rogers 1993; Shaw et al. 2005).

Common name	Scientific name	Tree	Comments
Insects			
Ips or bark beetles	*Ips confusus*	Pinyon	Considered the most important mortality agent and closely linked to drought; larva tunnel in the bole of the tree feeding on phloem.
Twig beetles	*Pityophthorus* spp. *Pityogenes* spp.	Pinyon Pinyon	Innocuous on healthy trees but can cause top-kill and mortality of drought-stressed or diseased trees.
Mountain pine beetle	*Dendroctonus ponderosae*	Pinyon	Feed on weakened trees and often reach epidemic levels during drought
Pinyon cone beetles	*Conopthorus edulis*	Pinyon	Minor to significant impact on maturing (2nd-yr) cone crops.
Juniper bark beetle	*Phloeosinus serratus*		Endemic infestations often occur during drought; adults feed on twigs and larvae on cambium.
Bark & pitch moths	*Dioryctria ponderosae* *Petrova* spp. *Sesiidae* spp. *Retinia arizonensis* *Semiothisa* spp.	Pinyon Pinyon Pinyon Pinyon Pinyon	Moth larva mine the sapwood damaging limbs, terminal shoots, and developing cones but rarely kill mature trees; trees form pitch masses at the base of limbs.
Stem-boring moth	*Dioryctria albovittella*	Pinyon	Result in stem mortality and chronic infestations can reduce seed crops > 90 percent.
Cone moth	*Eucosma bobana*	Pinyon	Reduces cone and seed crop.
Pinyon spindlegall midge Pinyon gallmidges	*Pinyonia edulicola* *Pinyonia* spp. *Janetiella* spp. *Contarinia* spp.	Pinyon Pinyon Pinyon Pinyon	Affects developing needles forming a small gall at the needle base; outbreaks typically small and short duration and rarely causes serious damage.
Pinyon sawfly	*Neodiprion edulicolus*	Pinyon	Feeds on the pinyon needles and can cause some mortality among trees < 4 ft; can result in major defoliation in local areas.
Pinyon needle miner Pinyon needle scale	*Coleotechnites edulicola* *Matsucoccus acalyptus*	Pinyon	Nymphs suck fluids from needles causing premature death of foliage; severe damage to trees usually associated with drought.
Western cedar borer	*Trachykele blondeli* *Heinrichisesa*	Juniper Juniper	Larvae extensively mine the sapwood; heavy infestations linked to stressed trees.
Juniper twig pruner	*Styloxus bicolor*	Juniper	Beetle larva mine the pith, stunting growth but damage usually minimal
Gall midge	*Walshomyia* spp.	Juniper	Common on western and Utah junipers forming artichoke-like galls.
Fungi			
Black stain root disease	*Leptographium wageneri*	Pinyon	Introduced by bark beetles and spreads belowground by contact of diseased roots with adjacent uninfected trees resulting in mort centers; ranks high as a damaging agent.
Pinyon needle rust Pinyon blister rust	*Coleosporium jonesii* *Cronartium occidentale*	Pinyon	Both rusts alternate host plants are currants and gooseberries (Ribes sp.); common in the Great Basin; symptoms are several dead branches in the crown with bark lesions (blisters) at the base
Red-belt fungus	*Fomitopsis pinicola*	Pinyon	Found in dead trees, and rarely in live trees near large wounds
Heart rot	*Phellinus pini*	Pinyon	Stem decay usually in the middle and upper portions of the trunk
Heart-rot fungi	*Antrodia juniperina*	Juniper	

(Continued)

Table 2-3—(Continued).

Common name	Scientific name	Tree	Comments
White heart-rots	*Pyrofomes demidoffii Diplomitoporous rimosus and Phellinus texanus*	Juniper Juniper Juniper	These fungi decay the heartwood core and spreads by spore dispersal infecting wounds; advanced stages can cause mortality and impacts the structural integrity of the tree; common in older trees > 200 years
Stem rust	*Gymnosporangium* spp.	Juniper	Causes brooming of foliage
Mycorrhiza			
Ectomycorrhiza Endomycorrhiza		Pinyon Juniper	Also associated with juniper
Mistletoe			
Pinyon dwarf-mistletoe	*Arceuthobium divericatum*	Pinyon	Primarily southern CA, central and southern NV and UT; can cause considerable damage increasing vulnerability to stem disease and ips; frequency and extent of occurrence increases with trunk diameter and height
Juniper mistletoe	*Phoradendron juniperunum*	Juniper	Extends across the Great Basin and Colorado Plateau infecting Utah, Rocky Mountain, and western junipers
Dense mistletoe Constricted mistletoe	*Phoradendron ligatum Phoradendron densum*	Juniper	Both species of mistletoe are found throughout the Intermountain West extending from Oregon into Mexico; *P. desnum* most common on western juniper
Lichens & mosses			
Wolf lichens	*Letharia columbiana Letharia vulpina*	Juniper	Both species commonly grow together and most abundant on dead, barkless branches, and snags

significantly reduced (Furniss and Carolin 1977; Keen 1958; Morgan and Hedlin 1960; Powell 1963). The most common groups of insects associated with pinyon and juniper that can result in stress are beetles and moths (table 2-3).

Ips (bark beetle) is considered the most important insect mortality agent in the Colorado Plateau (Hagle et al. 2003; Rogers 1993; Shaw et al. 2005). Drought is frequently associated with bark beetle outbreaks in conifers (Anderegg et al. 2015; Bentz et al. 2010; Kolb et al. 2016). Both species of pinyon pine are attacked by a variety of insect species, but pinyon ips typically cause the highest levels of mortality (Skelly and Christopherson 2003). During the severe drought between 2002–2004, a large outbreak of pinyon ips occurred across this region, resulting in 3 million acres of damaged pinyon pine (Kleinman et al. 2012). While outbreaks of pinyon ips often occur during periods of drought (Furniss and Carolin 1977; Gaylord et al. 2013; Kleinman et al. 2012), they are also associated with other factors that can stress trees, including dwarf mistletoe, root disease, and soils with low water holding capacity—less than 10 percent, typical of sandy soils (McMillin et al. 2008; Negrón and Wilson 2003; Peterman et al. 2013; Shaw et al. 2005). Beetle populations and levels of mortality return to endemic levels when more normal precipitation amounts return (Kolb et al. 2016; Meddens et al. 2015).

Studies of pinyon ips outbreaks show that levels of mortality are linked to tree size, drought, increased winds, and pathogens such as black stain root disease and pinyon dwarf mistletoe (Hessburg et al. 1995; Meddens et al. 2015; Negrón and Wilson 2003; Wilson and Tkacz 1992). However, studies disagree on the importance of stand density, elevation, soil characteristics, and absence of a previous disturbance in supporting outbreaks (Meddens et al. 2015). These disagreements may be due to differences in how different scientists define drought severity or duration and whether the most affected trees have colonized marginal sites during periods of highly favorable growing conditions (Greenwood and Weisberg

2008; Meddens et al. 2015). Under exceptional drought conditions, as occurred in the early 2000s, the top-down climate influence may simply override the effects of bottom-up factors such as stand density, elevation, and soil characteristics.

Defining the climate factors conducive to pinyon ips outbreaks has been difficult due to inconsistent definitions of drought severity and duration—and often a lack of drought condition description in the various studies (Meddens et al. 2015). Pinyon ips can produce two to five generations per year, depending on climate (Eager 1999; Skelly and Christopherson 2003), but none of the descriptions found describe what thresholds govern how many generations may occur. Since temperature is a significant control on insect development and activity, there are likely fewer generations where overall temperatures are lower. As temperatures increase under climate change, pinyon ips may produce more generations per year and more adults will survive overwintering (Anderegg et al. 2015; Meddens et al. 2015; Negrón and Wilson 2003), potentially leading to greater impacts to singleleaf pinyon than have been documented thus far.

Gaylord et al. (2013) determined that 1 or more years of precipitation at only 55 percent of average favored successful pinyon ips attack and 3 years of precipitation at that level resulted in significant reductions of resin duct size and number and in 70 percent mortality of mature trees. Resin is an important insect inhibitor. Wilson and Tkacz (1992) found that the risk of twoneedle pinyon mortality from pinyon ips attack increased when April through July precipitation was less than or equal to 75 percent of average, based on a more localized outbreak in 1990 and 1991. Twoneedle pinyon's reliance on summer precipitation when temperatures are highest during the year may also result in greater vulnerability to pinyon ips attack than singleleaf pinyon.

Nonvascular Plants: Fungi

Although there are numerous species of fungi associated with pinyon and juniper (table 2-3), black stain root disease, which infects pinyon, is considered one of the most serious fungal-damaging diseases in the West. It has been related to extensive mortality in southwest Colorado and southeast Utah (Shaw et al. 2005). The primary vector of infection is through root contact between infected and uninfected trees (Skelly and Christopherson 2003), resulting in dead patches or mosaics of trees (Kearns and Jacobi 2005). It is also spread by root-feeding beetles. This fungal pathogen infects water-conducting root and stem tissue, preventing water movement to leaf-foliage (Hagle et al. 2003; Shaw et al. 2005). It kills young trees within 1–2 years (Skelly and Christopherson 2003). However, older trees decline more slowly, becoming increasingly susceptible to drought and insect infestation.

Pinyon blister rust, closely related to white pine blister rust, is also common across the Great Basin and Colorado Plateau. Although seldom resulting in mortality, the stressed trees are more susceptible to ips. Heart-rots are also widely spread throughout both pinyon and juniper across the Great Basin and Colorado Plateau. Heart-rots are primarily found in older trees, particularly in old-growth stands (fig. 2-24). Affecting primarily the heartwood in the trunk and large limbs, these rots rarely directly kill the tree, but can structurally weaken them. There was a widespread occurrence of heart rot in western juniper around 1730, attributed to possible climatic conditions (Knapp and Soulé 1999). Stem rust also has been reported to frequently occur in Utah juniper (Bunderson et al. 1986).

Nonvascular Plants: Mycorrhizal Fungi

Mycorrhizae are thought to increase the availability of water and nutrients to host plants, especially in nutrient-poor soils (Meyer 1973). However, the importance of these fungi can vary with soil drying patterns, soil pore structure, and number of hyphal connections extending from the root into the soil (Allen 2007). Mycorrhizae populations

Figure 2-24—The outer-most band of heart rot on this tree occurred between 1750 and 1730. This band of heart rot is widespread across the range of western juniper and is attributed to a climatic event or conditions. (Photo by Rick Miller, Oregon State University.)

are also influenced by drought, insects, mistletoe, and competition (Gehring et al. 2014; Sthultz et al. 2007; Swaty et al. 2004; Vecchio et al. 1993). Pinyons and junipers form mycorrhizal associations that, at least for pinyons, are positively linked to both seedling and adult plant growth (Gehring and Whitham 1994; Swaty et al. 2004). Pinyons are often the only associate of ectomycorrhizal (EM) in woodland communities, which do not penetrate the host's root cells but form an intercellular interface consisting of highly branched hyphae—long, branching filamentous structures of fungus (table 2-3).

Juniper are primarily associated with endomycorrhizal fungi (arbuscular mycorrhizal or AM), which penetrate the cortical cells of the roots of a vascular plant (Reinsvold and Reeves 1986). However, Utah juniper was reported to be infected with both AM and EM in northern Arizona (Klopatek and Klopatek 1987; Reinsvold and Reeves 1986). Although there are thousands of EM species, woodland communities in northern Arizona were dominated by one or a few types (Gehring et al. 1998). In juniper dominant or pinyon and juniper codominant woodlands, AM was significantly more abundant than EM (Haskins and Gehring 2005).

Nonvascular Plants: Lichens and Mosses

Two species of foliose lichens commonly associated with semiarid conifers are *Letharia columbiana* and *L. vulpine* (fig. 2-25a,b; Miller et al. 2005). 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 abundant on dead, barkless branches or snags. *Lethari columbiana* is widely spread throughout the Intermountain West, whereas *L. vulpina* is primarily limited to the Northwest and northern Rocky Mountains.
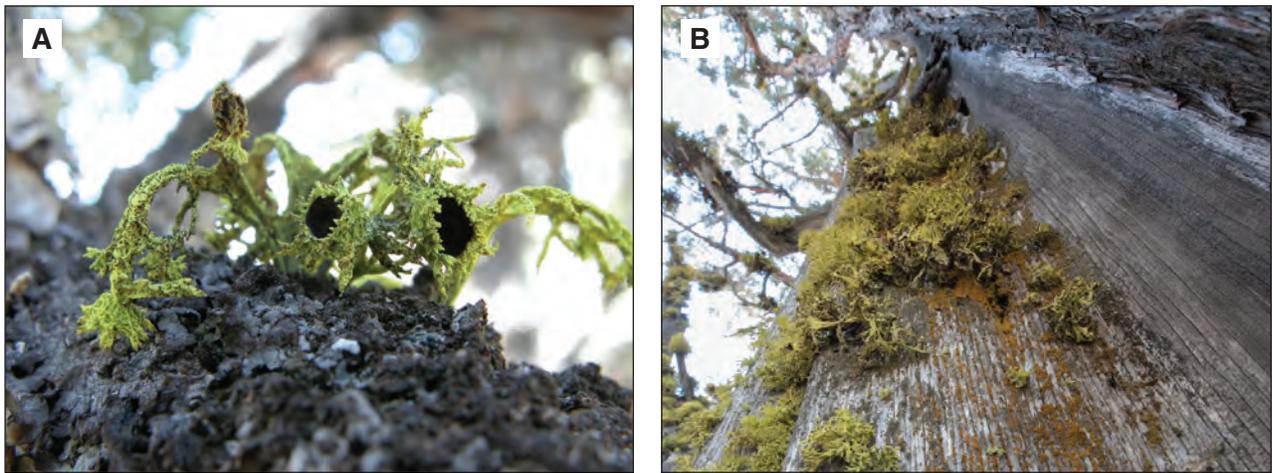
Figure 2-25—*Letharia columbiana* and *L. vulpina* are two species of foliose lichens commonly associated with semiarid conifers. (A) *L. columbiana* is widely spread throughout the Intermountain West and contains small disk-like fruiting bodies. (B) Both species of lichens are growing on the trunk of this tree. *L. vulpine* (on the lower left) primarily occurs in the northwest and lack the disk-like fruiting bodies. (Photos by Rick Miller, Oregon State University.)

Figure 2-26—Juniper mistletoe (shown here) and constricted mistletoe are the most common species infecting juniper in the Intermountain Region from Oregon southward into Mexico. Panamint Range, California. (Photo by Rick Miller, Oregon State University.)

Mistletoe

Several species of mistletoe commonly occur on juniper and pinyon across the West. Juniper mistletoe and constricted mistletoe are the most common species infecting juniper in the Intermountain region from Oregon south into Mexico (fig. 2-26). Their sticky seeds are primarily dispersed by birds, including American robins, Townsend's solitaires, cedar waxwings, flycatchers, and mountain bluebirds (Gill 1990; Sutton 1951). Seeds are also dispersed by wind and gravity, making dense stands more susceptible to infestation (Geils et al. 2002). Distribution among trees is often patchy with some heavily infested.

Pinyon dwarf mistletoe is restricted to pinyon species in the Intermountain regions south into Mexico and east into southwest Texas. It is considered a more serious conifer problem than leafy mistletoe in that it causes a higher incidence of mortality (Hreha and Weber 1979). In addition to increasing stress through high demands for water and nitrogen from its host, pinyon dwarf mistletoe also increases the potential for infestations by ips and stem diseases (Geils et al. 2002). Infestation was found to be directly related to increasing tree density and size (Hreha and Weber 1979). The mechanism for dwarf mistletoe seed dispersal, unlike the leafy mistletoe species, is primarily by its explosive discharge of seed (Geils et al. 2002). This frequently results in a clustered distribution of infected trees. However, birds and mammals are also important for long-distance dispersal.

Overstory/Understory Relationships

Succession

There is strong evidence supporting the inverse relationship between the tree overstory and understory vegetation, resulting in a successional gradient of changing woodland structure and composition (see Appendix B, sidebar 2) as trees become dominant. However, there is considerable variation in the rate of woodland development (fig. 2-27) and the relationship of its effects on understory vegetation (fig. 2-28; Johnson and Miller 2006; Roundy et al. 2014a). Variation in stand structure, composition, and rate of woodland development are largely determined by the key components of the ecological site (fig. 2-29), which vary considerably at multiple scales across the Great Basin and Colorado Plateau.

Early work described the successional stages following a stand-replacement fire in pinyon and juniper woodlands (Barney and Frischknecht 1974; Erdman 1970). Stages of succession included annual and/or perennial herbaceous vegetation transitioning to an herbaceous-shrub mix, followed by an increasing dominance of trees and decline in shrubs, grasses, and forbs (figs. 2-30, 2-31). More recent work supports this general sequence of succession following a stand-replacement event, although the potential for the persistence of invasive annuals creating an alternate steady state following a stand-replacement disturbance is of concern (Everett and Ward 1984; Miller and Heyerdahl 2008; Stringham et al. 2015a,b; Wangler and Minnich 1996).

Herbaceous Succession

The presence and persistence of an early annual herbaceous phase is dependent on the understory composition prior to the disturbance, soil moisture and temperature regimes, seasonal soil moisture and temperature, and the type and severity of disturbance, such as a low-, moderate-, or high-severity fire (Miller et al. 2013, 2014a; Roundy et al. 2018). In low- to moderate-severity fires, the postfire response of herbaceous cover and biomass are closely related to prefire plant composition (fig. 2-32; Allen et al. 2008b; Bates et al. 2014a; Dhaemers 2006; Koniak and Everett 1982; Miller et al. 2013). This is also usually true with other disturbances, resulting in mortality of the tree overstory with little to no impact on the understory such as mechanical tree reduction (Everett and Sharrow 1985a; Miller et al. 2014b; Roundy et al. 2014a). However, high-severity fires, which are characteristic of wildfires in late Phase II and III woodlands, usually result in more than 85 percent mortality of perennial grasses (Bates et al. 2011) and can consume 85–98 percent of the seed bank (fig. 2-33; Beckstead et al. 2011). These high-severity events can promote the dominance of invasive annuals, which may shift the plant community to an annual steady state (Appendix C; Bates et al. 2014a; Stringham et al. 2015a,b; Tausch 1999).

Sidebar 2

Space Versus Time

Numerous studies evaluate woodland succession in the Great Basin and Colorado Plateau. However, our understanding of succession comes largely from studies substituting space for time, where different locations (space) with different time-since-disturbance are compared. When space is substituted for time, it is critical to consider all of the key components (fig. 2-29) as to how they compare across the different locations. Only a few studies have measured vegetation change over time at the same location and are largely short-term (less than 10 years) (Miller et al. 2013), with few extending 20 to 42 years (Bates et al. 2017b; Bristow et al. 2014; Schaefer et al. 2003), and none we could find extending past 50 years.

Figure 2-27—Hypothesized amount of time required from initial western juniper establishment (early Phase I) to a minimum stocking level for stand closure (early Phase III), and estimated maximum potential for tree density and cover as related to site productivity (from Johnson and Miller 2006). Projected rates of closure are similar for pinyon pine and Utah juniper (from Tausch et al. 2009).
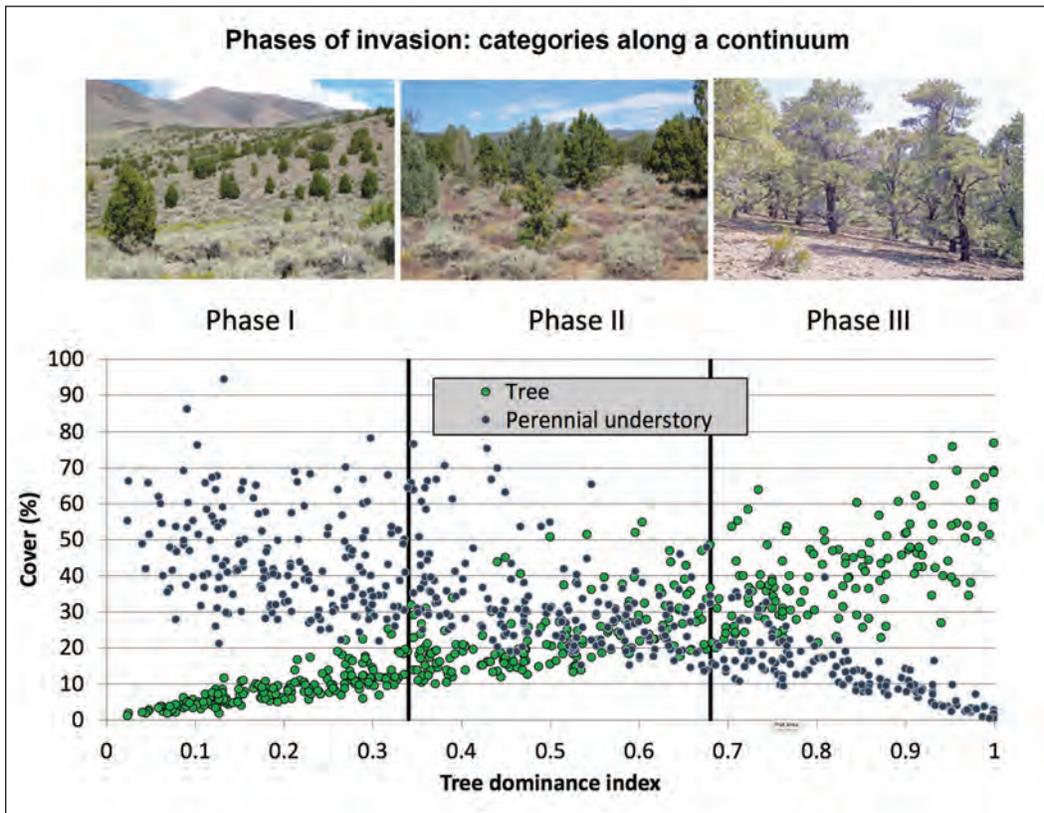
Figure 2-28—Changes in perennial understory cover and tree dominance (tree cover/ [tree+shrub+tall perennial grass cover]) for 11 Great Basin sagebrush steppe sites ranging from Phase I to Phase III pinyon-juniper (see Glossary for full definitions of Phase and TDI). The wide variation is largely a result of ecological site attributes (Roundy et al. 2014a).
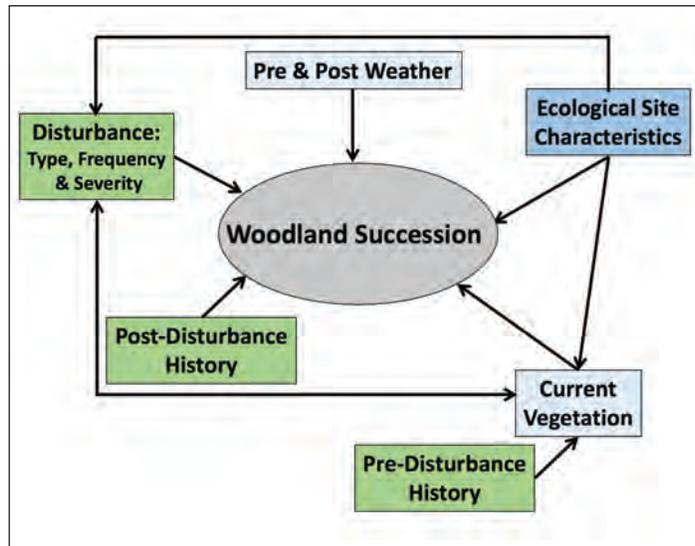
Figure 2-29—A simple conceptual model of the key components affecting postdisturbance woodland succession. Ecological site characteristics relate to climate, topography, and soils. Weather is the atmospheric condition occurring during the time-period of interest.
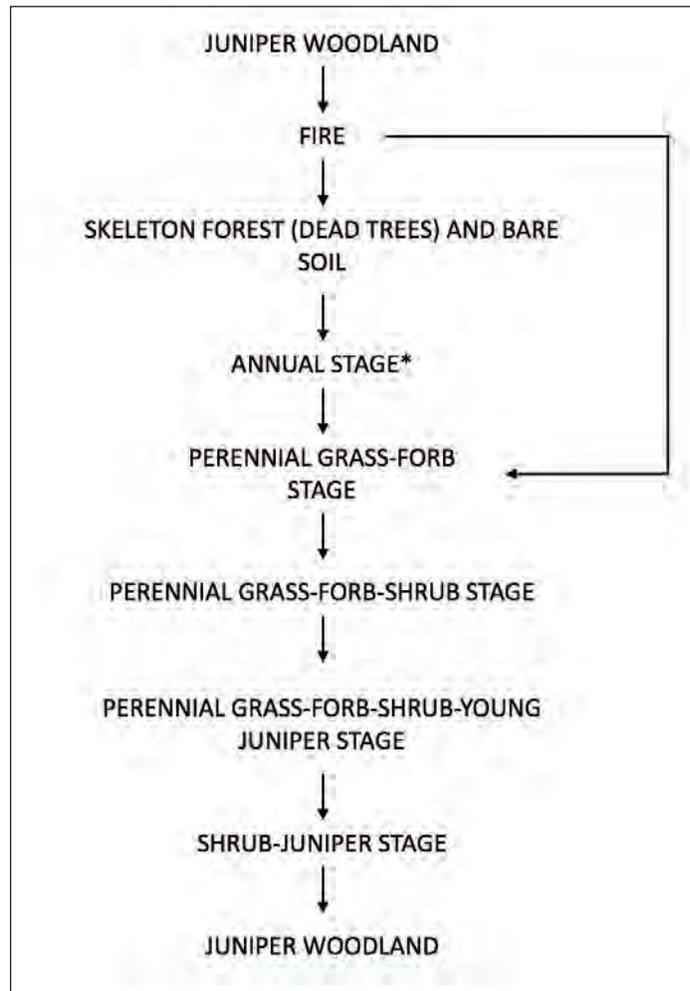
Figure 2-30—Suggested successional stages after fire in pinyon and juniper woodlands. Plant annual stage during early succession may be bypassed if a perennial understory is present prior to and persists following fire. (From Erdman 1970; Barney and Frischknecht 1974.)
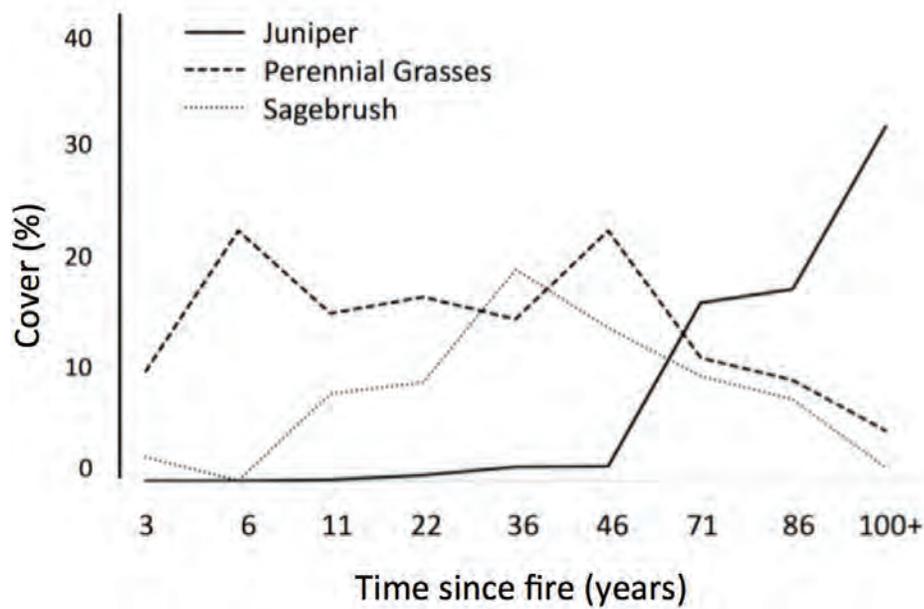
Figure 2-31—Surface cover of juniper, perennial grasses, and sagebrush by approximate time since fire (derived from Barney and Frishknecht 1974).

Figure 2-32—Following a moderate severity fire (note browned conifer needles remaining on trees), herbaceous response in the first growing season was primarily from residual perennial grasses (present prior to the fire). Sagebrush mortality was high—a result of both the fire and a prior infestation of Aroga moth in the area. Central Oregon. (Photo by Rick Miller, Oregon State University.)

Figure 2-33—Following a high severity fire, herbaceous response in the first growing season was primarily native annual forbs. Although the high severity fire could have resulted in high mortality of native perennial grasses, the lack of burned grass crowns near the soil surface indicates understory vegetation was likely severely depleted prior to the fire. The resistance of this site to invasive annual grasses is very low (lack of perennial herbs and warm and dry soils), resulting in the potential of large increases of invasives in the second and third year following fire. Central Utah. (Photo by Rick Miller, Oregon State University.)

Sequences of plant composition during succession after a stand-replacement event follows a general pattern (figs. 2-30, 2-31), but duration of each sequence can vary widely. If there is a native annual forb response, it is typically short-lived, persisting less than 5 years postfire (Barney and Frischknecht 1974; Bates et al. 2017a,b; Dhaemers 2006; Everett and Ward 1984; Koniak 1985). Perennial forb abundance is also inversely influenced by tree canopy dominance (Roundy et al. 2014a) and usually increases following a stand-replacement event reaching a peak within the first 2–5 years (Bates et al. 2017b; Everett and Ward 1984; Miller et al. 2014b) then declining to preburn levels (Bates et al. 2017b). Perennial grasses commonly peak in abundance within the first 10 years, followed by a decline and leveling off during the next 10 to 45 years, then declining with increasing tree dominance to less than 5 percent cover (fig. 2-34; Bates et al. 2017b; Miller and Heyerdahl 2008). However, perennial grasses may persist at relatively high levels of abundance in the late stages of woodland development on deep loamy soils and where grasses have not been overgrazed (fig. 2-35; Miller et al. 2005).

Shrub Succession

There is a very strong inverse relationship between the abundance of pinyon and juniper and with understory shrubs (fig. 2-36; Barney and Frischknecht 1974; Bybee et al. 2016; McHugh and Gehring 2006; Roundy et al. 2014a; Williams et. al 2017), although it varies across different ecological sites, plant associations, and shrub species (figs. 2-34, 2-35, 2-37; Miller et al. 2000). Sagebrush appears to be one of the most sensitive shrubs to increasing tree dominance, declining at more rapid rates than bitterbrush. Following a stand-replacement event, the shrub phase typically increases and peaks 30 to 50 years, followed by a decline with increasing tree dominance (fig. 2-31;

Figure 2-34—This closed Phase III pinyon and juniper woodland is approaching full dominance by the tree overstory. The lack of deep-rooted perennial grasses is the result of either tree competition, past poor grazing management, or both. Shrubs are also nearly absent, except for in the few openings, such as shown in the foreground, and sapling trees have limited annual growth. Grand Staircase, Utah. (Photo by Rick Miller, Oregon State University.)

Figure 2-35—Shrub skeletons are the only evidence that shrubs once occurred before trees dominated the site. However, the moderately deep to deep soils allow deep-rooted perennial grasses to persist under proper management in this Phase III pinyon and juniper woodland. Southern Utah. (Photo by Rick Miller, Oregon State University.)
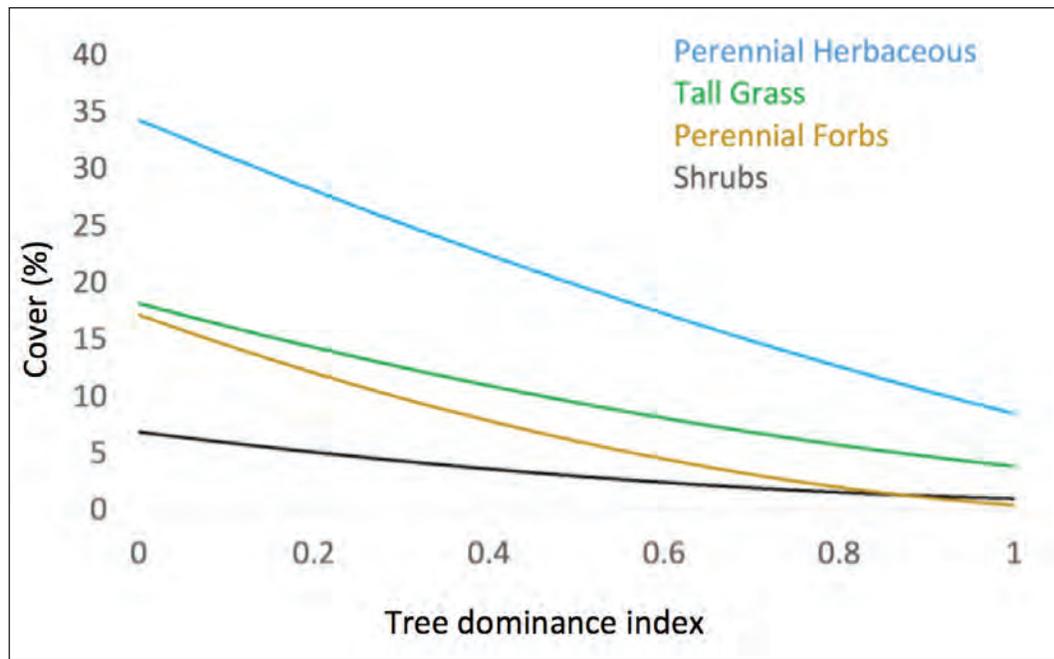
Figure 2-36—Understory vegetation cover in relation to increasing tree dominance index (TDI; see Glossary for full definition) (derived from Roundy et al. 2014a).
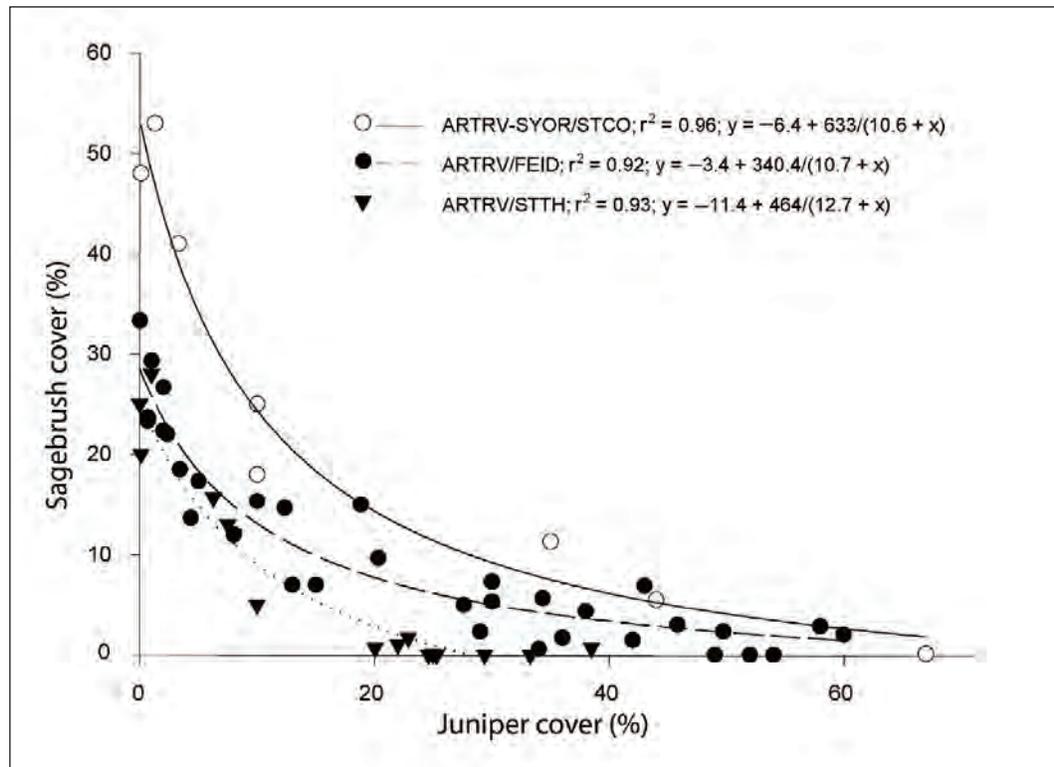
Figure 2-37—Relationship between juniper and sagebrush cover for the mountain big sagebrush/ Thurber needlegrass (ARTRV/STTH), mountain big sagebrush/ Idaho fescue (ARTRV/FEID), and mountain big sagebrush-snowberry/Columbia needlegrass (ARTRV-SYOR/STCO) associations. The more productive sites (Columbia needlegrass > Idaho fescue > Thurber needlegrass) show a higher level of juniper cover in relation to existing sagebrush cover (Miller et al. 2000).

Barney and Frishknecht 1972; Bybee et al. 2016; Miller et al. 2000; Roundy et al. 2014a; Schaefer et al. 2003; Tress and Klopatek 1987; Wangler and Minnich 1996; 2005).

However, the recovery of the shrub layer is dependent on seed availability for nonsprouting shrubs (e.g. sagebrush and bitterbrush, which is a weak resprouter) and the prefire density of sprouting shrubs—including green and grey rabbitbrush, horsebrush, and snakeweed. Mountain big sagebrush canopy cover usually approaches 20 to 30 percent within 20 to 35 years postfire—but can take up to 50 years or less than 20 years in small patchy burns (Bunting et al. 1987; Lesica et al. 2007; Miller and Heyerdahl 2008; Moffet et al. 2015; Nelson et al. 2014; Ziegenhagen 2003; Ziegenhagen and Miller 2009). The ability of bitterbrush to resprout is highly variable, and this plant is typically a weak resprouter in more arid environments (Miller et al. 2013).

Postfire reestablishment of bitterbrush in mountain big sagebrush communities followed a similar pattern to mountain big sagebrush and came largely from seed (Ziegenhagen and Miller 2009). Recovery of Wyoming and black sagebrush, which occur on drier and/or warmer sites and low sagebrush occurring on shallow to very shallow often heavy clay soils, are often considerably slower than mountain big sagebrush (Hosten and West 1994; Rhodes et al. 2010). Sprouting shrubs such as snakeweed and green rabbitbrush will reestablish and peak 10 to 12 years postfire—often followed by a decline (Harniss and Murray 1973; Rhodes et al. 2010; Wangler and Minnich 1996; Young and Evans 1974). However, these relatively shorter-lived species compared to sagebrush can persist at relatively high levels for up to 40 years (Barney and Frischknecht 1974) under regimes of frequent disturbance (Young and Evans 1974).

Tree Succession

Many studies have reported low tree dominance 45 years after a stand-replacement disturbance, followed by a rapid expansion in tree canopy cover (fig. 2-31; Barney and Frischknecht 1974; Miller et al. 2005; Miller and Tausch 2001; Tausch and West 1988; Tress and Klopatek 1987; Wangler and Minnich 1996). Timing and rate of tree-seedling establishment following a stand-replacement disturbance is a primary factor determining the rate of tree canopy development. Postdisturbance tree establishment is dependent upon seed source, dependent upon resource availability, and enhanced by the presence of a shrub canopy, which provides a more suitable environment for young trees—see nurse plants in seedling establishment processes section (Everett et al. 1986a; Johnsen 1962; Miller and Rose 1995; Wangler and Minnich 1996).

Once established, the growth rate of tree seedlings is relatively slow (Miller et al. 2005, 2007; Tausch et al. 2009). However, once trees reach 40 to 50 years of age, the rate of canopy expansion typically increases, although the rate of growth varies with site productivity. The presence of mature trees surviving a stand-replacement event can also more rapidly initiate the development of woodlands compared to stands lacking trees (Johnson and Miller 2008). During the late stages of tree canopy development, establishment and growth of understory trees is suppressed by the mature overstory trees (Miller et al. 2000). In addition, intraspecific competition among trees in late woodland development results in the reduction of tree branch leader growth and cone production (fig. 2-38).

Where ecological site conditions are suitable for both juniper and pinyon species, reestablishment of juniper frequently precedes pinyon after a stand-replacement event, followed by an increasing rate of pinyon establishment that often eventually exceeds that of juniper (Huffman et al. 2012; Tausch and West 1988; Wangler and Minnich 1996; Woodbury 1947). In late Phase II and III woodlands, the relative canopy cover of understory pinyon trees compared to juniper is often greater than in the overstory canopy (Abella et al. 2012). Changes in vegetation structure during succession modify microsites

Figure 2-38—Leader growth on these tree saplings indicate that: (1) this ecological site is relatively productive; (2) there is little intraspecific competition among trees; and (3) the shrub layer has little effect on tree growth. Although leader growth is shorter on less productive sites, the length of leaders on younger trees becomes severely reduced under intraspecific competition from the tree overstory, indicating a near closed stand of trees. Grand Staircase, Utah. (Photo by Rick Miller, Oregon State University.)

for seedling establishment and may be more beneficial for pinyon over time. Pinyon is more sensitive to dry soil conditions then juniper, shutting down photosynthesis and resulting in a disadvantage under droughty conditions (Barnes and Cunningham 1987; Breshears et al. 1997a).

There is likely significant competition for limited soil resources between pinyon pine and juniper (Haskins and Gehring 2004). Reduction of juniper roots results in a near twofold increase of pinyon roots and ectomycorrhizal associated with pinyon. Also, the establishment of juniper in Arizona was reported to be related to time since fire, while pinyon appeared to be more episodic, related to climate conditions (Huffman et al. 2012). But in Mesa Verde, Colorado, differences in initial establishment following a stand-replacing fire between juniper and pinyon were not clear (Erdman 1970).

Expansion of pinyon and juniper into meadows, riparian areas, and aspen has also been reported in the Intermountain West. Singleleaf pinyon and Utah juniper expanded into riparian and meadow communities following down-cutting of deep meadow soils in southwest Utah (Cottam and Stewart 1940) and western juniper into aspen communities across the northwest Great Basin (Wall et al. 2001). These conversions were attributed to improper grazing and/or browsing, and lack of fire.

Succession to Old-Growth

Old-growth pinyon and juniper woodlands are the late stages of succession with distinct characteristics that evolve over centuries. Intervals between stand-replacement events in these old-growth woodlands are hundreds of years (sidebar 3), allowing trees to attain ages of many centuries and woodlands to develop old-growth characteristics. Old growth is typically defined at the community level and encompasses the later stages of woodland or forest development that typically differs from earlier stages in a variety of characteristics, which include tree age and size, accumulations of large dead woody material, number of canopy layers, species composition, and ecosystem function (Kaufmann et al. 1992; USDA Forest Service 1993). Characteristics of old-growth pinyon and juniper woodlands vary widely across the Great Basin and Colorado Plateau (Kaufmann et al. 1992; Miller et al. 1999). The majority of old-growth woodlands are composed of old trees (more than 250 years) and the accumulation of large dead wood comprising snags, logs, and weathered stumps (table 2-4; Floyd et al. 2003; Miller et al. 1999, 2007, 2014a; Tausch et al. 2009; Waichler et al. 2001).

Characteristics of individual old pinyon and juniper trees compared to younger mature trees include differences in tree canopy shape, distinct bark characteristics including strips of shaggy bark and distinct vertical furrows, rot pockets, cavities, and branch structure, which may include large basal branches, and dead branches often covered with lichen (fig. 2-39a,b,c). These morphological characteristics commonly begin to develop between 150 and 200 years and become more developed over time (Miller et al. 1999). Utah and western juniper can exceed ages of 1,500 years, and pinyon over 900 years (Miller et al. 2005, 1999; Strachan and Biondi 2013; Swetnam and Brown 1992; Tausch et al. 1981). The oldest of the old pinyon and juniper are usually found on the harshest sites where there is little surface fuel and growth is slow, resulting in changes in wood structure and chemistry (Farjon 2010; Swetnam and Brown 1992; Christopher Baisan, Retired Dendrochronologist, Dendrochronology Laboratory, University of Arizona, Tucson, Arizona, personal communication, 1991). In the Southwest, pinyon and juniper woodlands older than 400 years are comparatively rare, possibly a result of severe drought in the late 1500s causing high tree mortality (Swetnam and Brown 1992). Following the drought, there was a substantial increase in tree establishment, beginning in the early 1600s that peaked around 1700.

As described in the soils section in more detail, old-growth woodlands are often associated with rock outcrops, knolls, ridges, and/or soils that are shallow, coarse, rocky, and often high in clay or sand (fig. 2-40; Barney and Frischknecht 1974; Bauer and Weisberg 2009; 1974; Cottam and Stewart 1940; Emerson 1932; Holmes et al. 1986; Miller and Rose 1999; Nicol 1937; Stringham et al. 2015; Woodbury 1947). But there is considerable variation in ecological site characteristics, which contributes to the wide variation in woodland structure and species composition on sites where persistent woodlands can occur. Old-growth pinyon and juniper woodlands can be grouped into three broad community types, which include woodland, juniper-low sagebrush tableland, and pinyon and juniper savanna (Miller et al. 1999).

Sidebar 3
Old-Growth, Persistent, and Presettlement Woodlands

In the literature, the terms presettlement, persistent, and old-growth are often used for woodlands existing on the landscape prior to Eurasian settlement. Although often used interchangeably, each has a different meaning and may or may not be one and the same. Presettlement woodlands established prior to Eurasian settlement and introduction of livestock around the 1860s may or may not exhibit old-growth characteristics. Persistent woodlands (synonymous with potential vegetation) are based on ecological site characteristics and disturbance regimes that allow woodlands to develop into a late successional stage. Old-growth is based on relative tree age and woodland structural characteristics. See Glossary for complete definitions.

Scattered old trees can also occur in shrub or grass dominated communities, often associated with microtopography (fig. 2-41). Established woodlands typically have more than 20 percent tree canopy cover with little to substantial perennial herbaceous understory and limited to no shrub layer. The abundance of the perennial herbaceous layer is largely attributed to soil characteristics, especially soil depth, disturbance regimes, and tree density (fig. 2-42; Miller et al. 2005; Swetnam and Brown 1992). Juniper-low sagebrush tableland occurs on very shallow soils over fractured basalt or claypans (most common in the western juniper region). It usually has less than 10 percent tree canopy cover and an understory of low sagebrush and Sandberg bluegrass (fig. 2-43). Pinyon and juniper savanna, most common in the Colorado Plateau, typically has less than 10 percent canopy cover with a predominant perennial grass understory (fig. 2-44).

In all three old-growth types, there has been significant infill throughout the Great Basin and Colorado Plateau, which started around 1900 and is attributed to (or a combination of) climate, grazing, and altered fire regimes (see Section 3). In addition to these three general types, scattered patches of old trees commonly occur on rock outcrops, knolls, and ridges across the landscape. Observations of Arizona savannas and grasslands in the early 1900s reported old pinyon and juniper seed trees occupying the rocky knolls and ridges adjacent to the grasslands (Leopold 1924; Nicol 1937).

Table 2-4—Characteristics that differentiate young and old-growth woodlands. There are several types of woodlands based on stand age in the Great Basin and Columbia River Plateau region. These include: (1) old-growth woodland; (2) woodlands that were formerly old growth, but are currently occupied by young trees (less than 150 years old) as a result of a stand replacing disturbance; (3) tree shrub savanna where the old trees are less than 10 percent canopy cover; (4) tree shrub savanna that is infilled by post-settlement young trees; and (5) sagebrush shrub-steppe occupied by young trees (less than 150 years) (modified from Miller et al. 2007 and Tausch et al. 2009).

Woodland characteristics and tree growth form		
Characteristic	**Postsettlement trees**	**Presettlement trees**
Juniper crown shape	Conical with point tip	Flattened, rounded, or uneven tops
Piñon crown shape	Conical with pointed to slightly rounded tip	Flattened, rounded, or uneven top
Juniper branch structure	Branches get progressively smaller from bottom to top of tree	In open stands, large branches near the base
Piñon branch structure	Branches become smaller from bottom to top of tree, general orientation is vertical	In open stands branches large near base and remain relatively large well into the crown, more randomly oriented
Juniper bark	Flaky, relatively thin with limited or shallow vertical furrows	Thick, fibrous with well-developed vertical furrows
Piñon bark	Relatively thin, flaky, with weak vertical furrows	Thicker, more plate-like structure than furrowed
Juniper leader growth	Terminal leader growth in the upper 1/4 of the tree usually > 2 in. In open stands, leader growth > 2 in from bottom to top	Leader growth in the upper 1/4 of the tree usually < 1 in
Piñon leader growth	Leader growth in pinyon similar to juniper but not directly visible; must look for bud scale scares to determine length	Leader growth in upper 1/4 of the tree usually < 2 in.
Tree canopy lichen	Little or no foliose lichen on juniper	Juniper often covered by bright green foliose lichen
Dead wood in standing tree	Little dead wood in bole, few to no dead trees, logs, or large stumps	Dead branches, bark missing, black stain and/or black lichen
Large wood across the site	Large diameter logs and stumps absent	Large diameter logs and stumps, often charred, scattered across the site
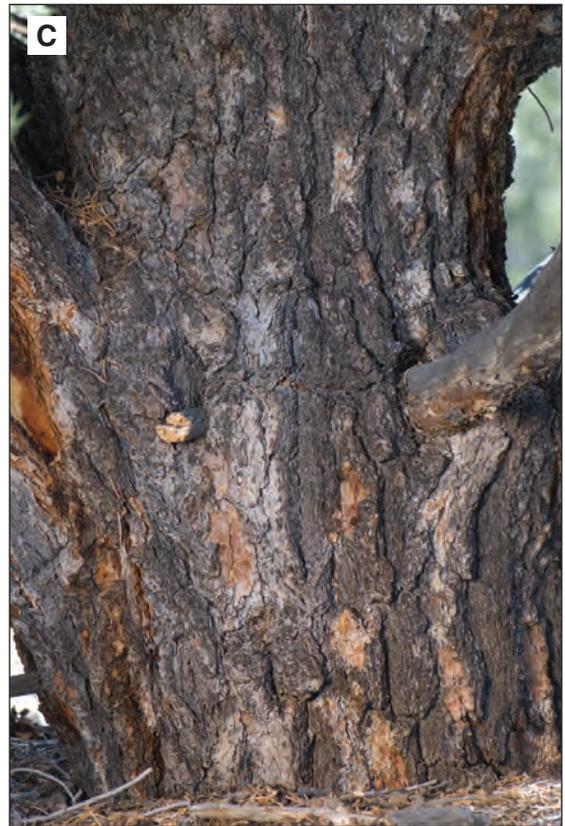
Figure 2-39—Pinyon and juniper trees begin developing old-growth characteristics as they approach 200 years of age, features that become more prominent as the tree becomes multiple centuries old. Characteristics include tree canopy shape, distinct bark characteristics, rot pockets, cavities, and branch structures—which may include large basal branches and dead or partially dead branches often covered with lichen. The stringy and deeply furrowed bark on the (A) western juniper and (B) Utah juniper are typical of trees over 300 years old. (C) The well-developed bark characteristics on this singleleaf pinyon is typical of trees over 200 years old becoming more developed with age. Note the standing dead tree in (A) has been dead for several centuries. Horse Ridge, Oregon and Schell Creek Mountains, Nevada. (Photos by Rick Miller, Oregon State University.)

Figure 2-40—The oldest trees are most commonly found on poor soils, where growth is slow and there are little to no surface fuels to carry fire. Some trees on these sites can exceed 1,000 years of age. Central Nevada. (Photo by Rick Miller, Oregon State University.)

Figure 2-41—The oldest known western juniper, near 1,650 years, occupies a protected micro-site surrounded by rocks and limited surface fuels. Horse Ridge, central Oregon. (Photo by Rick Miller, Oregon State University.)

Figure 2-42—Old-growth woodlands typically have more than 20 percent tree canopy cover with limited to substantial perennial herbaceous understory and limited to no shrub layer. The abundance of the perennial herbaceous layer is largely attributed to soil characteristics, especially soil depth, disturbance regimes, and tree density. Soils in this Phase III old-growth woodland are skeletal (more than 30 percent rock) moderately deep to deep sandy loams. Horse Ridge, central Oregon. (Photo by Rick Miller, Oregon State University.)

Figure 2-43—Scattered old-growth trees also grow in low sagebrush communities with shallow to very shallow soils but fractured bedrock. Tree canopies are usually less than 10 percent, and surface fuels are limited resulting in infrequent low intensity fires. Modoc Plateau, northern California. (Photo by Rick Miller, Oregon State University.)

Figure 2-44—Pinyon-juniper savannas most commonly occur in the Colorado Plateau. They typically have less than 10 percent canopy cover with a predominant perennial grass understory. Increases in low palatability shrubs—such as snakeweed—are indicators of drought and overgrazing. Grand Staircase, southern Utah. (Photo by Rick Miller, Oregon State University.)

Diversity and Richness

Floristic richness of pinyon and juniper woodlands has been reported to range from moderate (Christie 2009) to relatively low (Tueller et al. 1979; West et al. 1978b). Across 66 mountain ranges in the Great Basin, a total of 240 perennial and 127 annual species were recorded within the woodlands (West et al. 1978b). The researchers concluded this to be relatively low, considering the large area that was measured (463 woodland plots). At smaller scales (ranging from individual stands to watersheds), the variation in floristic diversity and richness can be very high, varying with topography, soils, disturbance regimes, and tree dominance (figs. 2-45, 2-46). Potential floristic diversity across woodlands in Arizona, New Mexico, and eastern Oregon was closely linked to the amount of available soil moisture, number of different soil types, and variation in solar radiation across the woodland or area measured (Harner and Harper 1976; Miller et al. 2000). There is also an inverse relationship between floristic richness and diversity with increasing tree dominance (Bates et al. 2000; Everett and Koniak 1981; Miller et al. 2000; Tausch et al. 1981). Plant diversity was 1.6 times greater on recently cut postsettlement woodlands compared to adjacent Phase III uncut woodlands (Bates et al. 2000). Seedbank diversity may (Koniak and Everett 1982) or may not (Allen and Nowak 2008) change with increasing tree dominance.

Competition

There is substantial evidence that there is a negative relationship between tree canopies and understory vegetation (figs. 2-28, 2-36, 2-37; Bates et al. 2000; Everett and Sharrow 1985a; Miller et al. 2000, 2014a; Roundy et al. 2014a). As trees increase in dominance, acquisition of soil resources increases and the microenvironment beneath the tree canopy changes.

Soil Water Competition

As pinyon and juniper increase on a site, a larger proportion of available soil water is used by the trees, resulting in significant declines in understory vegetation and length of the growing season. Rapid use of soil water by the trees occurs primarily during the growing season (Angell and Miller 1994; Bates et al. 2000; Emerson 1932; Miller and Shultz 1987; Roundy et al. 2014b). However, if soils are not frozen during the winter, western juniper was reported to draw down winter soil moisture, resulting in drier soils at the beginning of spring compared to adjacent sites with no trees (Jeppesen 1978). In areas where trees have been removed, the growing season of posttreatment herbaceous vegetation was increased by 3 to 6 weeks (fig. 2-47; Bates et al. 2000; Roundy et al. 2014b). During the spring, Utah and western junipers rapidly used soil water in the upper soil zone (less than 3 feet) where soil nutrient concentrations are highest and most important for the majority of herbaceous species (Angell and Miller 1994; Bates et al. 2000; Evans and Ehleringer 1994; Flanagan et al. 1992; Leffler et al. 2002; Ryel et al. 2010).

As upper soil layers become dry, trees will use deep water resources. On a site in southeast Oregon where the majority of western juniper had been removed, predawn water potentials in scattered sapling trees remained above -1.5 MPa (standard wilting point is less than -1.5) in the summer when upper soil layers were dry, indicating acquisition of available water from deeper water resources (fig. 2-47; Bates et al. 2000). Utah juniper was found to always extract water deeper in the soil profile than big sagebrush (Leffler and Caldwell 2005). Utah juniper was also able to take advantage of relatively small summer rainfall pulses of 0.39 inches (10 mm) that moistened dry surface soils (Evans and Ehleringer 1994; Flanagan et al. 1992; Leffler et al. 2002; Ryel et al. 2010). In comparison, twoneedle pinyon required greater amounts of rainfall (25 mm, 0.98 inches) to increase physiological activity (West et al. 2007a,b).

Figure 2-45—The literature reports a large variation in floristic diversity and richness in pinyon and juniper woodlands, ranging from low to moderate. The level of diversity and richness varies with topography, soils, disturbance regimes, and tree dominance. Floristic diversity and richness in this early Phase II woodland, encroaching into a mountain big sagebrush community with 14 to 16 inches of precipitation and moderately deep soils, is relatively high, but will decline with increasing tree dominance. Schell Creek Range, eastern Nevada. (Photo by Rick Miller, Oregon State University.)

Figure 2-46—Species diversity is very low in this closed pinyon and juniper woodland on shallow to moderately deep soils. There is also little recruitment of trees, and saplings have very limited leader growth, resulting from high intraspecific competition. Colorado Plateau, southern Utah. (Photo by Rick Miller, Oregon State University.)
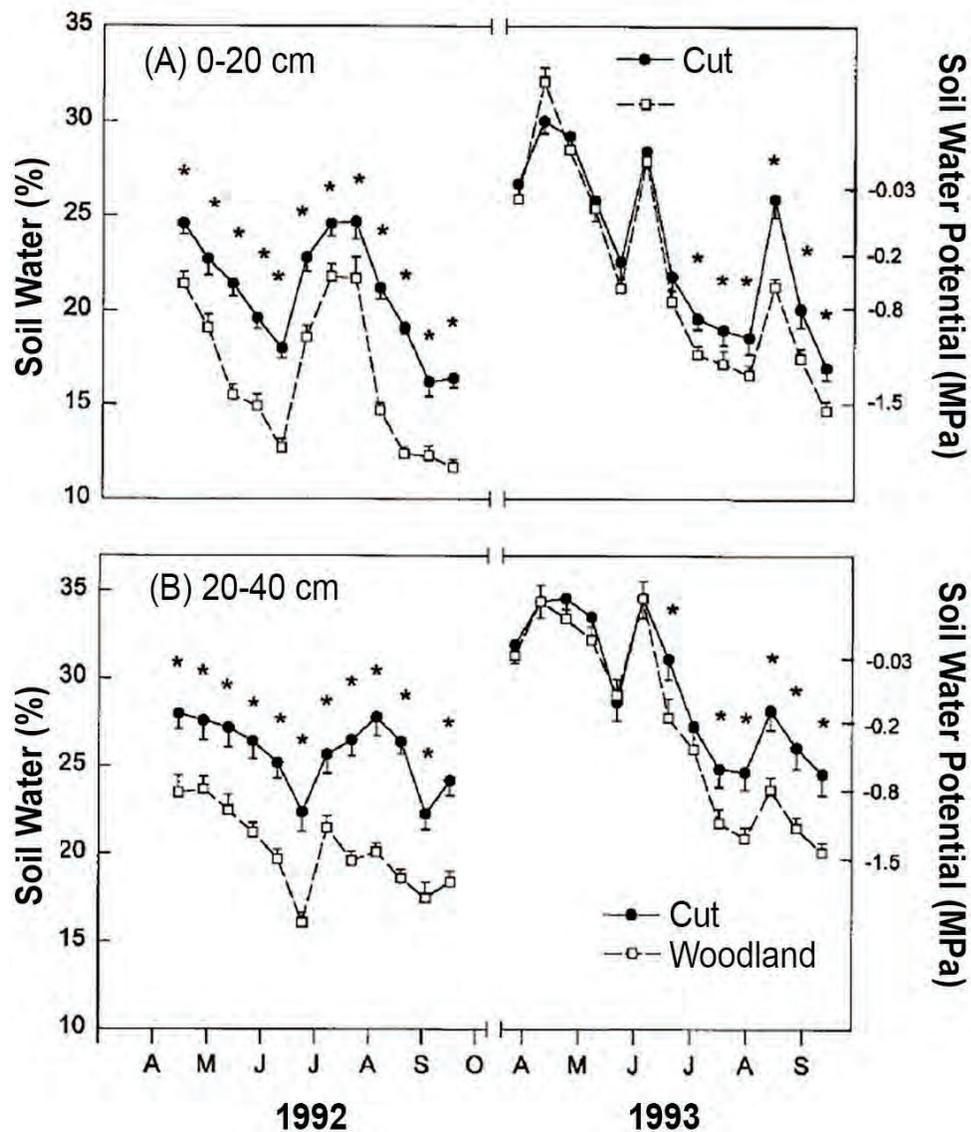
Figure 2-47—Gravimetric soil water content (percent soil water by weight) and soil water potential (less than -1.5 MPa = wilting point) in interspace soils from: (A) 0–20 cm and (B) 20–40 cm soil depth in cut and uncut Phase III woodlands. Soil water potentials were determined by predawn leaf water potential of sapling juniper trees left in the cut and uncut stands. Asterisks (*) indicate significance differences on that sampling date ($P < 0.05$) (Bates et al. 2000).

Precipitation Interception

Effective soil moisture is also reduced by pinyon and juniper through precipitation interception within the tree canopies where it is evaporated back to the atmosphere (Eddleman 1984; Skau 1964; West et al. 2007a; Young et al. 1984). Tree canopies also capture snow that is lost to the atmosphere through sublimation (Larsen 1993). The amount of moisture lost through canopy interception varies with the amount of tree canopy cover and proportion of the canopy made up of pine versus juniper, and the duration, intensity, and type of precipitation. Skau (1964) did some of the first work on the amount of rainfall interception by juniper, reporting precipitation interception by Utah juniper ranged from 7–25 percent, which varied with tree canopy density. Interception by several juniper species in the Intermountain Region ranged from 50–60 percent for low intensity and 5–35 percent

for high intensity storms (Pierson and Williams 2016). In western juniper, up to 74 percent of the precipitation was intercepted in the tree canopy area, and stem flow was absent or very limited (Eddleman 1984; Larsen 1993; Young et al. 1984). This is similar to results in a twoneedle pinyon and oneseed juniper woodland in northwestern New Mexico where snow water equivalent was 80 percent greater in openings between tree canopies compared to directly beneath the tree canopies (Breshears et al. 1997b). In addition to interception by the tree canopy, the litter-duff layer beneath the tree canopy can reduce infiltration, further reducing effective precipitation (Owens et al. 2006). For additional information see subsection "Interception" in Section 4, Ecohydrology.

Allelopathic Competition

Limited literature suggests juniper litter can reduce plant growth directly beneath the tree canopy through the release of allelopathic compounds (Jameson 1961, 1966). However, the possible production of chemical inhibitors by juniper species and what we observe in the field (fig. 2-48a,b) makes this issue unclear. In the Great Basin perennial bunchgrasses, especially Idaho fescue and bluebunch wheatgrass, often are well established beneath the tree canopy. However, seedling emergence was reported to be reduced with increasing juniper litter depth, and emergence was slower in juniper litter compared to sites with no litter (Horman and Anderson 2003). But, there was no difference 2 weeks postemergence in plant growth between plants growing in juniper litter and no litter. Beneath recently cut and downed western juniper canopies, herbaceous plants quickly establish in litter zones following tree cutting (Bates et al. 1998). In addition, seedlings and established plants grew larger than species in the adjacent interspaces, particularly annual grasses and forbs. The degree of impact of juniper litter on understory vegetation may also depend on the species growing in the duff zone (Gehring and Bragg 1992; Vaitkus and Eddleman 1987) and on litter density and depth, which can act as a physical barrier.
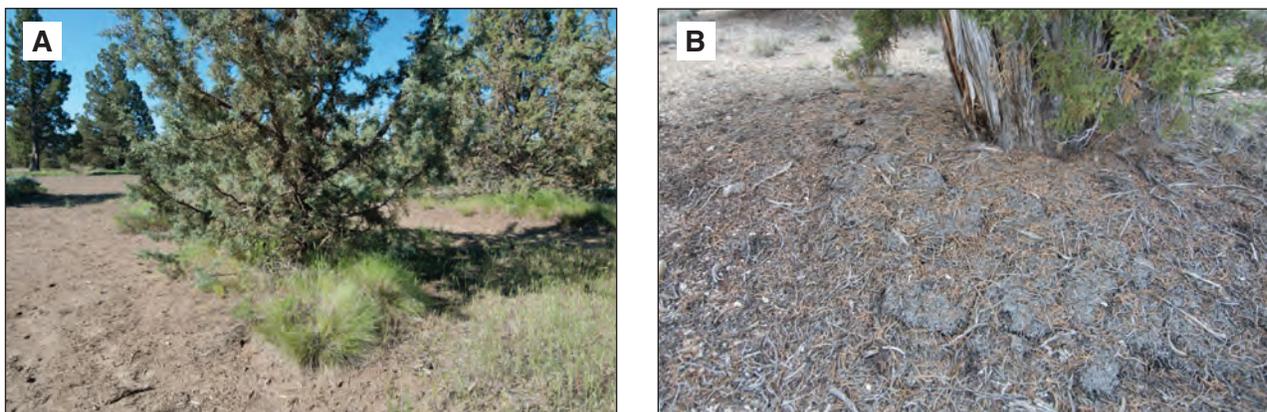
Figure 2-48—Do juniper species produce chemical inhibitors affecting understory growth? Discrepancies between current knowledge and what we observe in the field make the issue unclear. (A) Directly beneath the tree canopies, Idaho fescue can be found growing on the north side of the tree, while bluebunch wheatgrass is typically found on the south side—but in this case, cheatgrass grows on the south side of the tree. The lack of perennial grasses in the interspace indicate both poor past management and competition from the tree roots, which occupy the interspace. (B) Closed Phase III woodland with thick mats of duff beneath the tree canopy. Lack of herbaceous growth beneath may be due to chemical inhibitors, competition, or the physical barrier of the matted duff layer. (A) Central Oregon and (B) Great Basin National Park, Nevada. (Photos by Rick Miller, Oregon State University.)

Soil Nutrients

Carbon and Nitrogen

A number of factors influence the size of organic carbon (C) and nitrogen (N) pools—such as composition and structure of vegetation, depth to bedrock, coarse fragment content [particles between 0.08–9.8 inches (2–250 mm)], particle size distribution, soil bulk density, disturbance regimes, and climate (Jobbagy and Jackson 2000; Strand et al. 2008). The shift from sagebrush-grassland to pinyon and juniper woodland results in significant changes in the amounts and distribution of above- and belowground organic C and N pools (Rau et al. 2011a, 2011b, 2012). In treeless big sagebrush communities, total aboveground organic C and N averages 2 tons/acre (4.5 Mg/ha) and 0.13 tons/acre (0.3 Mg/ha), respectively (Rau et al. 2012). This is less than 10 percent of total estimated ecosystem C and N and a soil depth of 20.9 inches (530 mm) (figs. 2-49, 2-50). As pinyon and juniper increase to near closed-canopy conditions, aboveground biomass can account for 27.5 tons/acre (62 Mg/ha C) and 0.27 tons/acre (0.6 Mg/ha
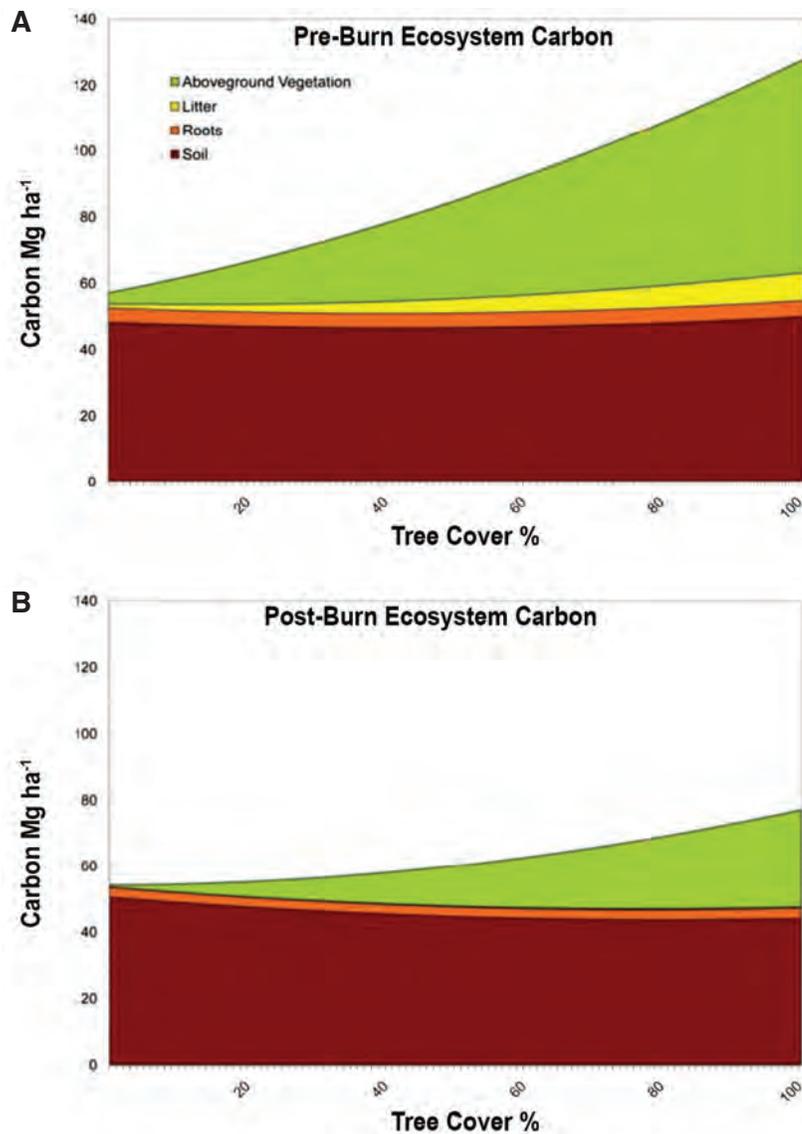
Figure 2-49—Changes in the carbon pool (A) before and (B) immediately after a moderate severity fire. Mass C Mg ha^{-1} is stacked, adding values for each ecosystem component starting at zero and ending at cumulative C mass for the ecosystem (Rau et al. 2012).

N), which is nearly 53 percent of total estimated ecosystem C and 13 percent of total estimated ecosystem N to a soil depth of 20.9 inches (530 mm). In addition to changes in aboveground pools, the increase in pinyon and juniper tree cover from early to late woodland succession resulted in an increase of 1.7 tons/acre (3.8 Mg/ha) of root organic C and 0.027 tons/acre (0.06 Mg/ha) of root N in the 0 to 35.4 inches (0 to 900 mm) of the soil profile. However, there is little change in total soil organic C and N pools, which includes coarse soil fragments.

Pinyon and juniper can also influence C and N cycling on the surface. Plant tissue can have C:N ratios of 30:1 or greater (Laungani and Knops 2009; Rau et al. 2011a). After microbial decomposition of soil organic C, C:N ratios are typically around 12:1. The higher ratios of C:N in pinyon and juniper litter compared to litter from shrubs and grasses in addition to dry climate can result in slow organic matter decomposition rates. Residence time of nondecomposed organic C beneath pinyon and juniper canopies can
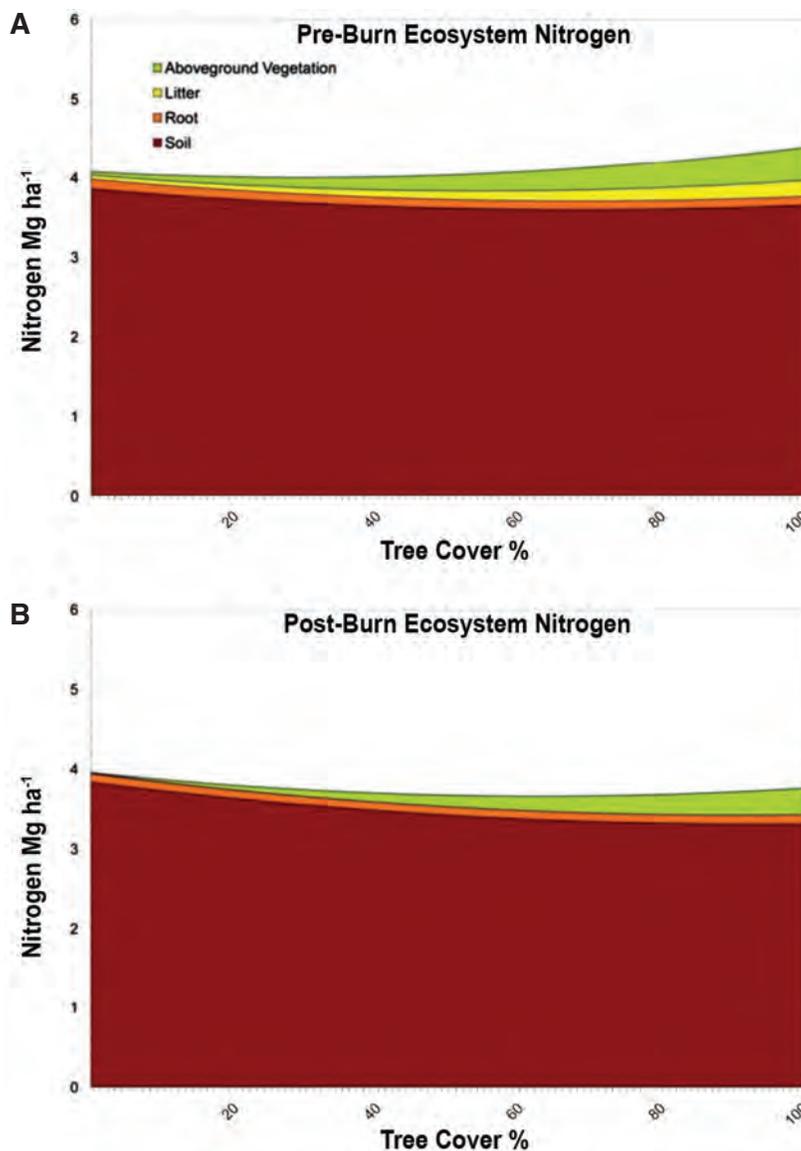
Figure 2-50—Changes in the nitrogen pool (A) before and (B) immediately after a moderate severity fire. Mass N Mg ha-1 is stacked, adding values for each ecosystem component starting at zero and ending at cumulative N mass for the ecosystem (Rau et al. 2012).

potentially be several centuries (Neff et al. 2009).

Although some consider the increased aboveground storage of C and N in pinyon and juniper woodlands a potential benefit, high-severity fires can result in large losses in both above- and belowground pools (Breshears et al. 2005; Breshears and Allen 2002; Rau et al. 2011b, 2012). The amount of biomass and total aboveground C combusted in pinyon and juniper woodlands is closely related to fire severity and plant composition. Low-intensity prescribed burns in sagebrush removed 83 percent of the aboveground biomass compared to 55 percent in denser woodlands (90 percent of the 1-hour fuels, with the majority of 10-, 100-, and 1,000-hour fuels remaining). However, since biomass in closed woodlands can be tenfold greater than sagebrush-grassland, significantly greater amounts of total aboveground organic C and N are lost. In high-severity wildfire events, combustion of trees released 70 percent of the organic C stored in aboveground biomass (fig. 2-51; Rau et al. 2010). It is likely that the majority of remaining aboveground dead biomass after such events, including tree roots, will decompose and result in the release of CO_2 directly into the atmosphere (Johnson and Curtis 2001). But charcoal and ash that remain are relatively resistant to decomposition and can persist many centuries in arid environments (Mensing et al. 1999).

Extensive tree mortality from droughts can also have large impacts on aboveground C and N, with losses similar to wildfires (Breshears et al. 2005; Breshears and Allen 2002). Although not widely documented in the Great Basin, significant drought mortality has occurred in the Colorado Plateau. An additional concern is the transition of woodlands, shrubland, and/or perennial grassland to exotic annual dominated communities, which could result in significant declines in C below 24 inches (600 mm) in soil depth (Rau et al. 2011a).

Canopy and Intercanopy Spatial Heterogeneity

There is significant spatial heterogeneity in organic C, N, and other nutrients at and near

Figure 2-51—In high severity wildfire events, combustion can release 70 percent of the organic C stored in aboveground biomass and considerable amounts of aboveground N. Stinkingwater Mountains, eastern Oregon. (Photo by Rick Miller, Oregon State University.)

USDA Forest Service RMRS-GTR-403. 2019.

the soil surface in persistent pinyon and juniper woodlands and more recent postsettlement woodlands. Litter and nutrient accumulation areas beneath the tree canopies can have three- to five times more organic C and N than the tree interspace (Klopatek 1987a,b; Neff et al. 2009). These large differences primarily occur at the surface and in the upper 3.9 inches (100 mm) of soil, with little or no differences at deeper soil depths (Neff et al. 2009; Reiley et al. 2010). Others have also reported tree canopy duff areas as accumulation areas of litter, organic C, and nutrients compared to the interspace (Bates et al. 2007a,b; Doescher et al. 1987; Everett et al. 1986a; Thran and Everett 1987).

Lower bulk density and higher pH, water holding capacity, and sand content also can occur beneath the tree canopy compared to the tree interspace (Klopatek 1987b). Benefits of these accumulation areas include relatively stable nutrient pools with very slow decomposition rates and protection of soil surface directly beneath the tree canopies from runoff and erosion. Residence time of organic C beneath tree canopies can be centuries (Neff et al. 2009). However, a drawback is that these pools are much more susceptible to loss from wildfires as compared to C stored below the top couple of inches of soil. Five to 8 years following tree removal, nutrient concentrations were still higher in the tree duff areas compared to tree interspace (Thran and Everett 1987). However, cutting and leaving trees on the ground increased decomposition rates, which were 37 percent greater than adjacent uncut woodlands (Bates et al. 2007b).

The benefits of these duff accumulation areas depend largely on how the interspace is being affected (Reiley et al. 2010). Mining of nutrients from the interspace to tree canopy duff areas has been reported by several researchers (Doescher et al. 1987; Klopatek 1987b; Tiedemann 1987). These accumulation areas may also be immobilizing nutrients for plant use. Nutrient cycling and uptake by understory vegetation may be more rapid in the tree interspace. In northcentral Arizona there were greater numbers of nitrifying bacteria in the interspaces than beneath the tree canopies, even though there was fourfold more total N beneath the tree canopies—suggesting greater N utilization in the interspace (Klopatek and Klopatek 1987). This difference could be due to slower decay rates caused by higher lignin content in tree foliage as compared to grass and shrub foliage in combination with generally drier conditions under the trees due to greater interception of rain and snow by the tree foliage as discussed in sections prior to this. However, energy of surface runoff can be increased by concentrating flows through the interspace, resulting in increased erosion resulting in the loss of N (Pierson et al. 2007). The potential negative effects of tree canopy accumulation areas depend largely on the structure and composition of interspace vegetation, which is influenced by soils (especially depth to a restrictive layer), disturbance history, and relative dominance of the tree canopy (fig. 2-52; Miller et al. 2000, 2005).

Climate Change and Effects on Pinyon and Juniper Persistence and Migration

Climates have been continually changing long before the formation of the first pinyon and juniper woodlands and have had significant impact on woodland distribution, migration, expansion, contraction, infill, composition, and structure. But recent concerns over natural climatic variation linked to anthropogenic effects on climate have raised considerable apprehension related to future impacts on agriculture, water, global economies, and wildland ecosystems including semiarid woodlands. Continued warming in the western United States is projected to have significant effects on pinyon and juniper woodlands and on factors related to all semiarid vegetation in the West, including wildfire, invasive plants, insect and disease outbreaks, snowpacks, and soil water availability.

Figure 2-52—Moderately deep to deep skeletal soils in this open Phase II old-growth woodland support perennial grass cover of 20–30 percent and 3–5 percent forb cover. Few sagebrush remain and there is limited recruitment of trees. Horse Ridge, central Oregon. (Photo by Rick Miller, Oregon State University.)

Projected Climate Change Across the Great Basin and Colorado Plateau

Global climate change models project future changes in temperature and precipitation based on concentrations of carbon dioxide (CO_2), and other greenhouse gases and information about the earth's surfaces and oceans. Climate change analyses use emissions models from the Coupled Model Intercomparison Project (CMIP) from either Phase 3 (CMIP3) or Phase 5 (CMIP5). The CMIP3 models, also known as the SRES models, assume a series of storylines concerning population growth, socio-economic development, and technological change with the A2 scenario representing business as usual (IPCC 2000). The CMIP5 models use representative concentration pathways (RCPs) based on the additional energy provided to the earth's system by greenhouse gas emissions, with RCP 8.5 representing high emissions rates and roughly equivalent to the A2 scenario (Taylor et al. 2012).

Predicting how potential change will impact pinyon and juniper depends on the degree of climate change and how climate, soils, and different plant species interact. While there is widespread scientific agreement concerning continued warming, uncertainty remains over how much additional warming will occur and how precipitation regimes will change (IPCC 2013; Melillo et al. 2014; USGCRP 2017). Changes in (1) snowpack and precipitation seasonality, amount, variability, and extremes, and (2) the frequency, duration, and severity of drought are important drivers of the potential niches for the different juniper and pinyon species. Changes in climate also drive disturbance regimes, such as insect and disease outbreaks, fire, and the extent and type of invasive plant species.

Average annual temperatures in the Great Basin and Colorado Plateau have increased, primarily in winter, with minimum temperatures increasing more than maximum temperatures (Abatzoglou and Kolden 2013; Kunkel et al. 2013a,b; USGCRP 2017). The daily difference between minimum and maximum temperatures have decreased in

fall, winter, and summer but have increased in spring (Rhines et al. 2017). The number of very warm days has increased in summer, decreased in winter, and has no clear trend in spring and fall (Yu et al. 2018). Warmer winters have led to more rain, reduced snowpack, earlier snowmelt, and earlier peak flows—with the largest changes in Oregon and northern Nevada (Fyfe et al. 2017; Gergel et al. 2017; Harpold and Brooks 2018; Harpold and Molotch 2015; Knowles 2015; Mote et al. 2018; Nayak et al. 2010). The number of days with snow cover has declined, particularly in November and March, largely due to increasing temperatures in these shoulder months (Klos et al. 2014; Knowles 2015).

Average annual precipitation has decreased in much of the western United States (Prein et al. 2016; Prein et al. 2017b; USGCRP 2017). Spring precipitation has increased slightly in the northern Great Basin (Abatzoglou et al. 2014; Kunkel et al. 2013b) while fall precipitation has increased in the central and southern Great Basin and Colorado Plateau (Kunkel et al. 2013a; USGCRP 2017). Some analyses show no statistically significant trend in annual or seasonal precipitation when analyzed over smaller geographic areas rather than multiple Major Land Resource Areas or multiple states (Abatzoglou et al. 2014; Kunkel et al. 2013a,b).

The western United States will continue to become drier overall, as projected increases in evapotranspiration outpace any increases in precipitation (Cook et al. 2004; Dai 2013; Peterson et al. 2013; USGCRP 2017). Climate scientists are uncertain how much drying will occur, largely due to difficulties in modeling summer precipitation (USGCRP 2017). Nonetheless, significant drying in spring and fall is expected by the end of the century if the current rate of greenhouse gas emissions continues.

Since 1901, the consecutive number of frost-free days has increased by 17 days, lengthening the growing season by 1–2 weeks in the western United States, but that has not resulted in an increase in plant productivity due to plant-specific temperature thresholds, plant-pollinator dependencies, and seasonal limits on water and nutrient availability (USGCRP 2017). Grasses green up earlier, but also senesce earlier as soil moisture is depleted earlier.

By mid-century, the growing season is expected to increase by 30–40 days in the northern Great Basin and by 20–40 days in the central and southern Great Basin and Colorado Plateau as compared to the 1976–2005 period (USGCRP 2017). However, growing season water balance deficits are expected to increase by mid-century and beyond except in areas with significant monsoonal precipitation (USGCRP 2017).

Temperature

Observed warming trends are expected to continue into the future (Abatzoglou and Barbero 2014; Dalton et al. 2013; Kunkel et al. 2013a,b; Melillo et al. 2014; USGCRP 2017), with the magnitude of any given change dependent on the climate model(s) and emissions or radiative forcing scenario used. At the current rate of emissions, climate models project average annual temperature to increase by 3–5 degrees Fahrenheit by mid-century and 5–9 degrees Fahrenheit by the end of the century (Garfin et al. 2014; Kunkel et al. 2013a,b; Mote et al. 2014; USGCRP 2017). Extreme temperatures would increase more than average temperatures, with the frequency and intensity of heatwaves increasing, and the frequency and intensity of cold snaps decreasing (Kunkel et al. 2013a,b; Melillo et al. 2014; USGCRP 2017). Winter temperatures are expected to increase more in the northern Great Basin than in the central and southern Great Basin and Colorado Plateau (Kunkel et al. 2013a,b; USGCRP 2017). Conversely, summer temperatures

are expected to increase more in the central and southern Great Basin than in the northern Great Basin (Kunkel et al. 2013a,b; USGCRP 2017). The number of days below freezing will decrease, while the freeze-free period and the number of days above 90 °F will increase (Melillo et al. 2014; USGCRP 2017).

Precipitation

Projected changes in precipitation are more uncertain than changes in temperature (IPCC 2013; Melillo et al. 2014; USGCRP 2017), although modeling of winter precipitation has improved in the western United States (USGCRP 2017). Changes in summer precipitation remain more uncertain due to the inability of climate models to represent the mesoscale convective storms that predominate in summer (USGCRP 2017).

At the current rate of greenhouse gas emissions, winter precipitation in the western United States is expected to increase 10–20 percent relative to the 1976-2005 average by the end of the century, while changes in spring, summer, and fall precipitation are expected to be minor relative to natural variability (USGCRP 2017). But many projections of precipitation show both a wide range of increases and decreases from current averages, thus including no change in precipitation amount (Abatzoglou et al. 2014; Dalton et al. 2013; Kunkel et al. 2013a,b), indicating high uncertainty. Pendergrass et al. (2017) project that precipitation variability will increase on all time scales (although less so in summer). The number of mesoscale scale convective storms will likely increase (Prein et al. 2017a), with individual storm cells both smaller and bringing heavier rainfall (Wasko et al. 2016).

Huang and Ullrich (2017) project an increase in the number of rainy days and the frequency of nonextreme precipitation through mid-century—but little change after that. The central and southern Great Basin and Colorado Plateau would experience an increase in warm season precipitation, and the northern Great Basin a decrease (Huang and Ullrich 2017). However, the North American monsoon is also expected to weaken significantly as the climate warms (Pascale et al. 2017), so the source of the warm season moisture for the central and southern Great Basin and Colorado Plateau is unclear. A potential source may be an increase in hurricanes and tropical storms from the Pacific that track into the southern Great Basin, bringing slow-moving, intense, and long-lasting thunderstorms (Kunkel et al. 2013a).

Precipitation extremes are expected to increase in frequency and intensity as warmer temperatures increase the water holding capacity of the atmosphere. Model results project greater change in the northern Great Basin than in the central and southern Great Basin and Colorado Plateau due to an increased incidence of long, narrow regions in atmospheric moisture flow during the cool season—narrow atmospheric rivers that transport much of the moisture from the tropics to the West (Dannenberg and Wise 2017; Prein et al. 2016; USGCRP 2017). The number of dry days and days with very light precipitation are projected to increase slightly, but the number of days with very heavy precipitation is projected to increase by more than 25 percent. The length of consecutive dry day periods may increase, although estimates on the magnitude of that change varies widely (Roque-Malo and Kumar 2017).

Snowpack

Snowpack provides a significant source of soil moisture throughout the range of pinyon and juniper woodlands (Gergel et al. 2017; Harpold and Molotch 2015), with peak annual soil moisture typically occurring during snowmelt (Harpold and Molotch 2015). Much of the Great Basin has an average winter temperature near freezing, making this area very sensitive to small changes in winter temperature (Lute and Abatzoglou 2014; Mote et al. 2005; Mote et al. 2018; Nayak et al. 2010; Safeeq et al. 2016; Tennant et al. 2017). By mid-century, at the current rate of greenhouse gas emissions, the area

expected to remain snow-dominated in winter would decrease by 3–42 percent and the area expected to become rain-dominated would increase by 23–65 percent, depending on location (Klos et al. 2014). The shift to rain-dominant winter precipitation will occur quickly in areas with gentle terrain and lower elevations across much of the northern Great Basin (Klos et al. 2014; Lute and Abatzoglou 2014). The result will be the decline in cover and amount of spring snow (Thackeray et al. 2016), although interannual variability in snow will remain an important driver of snowpack until mid-century, after which temperature will become the main driver (Safeeq et al. 2016; Thackeray et al. 2016). Snowpack is expected to all but disappear in nearly the entire pinyon-juniper zone by the end of the century (Safeeq et al. 2016).

Climate Change and Drought

There are many terms for drought—keeping the differences in mind are important for determining how drought may influence pinyon and juniper woodland extent, structure, and dynamics. Of most relevance to pinyon and juniper woodlands are (1) agricultural drought—soil moisture deficit insufficient for plant growth, (2) flash drought—relatively short periods of rapidly developing moisture deficit (Mo and Lettenmaier 2015, 2016), and (3) snow drought—snowpack deficit (Harpold et al. 2017). Ecological drought refers to the impacts on the environment and human economies from drought (Crausbay et al. 2017).

Multi-year and multi-decadal droughts are natural occurrences in the central and southern Great Basin and Colorado Plateau (USGCRP 2017). Severe droughts in the 12th, 13th and 16th centuries were of much greater spatial extent and longer duration, often lasting several decades, than any seen in the 20th century and thus far in the 21st (Peterson et al. 2013). Over the instrument record period, the interior West typically has experienced transient droughts characterized by short-term changes in duration, intensity—how hot and dry, and for how long—and severity, or overall impact (Ge et al. 2016). Drought duration and severity appear to be decreasing in the central and southern Great Basin and Colorado Plateau for intermediate and longer drought return intervals, but drought intensity is increasing for the Colorado Plateau for the longer return intervals, particularly in northern Arizona (Ge et al. 2016). Drought duration, intensity, and severity are increasing for 10-year interval time periods in the northern Great Basin but are more mixed for longer interval droughts (Ge et al. 2016). The evidence for flash droughts is relatively weak given high interannual variability in soil moisture, but one driven more by precipitation deficits is somewhat likely (Mo and Lettenmaier 2015, 2016). The number of months in moderate to extreme drought has been increasing in much of the Great Basin and Colorado Plateau (Peterson et al. 2013).

Some evidence indicates that droughts in the western United States are shifting from precipitation control to temperature control, but that evidence is limited by biases toward temperature in many models, and by the lack of soil moisture observations and subsurface modeling studies (USGCRP 2017). Other than how increasing temperatures are driving an overall increase in aridity, the evidence remains weak to show that climate change is currently driving an increase in agricultural drought frequency and severity outside the range of natural variability (USGCRP 2017). In simulations where temperature increase was not constrained, Jeong et al. (2014) projected a likely increase in drought, as measured by the Standardized Precipitation Evaporation Index (SPEI), over a 12-month period, along with increased drought duration and in cumulative drought and extreme drought severity across the Great Basin and Colorado Plateau. Longer-term droughts decrease deep soil moisture and groundwater levels. Evidence is much stronger that climate change is driving an increase in snow droughts due to the observed changes in snowfall and snowpack (USGCRP 2017).

Changes in flash droughts are more difficult to assess in semiarid climates. The incidence of heatwaves, an essential component of flash droughts, is increasing across the western United States with the northern Great Basin experiencing more heatwaves currently than in the recent past (Peterson et al. 2013). As indicated in the discussion of temperature, the frequency and severity of heatwaves are expected to increase in the future. That increase in heatwaves would seem to provide evidence for a potential increase in flash droughts, but it needs further study focused on the semiarid climates of the Intermountain West.

Climate Controls on Pinyon and Juniper Persistence and Migration

The climatic factors serving as top-down determinants of pinyon and juniper distribution include precipitation amount and precipitation seasonality, the frequency of freezing fogs in valley bottoms and Pacific fronts in late winter and early spring, and minimum winter temperatures (Cole et al. 2008a; Gibson 2011; Knutson and Pyke 2008; Munson et al. 2011; Nowak et al. 1994a; West et al. 1978). These factors also affect the frequency, duration, severity, and intensity of seasonal and longer droughts (Dai 2013; Ge et al. 2016; MacDonald and Tingstad 2007; Williams et al. 2010; Yu et al. 2018) with the exception of Pacific fronts and atmospheric circulation patterns, such as El Niño-Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO).

Decade-scale wet and dry periods play an important role in structuring semiarid plant communities with larger amplitude (greater differences between maximum and minimum wet and dry events) and shorter duration swings in climate, promoting more rapid vegetation changes than lower amplitude (small differences between wet and dry events), longer duration swings. Decadal to multidecadal variability may either dampen or amplify the net effects of changing climate. Prolonged wet and dry periods tend to affect large areas of the western United States simultaneously (Gray et al. 2006). Drought, insects, and small fires may influence structure of persistent pinyon and juniper woodlands more than large fires (Gray et al. 2006; Romme et al. 2009), resulting in episodic mortality and establishment events. Factors that limit or reduce tree canopy densities reduce the possibility of large crown fires, while lack of fine surface fuels restricts the spread of surface fires (Andrews 2018; Kennard and Moore 2013; Rothermel 1991). Hotter, drier sites with low soil moisture capacity limit pinyon and juniper recruitment following a significant mortality event (Redmond et al. 2018).

Projections of Future Distribution

Several studies have attempted to project how the ranges of pinyon and juniper might change using a bioclimatic envelope approach. Bioclimatic envelope models use climate to define and predict the potential geographic range of a species (Araújo and Peterson 2012; Jeschke and Strayer 2008). Although bioclimatic envelope studies generally cannot account for other important factors that determine a species' actual range, they still produce reasonable results since climate is a primary determinant of potential ranges at the broad scale (Araújo and Peterson 2012; Jeschke and Strayer 2008).

Results vary, depending on the use of single or multiple climate models, the specific climate models, and the emissions scenario. Common findings include species contraction along the southern limits of current ranges and an upward shift in elevation (Cole et al. 2013; Friggens et al. 2012; Gibson 2011; Kerns et al. 2018; Rehfeldt et al. 2006). Studies evaluating the potential impacts of climate change on the ranges of pinyon and juniper in the Great Basin are rare, with most focus on the Colorado Plateau and Southwest. Much of the recent research has focused on twoneedle pinyon following the significant mortality event in the early 2000s.

Three studies project expansion of twoneedle pinyon's climate niche into Nevada and Idaho and some displacement of singleleaf pinyon (Friggens et al. 2012; Gibson 2011;

Natural Resources Canada 2017 January 4c). But that possibility would depend on a northward shift in the North American monsoon, leading to a significant increase in summer precipitation (Gibson 2011). Since most climate models project a decrease in summer precipitation, and some models predict a weakening of the North American monsoon (Pascale et al. 2017), movement of twoneedle pinyon into the Great Basin seems unlikely. More likely responses are a reduction in extent on the Colorado Plateau with a shift upward in elevation and an increase in the central and southern Rockies (Cole et al. 2008b; Gibson 2011; Natural Resources Canada 2017 January 4c).

The singleleaf pinyon climate niche may expand farther into southern Idaho and is expected to persist in northcentral Nevada (Friggens et al. 2012; Gibson 2011), although some analyses project an increased risk of contraction in Nevada and Utah (Friggens et al. 2012). At least one modeling effort suggests that the climate niche for singleleaf pinyon could extend across eastern Oregon and southern Idaho by mid-century at higher elevations—assuming the current rate of emissions (Natural Resources Canada 2017 January 4d). However, the historical migration rates for singleleaf pinyon appear to be slower than the current rates of change (Cole et al. 2013).

Utah juniper's climate niche could expand in southern Idaho by 2030 and migrate into the southeastern corner of Oregon by mid-century, with a general shift northward early in the 21st century and contraction to higher elevations by the end of the century (Friggens et al. 2012). It could contract from western Nevada and persist or expand in eastern Nevada, southern Idaho, and western Utah (Gibson 2011). A different study projected that Utah juniper climate niche would move up in elevation by around 325 feet, retreat from Utah, persist in central Nevada, and expand into southwestern Idaho and southeastern Oregon (Rehfeldt et al. 2006). A more extreme analysis projects that the climate niche for Utah juniper could expand across eastern Oregon and Washington and southern Idaho and farther in Nevada and Utah (Natural Resources Canada 2017 January 4b).

One study projects that western juniper's climate niche could contract significantly and become limited to the Sierra Nevada Mountains in California and possibly the Blue Mountains of Oregon and Washington (Gibson 2011). Another predicts a similar contraction, but also indicates western juniper could persist near the Cascades in central Oregon as well (Natural Resources Canada 2017 January 4a). Kerns et al. (2018) found it unclear if western juniper would expand or contract in the Blue Mountains, as some dynamic vegetation models treat novel climates as unsuitable, but the actual suitability of those climates is not known. Conversely, Creutzburg et al. (2015) projected that western juniper would continue expanding in Oregon, but at a slower rate than under the current climate. Halofsky et al. (2013) found that western juniper could expand in central Oregon, moving into areas currently occupied by ponderosa and lodgepole pine through mid-century, but then decline to near current levels by the end of the century, with expansion and contraction rates and timing varying by climate model.

In order for pinyon and juniper ranges to shift upward in elevation and northward in latitude, they must be able to disperse in appropriate directions. However, recent emphasis on reducing pinyon and juniper encroachment into sagebrush to protect and restore habitat for sagebrush obligate species, reduce fire risks, and preserve certain landscapes and plant communities may reduce the species migrations discussed above. This potential conflict in certain land management objectives and adapting to climate change has been recognized at City of Rocks National Reserve in southeastern Idaho, the current northernmost

limit of singleleaf pinyon (Powell et al. 2013). Prevention of pinyon and juniper migration to meet other land use objectives could lead to species contraction, or in more extreme cases, local extirpation.

Factors Influencing Pinyon and Juniper Persistence and Migration

Recruitment

Pinyon and juniper expansion rates are limited by seed dispersal distances and the conditions needed for establishment (Bradley 2010). Climate and climate change influence interannual variability in seed production, seed viability, and seedling establishment (see Seed and Seedling Ecology subsection) as well as tree growth and maturation rates (Enright et al. 2015; Knutson and Pyke 2008; Munson et al. 2011; Redmond and Barger 2013; Redmond et al. 2012; Redmond et al. 2017; Williams et al. 2010). In addition, higher temperatures generally accelerate the development of insects while increased atmospheric CO_2 concentrations reduce the nutritional quality of plants by changing the C:N ratio. That change leads to higher levels of herbivory so that the animal can get enough protein (Bale et al. 2002; DeLucia et al. 2012), although the impacts of climate change on seed predators remains little studied (Lewis and Gripenberg 2008). If seed predators respond the same way as defoliators and bark beetles, climate change would effectively reduce seed crops in both pinyon and juniper. Most pinyon and juniper seedlings are associated with scatter caches by rodents and birds (pinyon) or fecal deposits from birds and mammals (juniper) with most establishment under nurse plants (Chambers 2001; Chambers et al. 1999a). As such, the potential migration or persistence of pinyon and juniper depends in part on how climate change may impact their dispersers and suitable nurse plants (Mueller et al. 2005b). Changes in recruitment and maturation in combination with changes in disturbance regimes can affect the ability of a species to migrate and persist in the face of climate change (Enright et al. 2015).

Drought Effects on Persistence and Migration

A cycle of multiple wet years followed by multiple dry years typifies the Great Basin (fig. 3-12a; Miller et al. 2011; Pilliod et al. 2017). Although not explicitly studied, a similar cycle of wet and dry years likely occurs in the Colorado Plateau (fig. 3-12b). Such cycles are related to the ENSO index with more protracted droughts driven in part by sea surface temperatures consistent with ENSO as well as the PDO and Atlantic Multidecadal Oscillation (AMO) (Breshears et al. 2005; Shinneman and Baker 2009). Drought is probably the single greatest cause of pinyon and juniper mortality—in large part because it is a subregional to regional event rather than a localized event and is often linked to insect and disease infestations. In both the Great Basin and Colorado Plateau, drought can kill pinyon and juniper directly or, more commonly, facilitate the occurrence of another factor such as insect outbreak, wildfire, or the spread of invasive annual grasses.

Drought directly kills pinyon and to a lesser degree juniper through carbon starvation, hydraulic failure, or both (McDowell et al. 2008; Plaut et al. 2013; Sevanto et al. 2014) and increased vulnerability of drought-stressed trees to infestations (Gaylord et al. 2013). In the early 2000s, drought events had been characterized as a climate-change-type drought (e.g., Bowker et al. 2012; Breshears et al. 2005; Breshears et al. 2009; Clifford et al. 2011). A climate-change-type drought is a prolonged, extreme drought resulting in mass mortality of woody plants across regional or subcontinental scales and driven by warmer temperatures (Bowker et al. 2012; Breshears et al. 2009; Huang et al. 2015). This type of drought results in large-scale reorganization of woody plant distribution in a matter of years or decades, instead of centuries or millennia. For example, severe drought in northern New Mexico in the 1950s resulted in a 1.2 mile upward shift in the ecotone

between ponderosa pine and pinyon and juniper woodland that has persisted for at least 40 years, with the change occurring in less than 5 years (Allen and Breshears 1998). A multi-decade drought in the 13th century on Dutch John Mountain in Utah apparently resulted in widespread mortality of Utah juniper, which was then largely replaced by twoneedle pinyon during subsequent wet periods in the 14th century (Gray et al. 2006). The drought in the early 2000s reduced canopy cover by 55 percent in northern Arizona, exceeding the 32 percent gain in canopy cover between 1936 and drought onset (Clifford et al. 2011).

Widespread mass mortality events have been documented and studied in the Southwest and Colorado Plateau from the 1950s and early 2000s, largely involving twoneedle pinyon and oneseed juniper. Drought-related mortality events of similar magnitudes have not been documented in the Great Basin, although smaller scale drought-related die-offs have occurred in Nevada and southern California (e.g., Biondi and Bradley 2013; Greenwood and Weisberg 2008; Meddens et al. 2015). The primary difference between the 1950s drought and the 2000s drought was temperature. The drought in the early 2000s was not as dry as the 1950s drought but was considerably warmer across the range of twoneedle pinyon (Breshears et al. 2005).

Warming temperatures are expected to lead to mass mortality events across the Southwest throughout the 21st century (Adams et al. 2009; Adams et al. 2017; Clifford et al. 2011; McDowell et al. 2016). Increasing temperatures lead to increased water stress by increasing atmospheric demand for water—accelerating the rate of soil drying and increasing tree respiration (Adams et al. 2009; Macalady and Bugmann 2014; McDowell et al. 2008). Sites already subject to chronic water stress and that have experienced high mortality in past droughts are more likely to experience higher tree mortality than wetter sites (Clifford et al. 2011; Greenwood and Weisberg 2008; Mueller et al. 2005a; Peterman et al. 2013). Adams et al. (2009) found that increasing temperatures shorten the time to mortality in twoneedle pinyon from about 6 months to 4 months with an expected fivefold increase in the frequency of regional die-offs. McDowell et al. (2016) projected a greater than 50 percent decrease in conifers across the northern hemisphere by the end of the 21st century. But they also stated that the decrease could be an underestimation due to the need to oversimplify tree physiology in dynamic global vegetation models— the models also underestimated tree mortality in the early 2000s drought. Those same models projected one to three die-offs in the southern Great Basin and southern Colorado Plateau but no mortality events in the remainder of the Great Basin and Colorado Plateau (McDowell et al. 2016).

Several studies have attempted to develop climate-related predictors of drought mortality. In northern New Mexico, twoneedle pinyon die-off occurred where the 2-year cumulative precipitation was less than 24 inches (600 mm) and where warm season vapor pressure deficit exceeded 1.7 kPa (Clifford et al. 2013). Huang et al. (2015) identified a tipping point in mortality rates when the 11-month standardized precipitation evaporation index (SPEI, based on both precipitation and evapotranspiration) reached -1.64, using July as the end month. Persistence of drought over 11 months at that severity was associated with negligible or no radial growth in twoneedle pinyon pine and subsequent mortality. Breshears et al. (2009) found twoneedle pinyon died when leaf water potential stayed below the zero carbon assimilation point for at least 10 months. Adams et al. (2017) found that the time to mortality for twoneedle pinyon seedlings decreased by 5.2 percent for each 1.8 °F increase in ambient temperature. A temperature increase of 7.2 °F would result in 7.5 seedling mortality events from drought per century while a 14.4 °F increase would result in 14.7 events per century (Adams et al. 2017). Since all these studies involved only twoneedle pinyon, it is not clear if these potential predictors would apply to singleleaf pinyon, and juniper is generally more drought resistant then pinyon.

Insect and Disease Outbreaks Effects on Persistence and Migration

Insects and disease outbreaks have been closely linked to climate, particularly drought conditions, resulting in extensive woodland mortality. These interactions are discussed in the subsection "Insects, Nonvascular Plants, and Disease Associated with Pinyon and Juniper." Future projections of climate change are expected to increase stress in trees, increasing their vulnerability to various pests.

Wildfire Effects on Persistence and Migration

Several studies indicate fire regimes are changing in pinyon and juniper vegetation types, with increases in burned area, in fire size (for at least the largest fires), and in the number of fires (e.g., Balch et al. 2013; Board et al. 2018; Dennison et al. 2014; Morton et al. 2013). The fire season is getting longer in the central and southern Great Basin (based on start dates for large fires), and fire rotations are getting shorter in all pinyon and juniper vegetation types (Board et al. 2018). Some of the shortening, however, may be more due to the effects of invasive annual grasses and increases in human ignitions than to changing climate (Balch et al. 2013; Floyd et al. 2017; Romme et al. 2009). Many studies found that historical fires in persistent pinyon and juniper woodlands were typically infrequent, small, and high-severity events (Bauer and Weisberg 2009; Floyd et al. 2008; Floyd et al. 2017; Romme et al. 2009; Shinneman and Baker 2009).

Fires in pinyon and juniper savanna and at the ecotone with ponderosa pine or mixed conifer were more likely spreading, low-severity fires (Biondi et al. 2011; Cheek et al. 2012; Margolis 2014; Miller et al. 2005). Infilling of trees that increase canopy continuity, invasive annual grasses, and wet and dry cycles all appear to increase the risk or incidence and size of mixed and high-severity fire (Balch et al. 2013; Board et al. 2018; Cheek et al. 2012; Floyd et al. 2004; Floyd et al. 2017; Romme et al. 2009). Projections of how much change has actually occurred depends on the time period assessed, the geographic divisions used, whether pinyon and juniper woodlands are lumped with forests, and how a burned area was defined. Regardless, the consistency of these analyses in change (increasing) indicates that changes are not artifacts of the various methodologies and data sets used.

Understanding climate-fire connections and interactions is important to understand why fire regimes have been changing in recent years, and how they may change in the future. Climate factors influencing fire regime characteristics include cool season storm track position (Dannenberg and Wise 2017), current and antecedent drought status (Abatzoglou and Kolden 2011; Kitzberger et al. 2017; Littell et al. 2009; Morton et al. 2013), and temperature and precipitation in different seasons (Abatzoglou and Kolden 2011; Littell et al. 2009; Morton et al. 2013; Sheehan et al. 2015)—all of which vary across the Great Basin and Colorado Plateau. By influencing soil moisture availability and how rapidly or slowly it is depleted, these factors affect the arrangement of different types of live and dead fuels, the dominant vegetation, and live fuel moisture, which equates to fuel amount, continuity, and availability. Further, these factors drive seasonal drying and drought status, the top-down climate drivers of fire regimes (Abatzoglou and Kolden 2011, 2013; Kitzberger et al. 2017; Littell et al. 2009; Morton et al. 2013; Sheehan et al. 2015). Short-lived critical fire weather and the incidence of dry lightning events are important drivers for individual fires as well (Abatzoglou and Kolden 2011).

It is not clear whether pinyon and juniper woodlands function more like forests or like sagebrush and grasslands for the relative importance of antecedent conditions verses year-of-fire conditions in controlling area burned. Analyses rarely separate these woodlands from other major vegetation types. In dense pinyon and juniper woodlands with little vegetation in the understory, year-of-fire conditions may well be more important, given that stand structure is more similar to moister conifer forests (fig. 2-53).
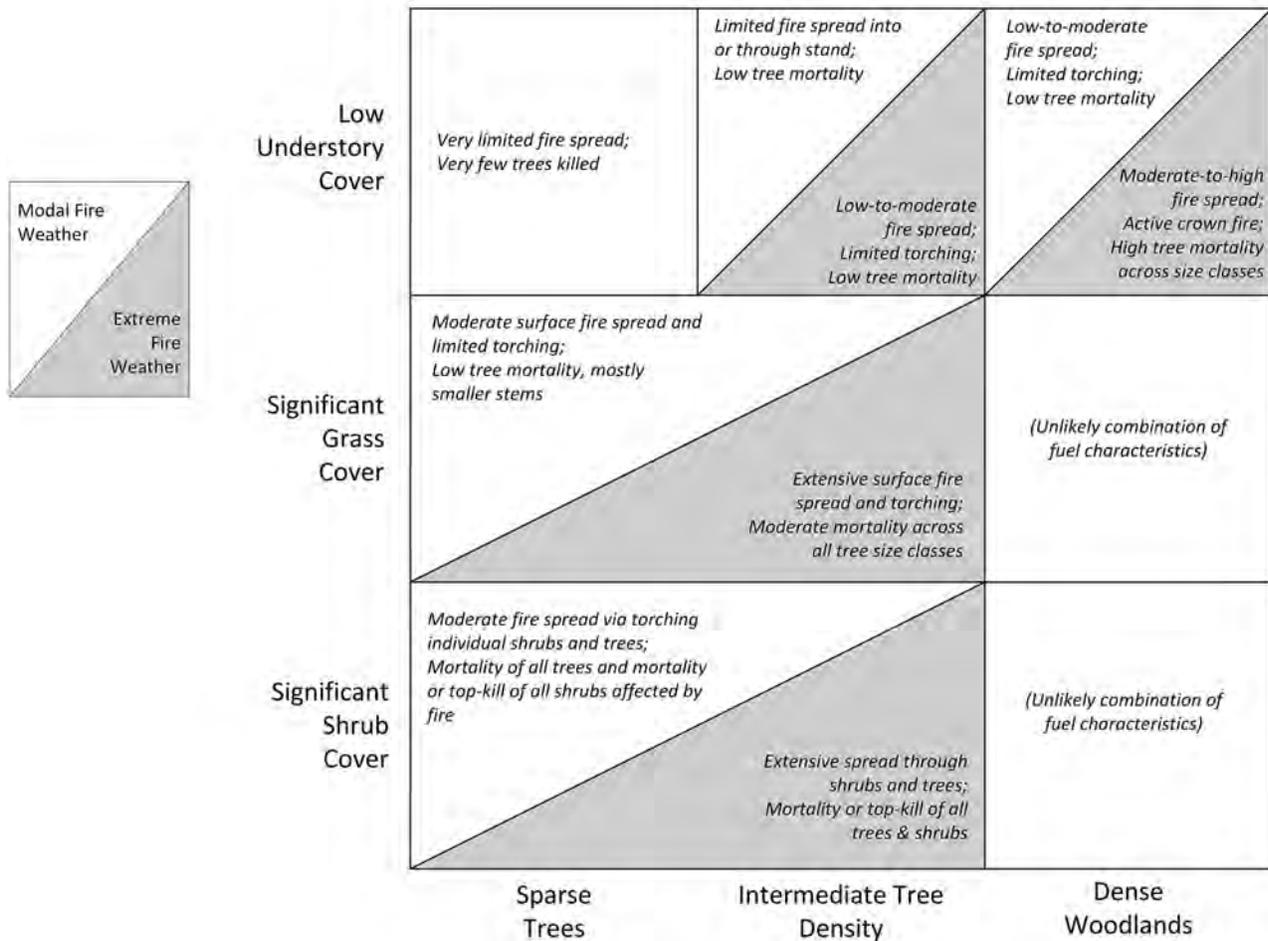
Figure 2-53—Probable fire behavior following a single ignition event in pinyon and juniper vegetation with respect to variability in tree density (horizontal axis) and understory fuel characteristics (vertical axis). Split cells reflect variable fire behavior, spread dynamics, and tree mortality under modal (80th percentile fire weather) in the unshaded upper left vs. extreme (95th percentile) fire weather conditions in the shaded lower right (from Romme et al. 2009).

In pinyon and juniper savannas with open canopies and a high proportion of grasses, or a shrub-steppe in the early stages of encroachment with high cover of shrubs and grasses, antecedent conditions may well be more important for building the necessary surface fuel loadings and continuity. In pinyon and juniper woodlands with an intermediate stand structure and understory composition between savanna and dense woodland, both antecedent and year-of-fire conditions may be important, the relative importance of each shifting as tree canopy cover increases and understory shrubs and grasses decline.

In forests, the amount of area burned is strongly correlated with temperature, precipitation, and drought in the year of the fire (Abatzoglou and Kolden 2011; Kitzberger et al. 2017; Littell et al. 2009; Westerling et al. 2006). These correlations are present but weaker in nonforest vegetation community types. Cooler temperatures and increased precipitation in the previous 1 to 3 years (usually winter and spring in the Great Basin and Colorado Plateau and summer in the Colorado Plateau) are more strongly correlated with area burned (Abatzoglou and Kolden 2011; Crimmins and Comrie 2004; Kitzberger et al. 2017; Littell et al. 2009; Margolis 2014). These conditions produce the needed surface fuel amount and continuity to support large fires in subsequent years (Littell et al. 2009; Morton et al. 2013). Morton et al. (2013) found that prolonged dry

periods of 2–6 months are still needed in grasslands and woodlands where live fuels dominate the surface fuels. Such conditions essentially convert live grasses into fine, dead fuels and are likely important in driving the live moisture content downwards in shrubs and conifers.

Climate controls on the amount of area burned change during the fire season. Even in forests, antecedent conditions are important only in the early part of the fire season (Abatzoglou and Kolden 2011; Morton et al. 2013), largely affecting season start date, as indicated by the first occurrence of a large fire, usually defined as a fire of at least 400 ha (1,000 acres). In-season conditions largely drive area burned and indicators of fire season severity in the middle and later parts of the fire season (Abatzoglou and Kolden 2011; Morton et al. 2013). In any ecosystem where live fuel moisture content is an important driver of fire season characteristics and, ultimately, fire regime characteristics, measures of moisture availability and moisture demand are better indicators of fire season potential than temperature and precipitation alone (Abatzoglou and Kolden 2011, 2013).

Potential Effects of Climate Change on Fire Regimes

Understanding how climate change may affect fire regimes in pinyon and juniper woodlands requires an understanding of how various indicators of drought occurrence, frequency, and severity may change. Measures to consider include seasonal and interannual changes in soil moisture, cloudiness, and vapor pressure deficit or evapotranspiration demand (Abatzoglou and Kolden 2011, 2013; Gergel et al. 2017; Kitzberger et al. 2017; Morton et al. 2013; Parks et al. 2016). However, predicting changes in cloudiness, which affects incoming shortwave radiation, and precipitation amount and timing have proven difficult in climate models, increasing the uncertainty in projections of future fire regimes. In addition, some expected changes differ across the Great Basin and Colorado Plateaus.

The northern Great Basin is expected to see an increase in the frequency of wet winters, with a greater fraction of annual precipitation occurring in November through March (Abatzoglou and Kolden 2011) and decreases in dead fuel moisture content (Gergel et al. 2017). The southern Great Basin and northern Colorado Plateau are projected to see little change in the frequency of wet winters and in dead fuel moisture content (Abatzoglou and Kolden 2011; Gergel et al. 2017). The southern portion of the Colorado Plateau is projected to experience an increase in the frequency of dry winters and decreases in dead fuel moisture content (Abatzoglou and Kolden 2011; Gergel et al. 2017). Across the area, a persistent northward shift in cool season storm tracks would reduce precipitation—with more area affected closer to the U.S./Canada border as the storm tracks shift (Dannenberg and Wise 2017), affecting soil moisture content in spring.

As discussed earlier, an increase in cool season rain and decrease in snow would lead to an earlier start to the growing season and earlier depletion of soil moisture, affecting live fuel moisture content during the fire season. Modeling by Parks et al. (2016) indicated soil water deficits would increase in summer, with greater deficit increases in the southern Great Basin and Colorado Plateau than the northern Great Basin. Littell et al. (2016) also found increases in soil water deficit, but the greater increases in the northern Great Basin and eastern Nevada, and decreases in much of the Colorado Plateau. In contrast, Gergel et al. (2017) projected increases in summer soil moisture in lowlands, but with considerable variability in model outputs, and decreases in higher elevations due to earlier snowmelt. Given the projected increases in temperature and associated evapotranspiration demand, it seems decreases in summer soil moisture deficit are more likely, except in areas where summer moisture remains abundant. Fire season would start earlier, with the median start date advancing by 1 to 4 weeks with an additional 1 to 3 weeks of extreme fire danger across the Great Basin and Columbia and

Colorado Plateaus (Abatzoglou and Kolden 2011). Fire return intervals are projected to decrease, and average or median annual area burned to increase (Abatzoglou and Kolden 2011; Sheehan et al. 2015). Continued aggressive fire suppression would somewhat lengthen mean fire return intervals and reduce average annual area burned as compared to the present but would likely lead to an increase in fire severity due to an increase in fuel loading and continuity (Parks et al. 2016; Sheehan et al. 2015).

Invasive Annual Grass Effects on Persistence and Migration

The ability of invasive annual grasses to establish and persist in a given area depends on several factors. First, environmental characteristics, such as the timing and amount of precipitation, soil temperatures, and other soil characteristics, determine those conditions in which a species can maintain itself in the absence of competition, facilitation, and other species interactions. This is the fundamental niche as defined by Hutchinson (1957) and it determines the broadest extents of a species' potential distribution on the landscape. Second, interactions of a species with the native plant community determine those conditions under which a species can actually persist. This is the realized niche; it is a subset of the fundamental niche and it determines the actual distribution of an invading species on the landscape (Hutchinson 1957). Finally, disturbances such as human development, inappropriate livestock grazing, wildfire, management actions, and climate change can alter both environmental conditions and species interactions and thus the fundamental and realized niches of invasive plant species.

The distributions of invasive annual grasses are related to mean annual maximum and minimum temperatures and mean annual precipitation (Brooks et al. 2016) and to the relative abundances of woody vs. perennial herbaceous species in the plant community (Bradford and Lauenroth 2006; Chambers et al. 2016a). Several studies show that cheatgrass is limited by cold soil temperatures at high elevations and low soil water availability at low elevations but is common over a wide range of intermediate elevations due to favorable conditions for its establishment, reproduction, and persistence (e.g., Chambers et al. 2007; Meyer et al. 2001). Perennial herbaceous species, especially grasses, are among the strongest competitors for invasive annual grasses, like cheatgrass (Bansal and Sheley 2016; Bradford and Lauenroth 2006; Brummer et al. 2016).

The relative abundance of woody plants or grasses is strongly influenced by differences in the overlap between the wettest part of the year and the warmest part of the year (Sala et al. 1997). The relative abundance of shrubs increases with a higher proportion winter precipitation and more deep soil water storage, while the relative abundance of perennial grasses increases with a higher proportion of spring and summer precipitation (Sala et al. 1997). Resistance to invasive annual grasses like cheatgrass increases as (1) temperatures decrease, precipitation increases, and productivity of the native community increases, (2) the proportion of precipitation that arrives in summer increases and lifeform dominance switches from shrubs to perennial grasses, and (3) the increase in number of years with insufficient precipitation for annual grass establishment and reproduction (fig. 2-53; Chambers et al. 2016b). In the Great Basin, cheatgrass cover after fire was lower on sites with wetter winters and early springs that favored perennial herbs, while cover was higher on sites with warmer and wetter falls and warmer late springs that favor its germination and growth (Roundy et al. 2018). Resistance can decrease with increased variability in precipitation, especially if rainfall is higher at times that enhance establishment and reproduction of invasive annual grasses (Abatzoglou and Kolden 2011).

Invasive annual grasses inhibit establishment and reestablishment of both pinyon and juniper by increasing the fire frequency (Nowak et al. 1994a) and preventing the reestablishment of suitable nurse shrubs (Miller et al. 2008). Climate change may favor

invasive annual grasses over native perennial grasses (Abatzoglou and Kolden 2011; Bradley et al. 2016), but this will depend on how climatic suitability for the individual species shifts, the capacity of a species to migrate to a climatically suitable area, and the interacting effects of both increasing human disturbance and changes in wildfire regimes. Pinyon and juniper expansion may also favor invasive annual grasses by reducing competitors in the understory in areas that are climatically suitable for the invasive annual grasses (Chambers et al. 2014; Miller et al. 2005; Miller et al. 2008). As the native understory declines, invasive annual grasses, such as cheatgrass, can exploit the increase in soil water and nutrient availability, especially after fire or management actions that reduce trees (Roundy et al. 2018, 2014a). Most research available on invasive species investigates how cheatgrass may respond to climate change; other invasive annual grasses remain little studied.

Less frequent but heavier spring rain events appear to favor medusahead over cheatgrass due to enhanced root growth (Bansal et al. 2014). Cheatgrass and ventenata, or North Africa grass, appear to respond more to total cumulative soil moisture than to different sizes and frequencies of water pulses (Bansal et al. 2014). Ventenata closely associates with medusahead in ephemerally wet soils with a higher clay content and lower phosphorus and potassium concentration (Jones et al. 2018) so may respond similarly to medusahead as the climate changes.

Cheatgrass exhibits high phenotypic plasticity for flowering, growth, and seed production and has the potential for rapid response to climate change (Hufft and Zelikova 2016). Warming promotes earlier growth and flowering, a longer effective growing season, and increased seed and biomass production, but only following wet winters and early springs with high soil water availability (Hufft and Zelikova 2016; Wolkovich and Cleland 2014). Warming and reduced summer precipitation is projected to favor expansion of cheatgrass into higher elevations and contraction in lower elevations; the greater the change in summer conditions, the greater the expected shift in cheatgrass (Bradley et al. 2016). Red brome is already present throughout the Great Basin and may replace cheatgrass at lower elevations (Bradley et al. 2016).

In the warmer central and southern Great Basin, increasing aridity and drought frequency will likely increase native plant stress and mortality, thereby creating opportunities for invasive annual grasses, with cheatgrass expansion at higher elevations and red brome expansion at lower elevations (Abatzoglou and Kolden 2011; Bradley et al. 2016). In the cooler northern Great Basin, warmer, wetter winters and an increased frequency of wet winters will likely favor cheatgrass germination, growth, and seed production (Abatzoglou and Kolden 2011; Balch et al. 2013; Bradley et al. 2016; Pilliod et al. 2017; Wolkovich and Cleland 2014; Zelikova et al. 2013). In contrast, if winters are drier and warmer, cheatgrass may exhibit progressively lower biomass and reproduction as temperature increases (Zelikova et al. 2013).

Across the Great Basin, climate change will likely enhance the annual grass-fire cycle (Abatzoglou and Kolden 2011). Several studies found that increased atmospheric CO_2 concentrations resulted in increased cheatgrass biomass, and thus fuels, when soil water and nutrients were not limiting to its establishment and growth (Hungate et al. 1996; Larigauderie et al. 1988; Nowak et al. 2004). Elevated CO_2 may also increase cellulose and lignin content in cheatgrass, which reduces digestibility and decomposition rates (Ziska et al. 2005) thereby increasing down and standing litter. Cheatgrass may have a more noticeable response to higher atmospheric CO_2 concentrations in the northern Great Basin if projections for wetter winters are accurate.

SECTION 3: 20,000 YEARS OF WOODLAND HISTORY

Climate is the primary driver of vegetation dynamics in both time and space—through its effects over seed crops, plant establishment, mortality, persistence, and the long-term and pervasive influence on disturbance regimes.

Summary

Presettlement

Climate is the primary driver of woodland dynamics, resulting in expansion, contraction, migration, and changes in woodland structure and species composition (Cole et al. 2013; Nowak et al. 1994a; Wigand et al. 1995). During the Glacial Maximum 20,000 years ago, precipitation was 1.3 to 1.5 times greater across the Intermountain Region, and temperatures 9.9–12.6 °F cooler in the Great Basin and 5.4–7.2 °F cooler in the Colorado Plateau than current conditions. In response to long-term periods of changing climate, pinyon and juniper woodlands moved up and down in elevation by as much as 3,000 feet and individual tree species have migrated hundreds of miles north. Transitions between cool and wet and warm and dry periods typically resulted in significant changes in woodland distribution, abundance, and structure.

Although climate is a major factor influencing pinyon and juniper woodland dynamics, it is closely linked to important woodland disturbance factors including fire, insect infestations, and disease. The transition from the cool and wet Neoglacial to the warm and dry Medieval Warm Period resulted in a large reduction of pinyon and juniper woodlands and is marked by significant increases in ash and large charcoal suggesting a large increase in fire across the Intermountain West. It is also likely that insect infestations increased during increasing drought conditions. Although climate is the primary driver of vegetation dynamics, its effects on woodland dynamics cannot be separated from other disturbance factors.

Postsettlement

The 1850s marked the end of the Little Ice Age and the early 1860s a significant rise in modern anthropogenic impacts throughout the Intermountain West. Prior to Eurasian settlement, evidence suggests pinyon and juniper woodlands were slowly expanding and infilling (Mehringer and Wigand 1990). Based on multiple tree-ring studies across the Great Basin and Colorado Plateau, rates of tree infill into woodlands and savanna, and expansion into adjacent sagebrush steppe, shrublands, grasslands, and riparian have significantly increased shortly after Eurasian settlement (Miller et al. 2008). This increase most commonly peaked in the early 1900s across both the Great Basin and Colorado Plateau, during a persistently wet period in the West between 1905 and 1917 (Barger et al. 2009; Biondi and Bradley 2013; Floyd-Hanna et al. 2004; Margolis 2014; Miller et al. 2008). The rapid increase in tree expansion and infill is attributed to climate, grazing, reduced fire occurrence where surface fuels were once adequate to support surface fires, and increased CO_2 levels, which increases water use efficiency in conifers.

Climate conditions in the late 1800s and early 1900s were ideal for cone production, tree-seedling establishment, and rapid growth rates. However, periods of increased moisture can result in the accumulations of fine fuels, resulting in the increase of widespread fires across the West. But large numbers of livestock in the late 1800s and early 1900s significantly reduced fine fuel loads and coincides with a significant decline in fires. The sudden decline in fire occurrences in the late 1800s was like no other in the last 3,000 years (Marlon et al. 2012). Several forest inventories in Nevada, Utah, and eastern California reported over 60 percent of the pinyon and juniper woodlands were

less than 150 years old (Bolsinger 1989; Menlove et al. 2013; O'Brien and Woudenberg 1999). But tree chronologies and inventories also indicate a significant portion of pinyon and juniper woodlands in the West are persistent, often showing old-growth traits. However, many of these stands have also changed in structure resulting from infill over the past 170 years.

Prehistory of Expansion, Infill, and Contraction

Dramatic changes in temperature and the abundance and seasonal distribution of precipitation have occurred across the Intermountain Region over the past 100,000 years. The most significant change in climate occurred during the transition between the Glacial Maximum (20,000 ± 2,000 years ago) in the late Pleistocene (126,000 ± 5,000 to 11,000 years ago) to the peak of the Holocene Thermal Maximum (around 6,000 years ago) (fig. 3-1; Renssen et al. 2009). The Glacial Maximum was the period when ice sheets extended across the most land surface, temperatures were coolest, and evapotranspiration at its lowest in the past 130,000 years (fig. 3-2). In contrast, the Holocene Thermal Maximum was the warmest and driest period resulting from storm tracks being deflected northward. These changes in temperature and moisture had significant effects on woodland and shrubland communities, soils, and hydrology across the Intermountain Region. Lakes and marshes expanded and contracted and elements of current shrubland and woodland communities moved up and down in elevation, in some cases forming communities with entirely new compositions of plants. Some tree species migrated hundreds of miles northward with rising temperatures during the late Pleistocene and Holocene. But, arguably, the most dramatic changes in Intermountain vegetation have occurred in the last 200 years.
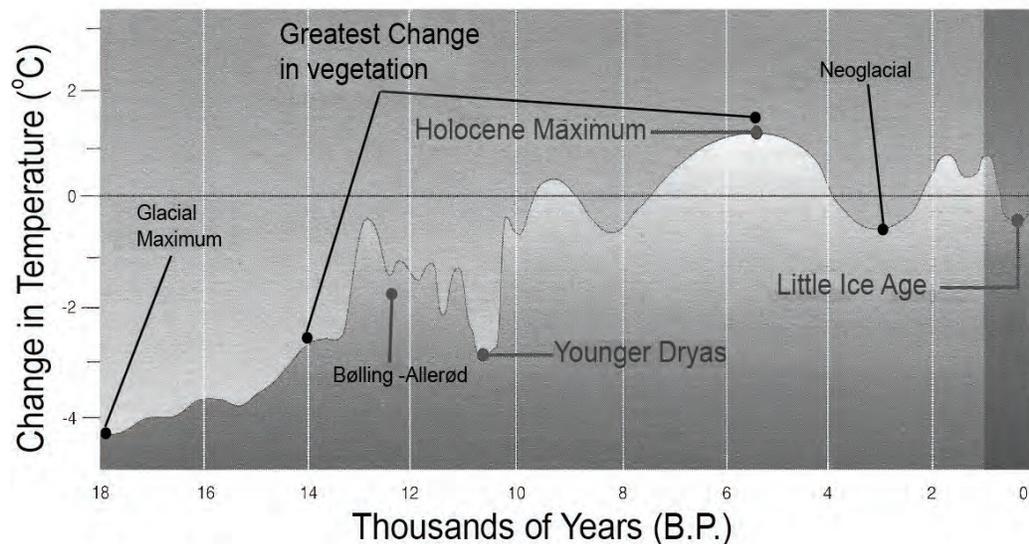
Figure 3-1—Variations in surface temperature for the past 18,000 years as estimated from a variety of sources, principally isotope ratios from Greenland ice cores. The onset and subsequent surface temperatures characteristic of the present interglacial or Holocene Epoch are highlighted. Of note are century-scale oscillations in temperature during the period of deglaciation between 15,000 and 10,000 years B.P. and a broad Holocene maximum about 5000 to 6000 years B.P. when summer temperatures may have been 1 to 2 °C (1.8 to 3.6 °F) warmer than the present era (Eddy and Bradley 1991). The estimated greatest change in vegetation occurred between 5,000–14,000 years ago.

USDA Forest Service RMRS-GTR-403. 2019.
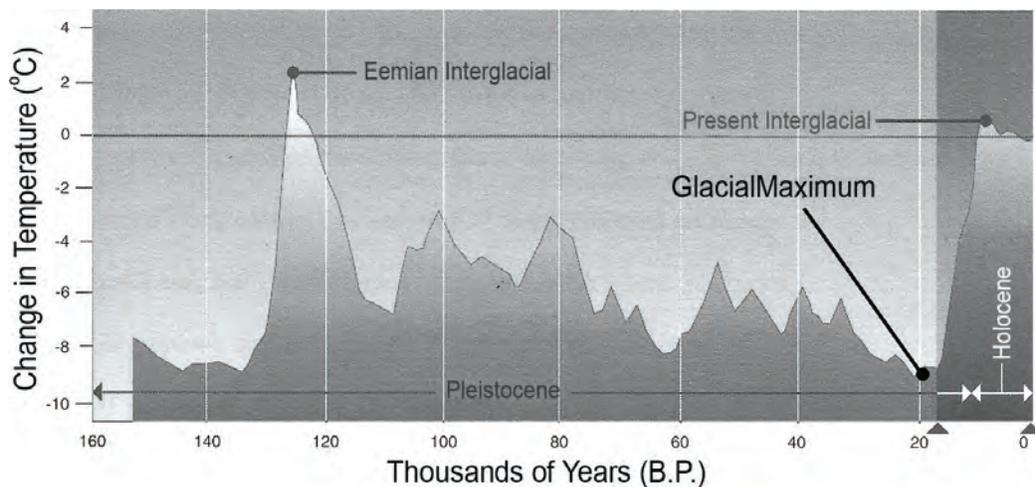
Figure 3-2—Air temperature changes occurring over the Antarctic over the last 160,000 years. Highlighted are two interglacial periods, the present Holocene and warmer Eemian, each are characterized by rapid onsets of warmer temperatures. The coldest period during the late Pleistocene (last 120,000 years) was the Glacial Maximum. Estimates derived from hydrogen/deuterium ratios measured in ice cores (Eddy and Bradley 1991).

How does understanding woodland dynamics and climate change over the past tens of thousands of years help us to understand vegetation management under current and future conditions? Studying past spatial and temporal dynamics of woodlands and shrublands increases understanding about how current and future variations in temperature and precipitation influence woodland and shrubland migration and local extinction. It provides insights into the dynamics between woodlands and sagebrush ecosystems and the primary drivers of woodland expansion, infill, and contraction over the past 200 years. And it provides clues to the potential consequences of climate change in the next 200 years for woodland and shrub communities (sidebar 4).

A variety of evidence helps us reconstruct prehistorical conditions for temperature, precipitation, and vegetation dynamics over time and space. The evidence includes: geomorphic footprints of glaciers and lake shorelines; pollen and volcanic ashes; diverse sediments from lakes and ponds; macro fossils from packrat middens, dry rockshelters and caves; extinct animal remains; changes in the isotopic 18O/16O ratios of ice cores, plants, and animal bones; and changes in the δ13C and deutereum ratios in plants (Wigand et al. 1994a; Wigand 1999). This evidence is used to determine the presence, persistence, and dynamics of plant species at a given location or region and temporal changes in climate. This combination of evidence allows us to reconstruct the general climate and the elevational/latitudinal movement of pinyon and juniper in response to changes over the past 20,000+ years across the Great Basin and Colorado Plateau. It also allows us to examine changes in associated species and the spatial movement of plant species and their varying community combinations through time.

A limitation of this evidence is the difficulty to differentiate pollen among different sagebrush species and subspecies (Peter Wigand, Graduate Faculty, Department of Geography & Research Faculty, Graduate Program of Hydrological Sciences, University of Nevada, Reno, personal communication, 2017), which as a group occurs across a broad range of moistures and temperatures (fig. 3-3). As a result, a number of paleobotany papers simply lump *Artemisia*, which limits moisture and temperature

Sidebar 4
Vegetation Zones
Names of vegetation zones frequently used in the paleobotany literature for the Great Basin and Colorado Plateau are shadscale, desert scrub, lower sagebrush-steppe, woodland, upper sagebrush-steppe, montane forest, and subalpine forest (defined in the Glossary).
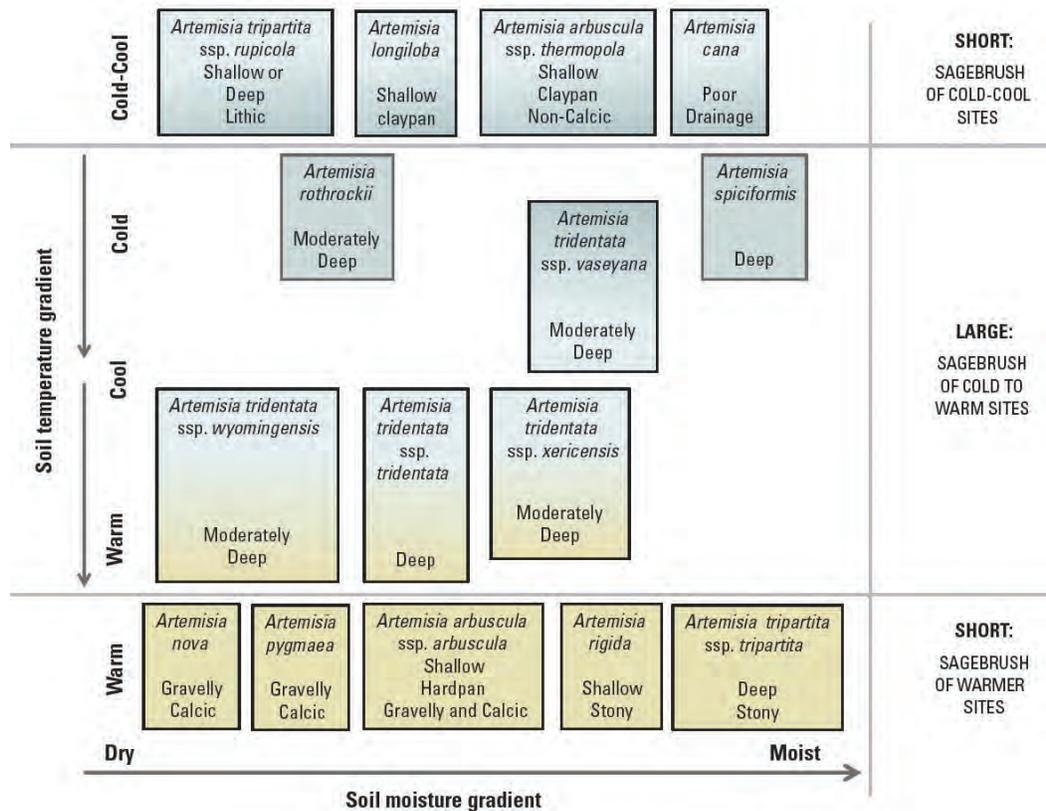
Figure 3-3—Sagebrush taxa in the Great Basin and Columbia Basin positioned along soil temperature and moisture gradients (Pyke et al. 2015). As a group, they occur over a relatively broad moisture and temperature gradient.

interpretations. However, the mention of associate shrub species or elevation placement in relation to the pinyon and juniper woodland zone can help to narrow down the likely *Artemisia* species or subspecies, which allows for a more accurate interpretation of available moisture. Pinyon and juniper are identified to species in the analysis of geologic evidence, making the reconstruction of their history more straightforward.

Late Pleistocene—20,000 to 10,000 Years Ago

Distribution of vegetation across the Great Basin and Colorado Plateau during the late Pleistocene was dramatically different from presettlement and present-day vegetation. The latter portion of the Pleistocene Epoch includes the last interglacial episode and the last major advance of continental glaciers in North America, which began around 120,000 years ago and ended 10,000 to 11,000 years ago (fig. 3-2; Eddy and Bradley 1991). Temperatures and precipitation fluctuated throughout this epoch with the coldest (the Glacial Maximum) and warmest temperatures (Bølling–Allerød period) during the late Pleistocene (fig. 3-1). The warmest and driest conditions occurred after the Pleistocene in the Holocene during the Thermal Maximum (Renssen et al. 2009).

Climate

The coldest period over the past 120,000 years across the West occurred during the Glacial Maximum (approximately 20,000 years ago) (fig. 3-2) persisting with a gradual temperature increase until 14,500 years ago (fig. 3-1; Davis 1987; Eddy and Bradley 1991; Spaulding 1985; Van Devender and Spaulding 1979; Wells 1983; Woolfenden

USDA Forest Service RMRS-GTR-403. 2019.

1996). Lake Bonneville reached a peak elevation level of 5,102 feet approximately 15,500 years ago, covering 20,000 square miles of northwestern Utah at a depth of 900 feet (Patrickson et al. 2010). The greatest elevation level of Lake Lahontan in Nevada around this same period was 4,370 feet (Morrison 1964) covering more than 8,000 square miles of northwestern Nevada and reaching a depth of 400 feet at Fallon, Nevada.

Temperatures during the Glacial Maximum across the Great Basin were estimated to have been 9.9–12.6 °F cooler than current conditions (Dohrenwend 1984) but varied regionally from north to south and east to west (table 3-1). The temperature variations likely represent a combination of regional differences in climate and are possibly a result of the methodologies used. Bevis (1995) concluded differences in summer temperatures between the Glacial Maximum and current conditions were considerably larger in the northwestern than southcentral Great Basin. In the Tushar Mountains of central Utah (latitude 38°22'19.49"), summer temperatures were 7.2 °F colder than today, but on Steens Mountain in southeast Oregon (latitude 42°44'11.49"), summer temperatures were 16.5 °F colder than today (table 3-1). Bevis concluded mean summer temperature depressions in the southern Great Basin were possibly mediated by increased annual precipitation. Toward the end of the late Pleistocene, precipitation estimates were almost 1.3-fold greater than current levels based on vegetation distributions, Pleistocene lake levels, and the lower glacier boundaries (Bevis 1995; Porter et al. 1983). Between 14,000–13,000 years ago temperatures began to increase more rapidly (Bølling -Allerød period) but still remained cooler than present (fig. 3-1).

Table 3-1—Estimated temperature differences between the last glacial maximum and current conditions from northerly to southerly locations across the Great Basin (adapted from Grayson 2011). MAT = Mean Annual Temperature.

Location	Temperature (°F)	Source of estimate	Reference
Steens Mt, southeast OR	-16.5 summer	Glacier formation	Bevis 1995
Pine Forest Range, northwestern NV	-10.8 summer	Glacier formation	Bevis 1995
Lahontan Region, northwestern NV	-14.4 to -16.2 MAT	Vegetation change	Wigand and Rhode 2002
Ruby Marsh, northeastern NV	-12.6 MAT	Glacier formation	Bevis 1995
Northern Bonneville Basin, northwestern UT	-5.6 to -16.2 MAT	Vegetation change	Davis 2002
Northern Bonneville Basin, northwestern UT	-10.8 to -12.6 MAT	Computer modeling	Laabs et al. 2006
Deep Creek Mountains, northwestern UT	-5.6 summer	Glacier formation	Bevis 1995
Northern Uinta Mt, northcentral UT	-9.9 to -14.4 summer	Glacier formation	Munroe and Mickelson 2002
Western Uinta Mt, central UT	-9 to -12.6 MAT	Glacier formation	Refsnider et al. 2008
Tushar Mts, central UT	-7.2 summer	Glacier formation	Bevis 1995
Southern Nevada	-10.8 winter	Vegetation change	Spaulding 1985
Southern Nevada	-10.8 to -12.6 MAT	Vegetation change	Spaulding 1985
Southern Nevada	-12.6 to -14.4 MAT	Vegetation change	Spaulding 1985
Owens Valley, eastcentral CA	-18 MAT	Plant hydrogen isotopes	Jennings and Elliot-Fisk 1993
Death Valley, southeastern CA	-14.4 to -25.2 summer	Vegetation change	Woodcock 1986

In the Colorado Plateau, estimated temperature differences between the Glacial Maximum (14,000–23,000 years ago) and current were 5.4 to 7.2 °F cooler (Anderson et al. 2000), similar to differences estimated for the southern Great Basin. Glaciers do not exist in the Colorado Plateau today, but during the late Pleistocene they occurred on several ranges with the lower edges dropping to as low as 8,600 feet on San Francisco Peak just north of Flagstaff, Arizona (Anderson et al. 2000). Estimates of annual increases in precipitation during the Glacial Maximum were about 1.5 times that of current (Betancourt 1984). However, summer precipitation may have been less during much of the late Pleistocene (Anderson et al. 2000). There appear to be discrepancies in the literature related to amounts of precipitation during the late Pleistocene. But this is likely related to regional differences. Some authors reported conditions were dry and cold (Brakenridge 1978; Galloway 1970, 1983; Thompson and Mead 1982) and others wet and cold (Wells 1979).

Following the Glacial Maximum, temperatures warmed gradually until the beginning of the Bølling-Allerød period (14,700 to 11,000 years ago) when they warmed rapidly (fig. 3-1). This was a relatively moist period—possibly the warmest in the last 10,000 years of the Pleistocene. Immediately following the Bølling-Allerød period was the Younger Drayas cold snap, a 500-year phase, preceding the onset of the Holocene—11,000–10,000 years ago (Carlson 2013; Eddy and Bradley 1991). In the West, temperatures during the Younger Dryas cold snap cooled more than 5.5 °F and conditions were wetter than the Bølling-Allerød period (Carlson 2013; Cole and Arundel 2005). The transition between the Younger Dryas cold snap and the Holocene was very rapid, with temperatures rising as much as 18 °F in 60 years (Grayson 2011).

Vegetation

Pleistocene vegetation responded to these long-term changes in climate through latitudinal migration, changes in elevation and aspect, and extinction. During the late Pleistocene Epoch, vegetation across much of the Intermountain West was dominated by plant species present in the region today (Grayson 2011; Nowak et al. 1994a,b, 2017; Spaulding 1990; Thompson 1990). Sagebrush, snowberry, cinquefoil, saltbush, greasewood, and many grass and forb species common today were present during the Pleistocene. The most abundant and frequently occurring plant group in pollen and macro fossil records is sagebrush (Grayson 2011). However, in the northerly latitudes of their current ranges (with the exception of Utah juniper), pinyon pines and western juniper were absent during the Pleistocene Epoch (Thompson et al. 1986; Wigand and Nowak 1992).

The likely reason many species in the semiarid Intermountain West persisted throughout the Holocene and Pleistocene is the highly variable topography and the mountain ranges oriented predominately north and south. This variability allowed plant species to stay within their moisture and temperature ranges by migrating up or down in elevation and/or across aspects and more easily moving north and south. Over the past 20,000 years, some species shifted as much as 3,000 feet in elevation (fig. 3-4; Spaulding 1990). These shifts appear have been greater in the south than the north, possibly a result of lower elevation permanent snowpacks in the north. During the Glacial Maximum, subalpine woodlands (typically composed of bristlecone, limber or whitebark pine, and occasionally Engelmann spruce; see Glossary for the use of the term subalpine woodlands) were considerably more extensive than they are today. Subalpine woodlands occupied mid- and lower- slopes of the Great Basin and Colorado Plateau where pinyon and juniper woodlands commonly occur today (Spaulding 1985; Thompson 1990; Thompson and Mead 1982; Van Devender and Spaulding 1979; Wells 1983; Wigand et al. 1995; Woolfenden 1996).

Wells (1983) argued that subalpine woodlands even occupied the valley floors. However, Thompson (1990) suggested these Pleistocene woodlands were restricted to coarse-textured soils of the mountain slopes, and expansion into the fine-textured
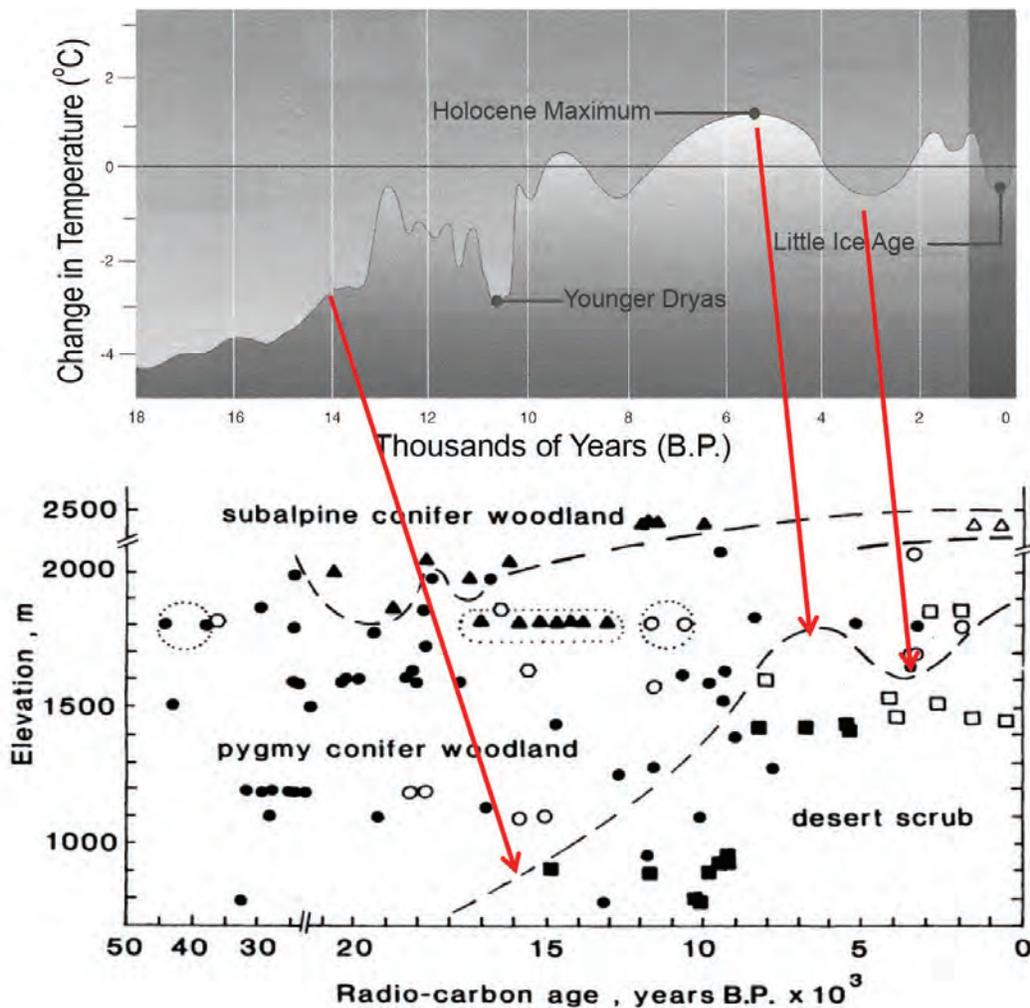
Figure 3-4—Pleistocene and Holocene elevation changes for pinyon, juniper, and subalpine conifer woodlands in the northern Mojave Desert based on woodrat middens. Subalpine woodlands (solid black triangles), subalpine woodlands with abundant Utah juniper (open triangles), Utah juniper (solid black circles), and mixed Utah juniper and pinyon pine (open circles) (modified from Spaulding 1985; Eddy and Bradley 1991).

valley floors was limited. Today, the lower boundary of juniper in the Basin and Range of Nevada and western Utah commonly occur near the toe-slopes where moisture is sufficient, but they also often extend along the concave depressions of the drainages, which reach out into the valley floors. It is clear that white fir (*Abies concolor* (Gord. & Glend.) Lindl. Ex Hildebr.) descended in elevation in the Paranagat Range north of Las Vegas, Nevada, and disappears from record around the Lake Lahontan area around the Glacial Maximum. White fir also remained present at lower elevations in mountains of the Nevada Test Site as late 11,000 years ago (Wigand 2017).

Subalpine woodlands were largely open, with abundant understory communities, often with sagebrush (possibly mountain big sagebrush) as the dominant or subdominant shrub (Thompson 1990). Even today it is not uncommon to observed subalpine woodlands intermingle with high elevation sagebrush communities. Subalpine tree species common in the southern and eastern portions of the Great Basin and Colorado Plateau included bristlecone pine, limber pine, common juniper, Engelmann spruce, and Rocky Mountain juniper. Bristlecone pine was most prominent, based on its relative abundance in material from packrat middens.

In the northwestern Great Basin, common conifer species were whitebark pine, creeping juniper, and common juniper (Wigand et al. 1994b). In the northwestern portion of its range, Utah juniper persisted near its current latitudinal limit at the lower elevations. However, at higher elevations in Utah, its northern limit was where the Green River enters the Colorado River (Anderson et al. 2000), nearly 450 miles south of its current northern boundary. In the Ruby Mountains and Snake Range in central and eastern Nevada and the western shores of Lake Bonneville, Utah juniper was either uncommon or absent (Thompson 1990, 1992).

A major difference in conifer geographical distribution between the Pleistocene and Holocene was the absence of three tree species common today. During the Glacial Maximum, both singleleaf and twoneedle pinyons were located at the very southern edge of their current range (fig. 3-5; Cole et al. 2013). And there is no evidence of western juniper occurring in the Great Basin until the very end of the late Pleistocene (Nowak et al. 1994a; Thompson et al. 1986).

Subregions

Northwestern Great Basin (Klamath and Shasta Valleys and Basins, and Malheur High Plateau; MLRAs 21 and 23)

Late Pleistocene landscapes were predominately shrub dominated by sagebrush intermixed with other shrub species adapted to colder environments. Associated species included creeping and common juniper (Wells 1983; Wigand et al. 1994b) and possibly other cold adapted shrubs such as mountain and/or low sagebrush, snowberry, cinquefoil, currant, and rabbitbrush. There appeared to be a complete lack of junipers with a tree growth form, pinyon or subalpine woodland tree species occurring this far north. The oldest pollen record in the area is from deep sediment cores at Tulelake in northeastern California. Pollen abundance and ratios suggest sagebrush has been present and probably a dominant component of the vegetation for over 1 million years (Woolfenden 1996). Evidence of creeping and common junipers were found in the Owyhee River Valley 27,000 years ago, near the border of Oregon and in Idaho southwest of Boise, Idaho (Wells 1983), and possibly near Summer Lake in eastern Oregon between 12,000 and 50,000 years ago. The presence of these two low-growth form juniper species strongly suggests a colder and wetter climate than current conditions. Depending on soil characteristics, these two conifers were likely associated with mountain big and low sagebrush.

There is no evidence of western juniper occupying its current range during the Pleistocene, nor did it occur farther south in the Great Basin. The earliest evidence of western juniper (12,070 years ago) was found on the northeastern shores of Winnemucca Lake, south of its current range (Thompson et al. 1986; Wigand and Nowak 1992). A possible refugium for western juniper during most of the Pleistocene may have been in the foothills and low mountains surrounding the northcentral valley of California (Peter Wigand, Graduate Faculty, Department of Geography & Research Faculty, University of Nevada, Reno; Robin Tausch, Retired Range Scientist, USDA Forest Service, Rocky Mountain Research Station, Reno, Nevada; and Rick Miller, Professor Emeritus Range Ecology, Oregon State University, Corvallis, Oregon, personal discussion, 1998). Partially supporting this hypothesis is the lack of Utah juniper DNA markers in western juniper in this area today, suggesting a long-term isolation between the two species (Terry et al. 2000). East of the Cascades, Utah juniper DNA markers are commonly found in western juniper.
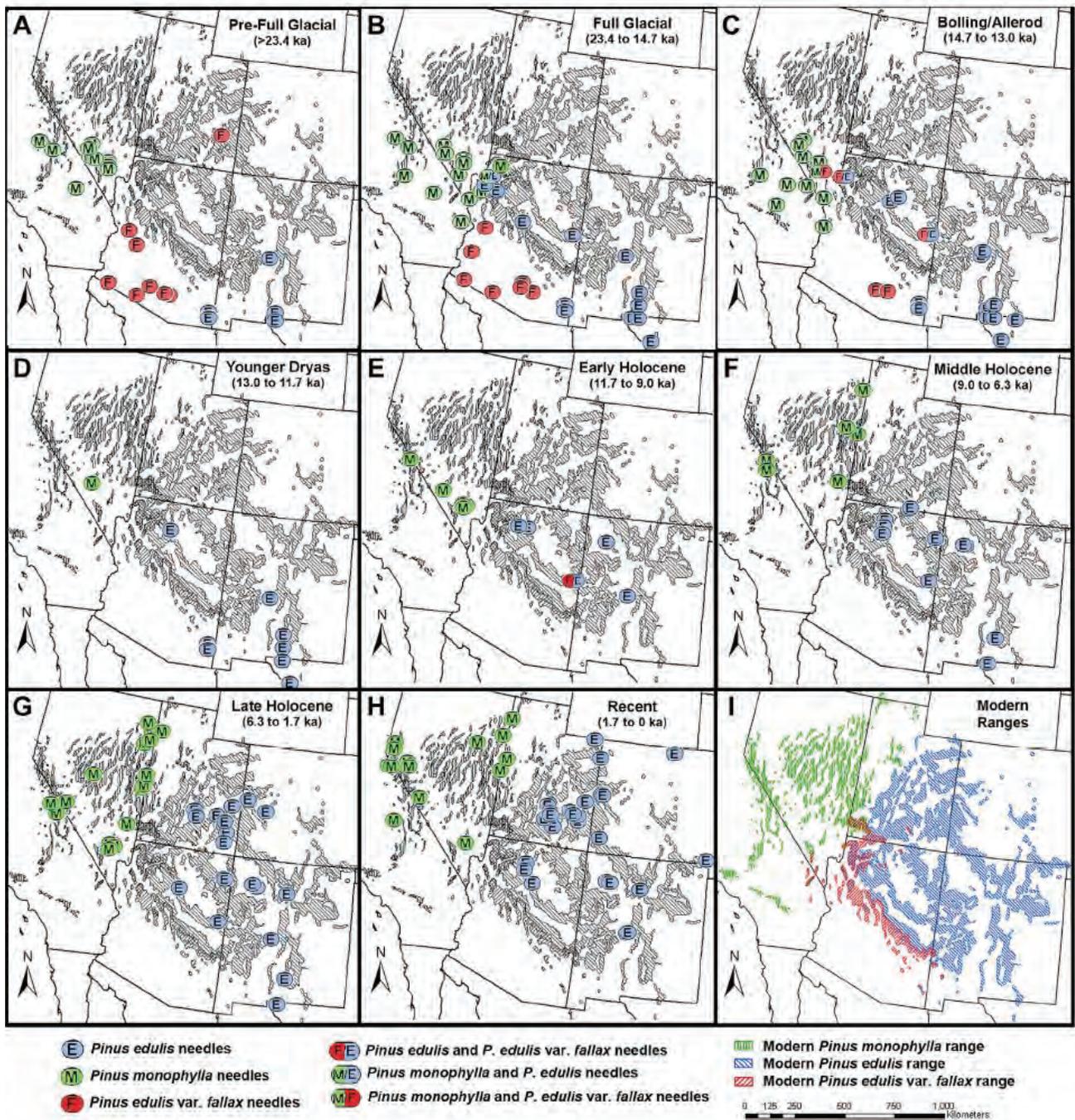
Figure 3-5—Fossil records of pinyon types obtained from packrat middens spanning eight time periods (A-H) as compared to modern ranges (gray background patterns) and (I) modern ranges of two pinyon species and one subspecies (Cole et al. 2013).

West-Central Great Basin (Carson Basin and Mountains, and Fallon-Lovelock; MLRAs 26 and 27)

Sagebrush was present throughout the late Pleistocene in the west-central Great Basin as well (Grayson 2011; Nowak et al. 1994a,b). Under these wetter and cooler conditions, it dominated areas above late Pleistocene lake levels now occupied by *Atriplex* species and greasewood (Nowak et al. 1994a,b). However, unlike the northwestern Great Basin, temperatures appeared to be moderate

enough for trees to occupy this region. Between 30,000–12,500 years ago, the lower elevational treelines of subalpine (primarily whitebark pine with some limber pine) and Utah juniper woodlands were 3,300–3,600 feet lower than today (fig. 3-6; Nowak et al. 1994b; Wigand and Nowak 1992). Semiarid woodlands were composed of Utah juniper with an abundant understory of sagebrush. Pinyon pine was absent and remained so until the late Holocene (Cole et al. 2013; Nowak et al. 1994b). The lower band of juniper woodlands occupied the landscape below 4,250 feet until around 9,500 years ago when they were replaced by semiarid steppe or semiarid shrublands (Wigand and Nowak 1992). At the mid-elevations above 4,250 feet, Utah juniper has changed little in the fossil record (Nowak et al. 1994a,b).

In the Carson Sink area sagebrush was the dominant plant and Utah juniper was present throughout the late Pleistocene—where Atriplex species are common today (Grayson 2011). Limber pine and curlleaf mountain mahogany were also present in the area, growing more than 3,000 feet below their current elevation range. The northeast end of Winnemucca Lake may have been one of the early arrival points of western juniper into the Great Basin 12,500 years ago (Thompson et al. 1986). Cool season grasses common in the area today were also present throughout the late Pleistocene as were sagebrush and mountain mahogany (Nowak et al. 1994b). Whitebark pine was also abundant at lower elevations close to the Pleistocene high stand of Pluvial Lake Lahontan in the Virginia Mountains from about 22,000–11,000 years ago, where it was mixed with Utah juniper and mountain mahogany (table 3-2; Wigand and Nowak 1992).

Figure 3-6—The range of Utah juniper has shifted little in latitude over the past 30,000 years in the westcentral part of the Great Basin as compared to the central and eastern Great Basin. However, it has moved up and down in elevation by more than 3,000 feet in response to climate in the western Great Basin. Relict woodland in the Virginia Mountains, Nevada. (Photo by Robin Tausch, USDA Forest Service, Rocky Mountain Research Station, Reno, Nevada.)

Table 3-2—Estimated upper elevational boundaries for woodlands in the southern portion of the Great Basin (Mojave Desert) during the late Pleistocene to present day (derived from Anderson et al. 2000; Spaulding 1985, 1990; USDA-NRCS plant guide).

Woodland type	Elevation (ft)	Years ago
Pygmy woodland	4,560	18,000 glacial max
	5,248	16,000
Singleleaf pinyon pure stands	< 9,200	Present
Utah juniper	1,200–3,300 (6,600)	18,000 glacial max
	2,500–3,000	13,200–11,700
	4,260–6,600	9,000
	5,900–7,200	Present
Temperate desert scrub	3,900–5,900	Present
Hot desert scrub	< 3,900	Present

Great Salt Lake and Central Nevada Basin and Range (MLRAs 28a, b)

Subalpine woodlands were an important component in the Snake, Wah Wah, and Confusion ranges of the eastern Great Basin during the late Pleistocene (Thompson 1984, 1990; Wells 1983). They were a common component of vegetation on the lower mountain slopes, 2,200 to 2,750 feet lower than today (Thompson 1984). Subalpine woodlands may have extended across the valley floors—based on the elevation of material found in a midden located in a wash (Wells 1983). But Thompson (1984) argued subalpine woodlands were largely restricted to the mountain slopes growing on the coarse-textured soils. Wigan reported bristlecone pine was located just above the valley floor in the Egan Range of eastern Nevada (Peter Wigand, Graduate Faculty, Department of Geography & Research Faculty, University of Nevada, Reno, personal communication, 2019).

Subalpine woodlands formed a continuous matrix with sagebrush ecosystems throughout this region (Thompson 1990) and were largely composed of bristlecone pine, limber pine, common juniper, and—in the Snake Range—Engelmann spruce. Rocky Mountain juniper first appears in the midden samples right after the Glacial Maximum, around 17,000 years ago. Utah juniper was present in the early Wisconsin Glacial Period, 30,000+ years ago—however, it appears to be absent during full glaciation, not returning until around 14,000 years ago and then becoming relatively abundant during the Holocene (Thompson 1990). Cold temperatures rather than precipitation possibly were the primary factor limiting the distribution of Utah juniper during the late Pleistocene.

Farther north, between 30,000–10,000 years ago, woodland species were uncommon or absent across much of the northern portion of the central Great Basin (i.e., the Ruby Mountains and Ruby Marsh) (Thompson and Mead 1982). Sagebrush species were common, and only limited pine (no species named) occurred in the area. However, just to the east on the western shores of Lake Bonneville, Engelmann spruce and limber pine were common between 17,000–14,000 years ago. A combination of lower elevations and possible lake-effect on climate may have resulted in more moderate temperatures, allowing conifers to survive. Prior to the Glacial Maximum more than 40,000 years ago, Utah juniper was present and associated with sagebrush, snowberry, and horsebrush in the Goshute Mountains, just west of Lake Bonneville and 30 miles to the east of the Ruby Mountains. But Utah juniper generally declined in the northern portions of this region during the onset of the Wisconsin period and then disappeared—reappearing around 8,000 years ago (Grayson 2011). Mountain shrub, composed of sagebrush and snowberry, were common on the lake shores, becoming dominant 14,000–13,000 years ago, and suggesting significantly greater moisture than today (Grayson 2011; Thompson 1990).

Southern Nevada Basin and Range and High Mountains of the Mojave (MLRA 29, 30)

During the Glacial Maximum, the midden record indicates Utah juniper woodlands were the dominant vegetation type at elevations now occupied by desert shrubs (Jennings 1988, 1995). This is about 2,000 feet lower than where pinyon and juniper woodlands are found today (Jennings and Elliot-Fisk 1993). Desert shrub communities were found only east of the double rain shadow created by the Sierra Nevada and the White and Inyo mountains. Although Utah juniper was the most abundant tree in semiarid woodlands, pinyon did occur infrequently in the upper portions of these woodlands between 3,300 and 5,900 feet, converting to subalpine woodland at the higher elevations (fig. 3-7a,b; Spaulding 1983, 1985; Wells 1983).

During the Glacial Maximum, Utah juniper was growing down to a 1,200 feet elevation (fig. 3-4; Spaulding 1990). Although Utah juniper was typically the dominant tree (Grayson 2011), the southern Nevada Basin and Range formed the northernmost distribution of singleleaf and twoneedle pinyon pines (fig. 3-5; Cole 1990; Cole et al. 2013; Spaulding 1990). Singleleaf pinyon was growing, at times, at the southern end of the Owens Valley, on the east slopes of the Sierras, and the Sheep and Spring mountains in the Mojave near Las Vegas, Nevada (Cole et al. 2013). In the Mohave, Utah juniper and pinyon pine occupied mountain ranges that no longer support conifers today. Examples are the Hidden Hills north of Las Vegas and the Tinajas Atlas Mountains just southeast of Yuma (Davis 1987; Wells and Jorgensen 1964). In the Owens Valley near the Owens Lake shoreline, middens dating back more than 30,000 years showed that Utah juniper occupied the area in the late Pleistocene (Koehler and Anderson 1994, 1995) and both pinyon and juniper 11,450 years before present (Peter Wigand, Graduate Faculty, Department of Geography & Research Faculty, University of Nevada, Reno, 2019). However, by 9,000 years ago it disappeared from the valley floor moving upslope—from less than 4,500 feet to over 6,200 feet. Rocky Mountain juniper was also found in the Owens Valley 17,500–16,000 years ago (Koehler and Anderson 1994). It is no longer found in California and the nearest population today is in the Spring Range near Las Vegas, Nevada, 140 miles to the east (Charlet 1996).

Temperatures gradually increased following the peak of Glacial Maximum with dramatic increases in temperatures occurring during the onset of the Bølling-Allerød period, 14,700–13,000 years ago (fig. 3-1). Pinyon and juniper responded by migrating upward in elevation, replacing the subalpine species and forming semiarid woodlands (table 3-2; Spaulding 1990). Toward the end of the late Pleistocene, 10,000–12,000 years ago, vegetation mosaics of woodland on the northerly and desert scrub on the southerly aspects occurred below 3,300 feet (Spaulding 1990; Wells and Woodcock 1985). Today, these areas are occupied by desert scrub while pinyon and juniper woodlands occupy areas that had been covered with subalpine woodlands made up of limber and bristle cone pines and understories of sagebrush (Spaulding 1990).

Colorado Plateau (MLRA 35)

During the Glacial Maximum, plant species and vegetation zones were growing 2,000–3,300 feet lower than today (Anderson et al. 2000; Coats et al. 2008; Cole 1985). The timberline dropped to around 8,500 feet (today it is at more than 11,000 feet) and subalpine tree species, largely Engelmann spruce and subalpine fir, grew above 7,200 feet (today they grow at 10,000–11,500 feet). The mixed conifer forest, largely composed of limber pine, Douglas-fir, white fir, Rocky Mountain juniper, and occasionally blue spruce, was probably the most extensive forest in the Colorado Plateau during full glaciation. These forests were commonly found between 5,200 and 6,900 feet (Anderson et al. 2000) but in some areas extended down to 4,500 feet (Cole 1985) and occupied much of the area that is currently occupied by ponderosa pine. Mixed conifer forest was typically open with associated understory species including big sagebrush, rabbitbrush,

Figure 3-7—During glacial maximum, Utah juniper dominated the semiarid woodlands between 3,300 and 5,900 feet on the White Mountains of California. Pinyon was a minor component during the late Pleistocene. Today, pinyon and juniper woodlands occur between (A) 6,500–9,500 feet where bristlecone and limber pine occurred 20,000 years ago. At near 11,000 feet, the limber and bristlecone pines intermingle with mountain big sagebrush communities. (B) During glacial maximum, these bristlecone and limber pine woodlands grew 3,000 feet lower where pinyon and juniper woodlands now occur. White Mountains, California. (Photo by Rick Miller, Oregon State University.)

rose, and skunkbush sumac. Juniper woodlands often dominated the lower elevations below the mixed conifer forests. The juniper-shadscale-sagebrush zone in the western Grand Canyon occurred at elevations from 4,800–6,200 feet (fig. 3-8; Cole 1985) and in the eastern Grand Canyon at elevations up to 4,750 feet (Anderson et al. 2000). In the central portion of the plateau near Canyonlands, juniper woodlands present before the Glacial Maximum were replaced by mixed conifer during the Glacial Maximum (Anderson et al. 2000). The mixed conifer woodlands were associated with an understory of big sagebrush, rubber rabbitbrush, and species in the rose family. As conditions warmed following the Glacial Maximum, Utah juniper and serviceberry replaced the mixed conifer woodlands by the Bølling -Allerød period, 12,550–11,000 years ago.

Juniper woodlands were largely composed of Utah juniper, mixing with oneseed juniper in areas where summer precipitation was more than 30 percent of the total, and Rocky Mountain juniper occupied the cooler and moister sites (fig 2-4a). Twoneedle pinyon pine was absent across most of the northern Colorado Plateau during the late Pleistocene. Its northern limit occurred along the Mogollon Rim in northwest Arizona extending into central New Mexico then south to western-most Texas (fig. 3-5b; Cole et al. 2013). There is only one record of twoneedle pinyon occurring in the Colorado Plateau prior to the Holocene in the Wupatki National Monument, just north of Flagstaff, Arizona, 16,300–12,800 years ago (Anderson et al. 2000). There is no indication of northward movement during the late Pleistocene.

Figure 3-8—The juniper-shadscale-sagebrush zone in the western Grand Canyon occurred at elevations up to 6,200 feet and in the eastern Grand Canyon 4,750 feet during glacial maximum, below where they occur today. Grand Canyon, Arizona. (Photo by Rick Miller, Oregon State University.)

Although there were some similarities in conifer migration in the Colorado Plateau and Great Basin during the colder and wetter climate conditions of the late Pleistocene, there were also some distinct differences. Unlike the northwest Great Basin, where Utah juniper appeared to stay in place latitudinally (fig. 3-6; Nowak et al. 1994a,b), its northern boundary in the Colorado Plateau was over 400 miles south of the northeast edge of its range today. Utah juniper did persist throughout the late Pleistocene across portions of the Colorado Plateau (Coats et al. 2008). Like the Great Basin, sagebrush in the Colorado Plateau has been present in the area well over 50,000 years, and it frequently was a codominant species in the understory of both mixed conifer and juniper woodlands. However, conifer species were more diverse and abundant in the Colorado Plateau than the northwestern and west-central Great Basin. In both the southeastern Great Basin and Colorado Plateau, subalpine woodlands were probably the most important woodland and were composed of limber pine and bristlecone. But in the Colorado Plateau, Engelmann spruce and subalpine fir also became more common in both the subalpine and in mixed conifer at lower elevations.

Holocene 10,000 Years to A.D. 1800s

This section focuses on the distribution shifts of pinyon and juniper woodlands and associate vegetation zones between the end of the late Pleistocene and before Eurasian settlement. Later in this section we will look at the combined impacts of climate and anthropogenic disturbances on woodlands over the past 200 years. The beginning of the Holocene is frequently debated but generally accepted to be around 10,000 years ago (Grayson 2011; Spaulding 1990) immediately following the Younger Dryas cold snap (fig. 3-1). This was a period of warm and dry conditions as Pleistocene lakes and glaciers retreated across the Great Basin and Colorado Plateau. This was also the beginning of relatively rapid elevational and latitudinal migrations of plant species (Cole 1985; Grayson 2011; Wigand et al. 1995) and local extinctions (Nowak et al. 2017). Although the Holocene marked the beginning of warmer and drier conditions, moisture and temperature varied widely across the region in both time and space throughout the period (Eddy and Bradley 1991). Long-term shifts in climate distinguish the early, mid, and late Holocene periods, all of which had significant impacts on local plant species distributions.

Climate in Holocene Period

Climate: Early Holocene 10,000–7,500 years ago

The early Holocene was considerably warmer than the late Pleistocene but was generally cooler and wetter than the mid and late Holocene (fig. 3-1; Antevs 1948; Bedwell 1973; Grayson 2011; Wigand 2017; Wigand et al. 1994a,b, 1995, 2017). Pluvial lake levels dropped—but there were many shallow lakes and marshes that today are smaller, shallower, or dry (Bedwell 1973; Benson et al. 1990; Grayson 2011; Morrison 1964). The highest water levels in Lake Lahontan during the Holocene Epoch occurred between 10,000–8,000 years ago (Morrison 1964). But changes in lake levels were not synchronous across the Great Basin, indicating geographic variations in temperature and/or moisture conditions. The Warner Mountains and Chewaucan Basin in the northwestern Great Basin were drier in the very early part of the early Holocene (approximately 9,000 years ago) (Minckley et al. 2007) as were the Sierra Nevada Mountains (Grayson 2011). Eventually the northwest and western Great Basin became cooler and wetter, as reflected by increased lake levels. Native American populations were at their highest densities throughout the Great Basin during the early Holocene, based on evidence of the frequency of high cave occupancy across the Great Basin, and abundance of middens at cave sites. This is likely a result of relatively

moister conditions leading to more abundant resources (Aikens 1993; Bedwell 1973). Although there are no Pleistocene lakes in the Colorado Plateau, the presence and absence of various plant species at different elevations indicate this region was also generally wetter and cooler during the early Holocene than today (Cole 1990).

Climate: Mid Holocene (Holocene Maximum) 7,500–5,000 years ago

The mid Holocene, which began around 7,500 years ago and persisted for 2,500 to 3,000 years, represents the driest and warmest period in the past 10,000 years (fig. 3-1; Jennings and Elliot-Fisk 1993; Wigand et al. 1995; Woolfenden 1996). In the northern Great Basin, there was a decrease in both summer and spring precipitation around 5,500 years ago (Wigand 2017). Lake levels dropped below current levels and, in many instances, became totally desiccated (Allison 1982; Antevs 1938, 1948; Hansen 1947; Morrison 1964; Reveal 1979; Thompson 1990).

The transition from the early Holocene to mid Holocene and warmer, drier conditions was not abrupt—and there was wide regional variation in timing and rates of contracting lakes levels across the Great Basin. However, between 7,000–6,000 years ago, the entire Great Basin and Colorado Plateau were dry (Cole 1990; Grayson 2011; Wigand 2017) and 2–3 °F warmer than at present (Eddy and Bradley 1991). This shift in climate resulted in a rapid change in the spatial distribution of plant species (Cole 1990). Plant communities moved upward in elevation to adjust to the warmer and drier conditions (Grayson 2011). In southeastern Oregon, fossil evidence from human-inhabited caves indicated diets and fuel wood shifted from plants and mammals occupying moist habitats—i.e,. waterfowl, aquatic plants, and ponderosa pine—to animals and plants occupying dry habitats—jackrabbits and sagebrush (Aikens 1993; Bedwell 1973). Many caves in the driest regions of the Great Basin were abandoned and human populations across the Great Basin declined as well (Bedwell 1973; Morrison 1964).

Climate: Late Holocene 5,000 years Ago to 1850

The mid Holocene drought (Holocene Maximum) ended and the moister late Holocene began around 5,500–4,500 years ago. Conditions initially remained warm, but summer and winter precipitation increased (fig. 3-1; Davis 1982; Grayson 2011; Wigand et al. 1995). Although the late Holocene was generally cooler and wetter than the mid Holocene, variations in temperatures and precipitation continued, resulting in elevational and latitudinal shifts in vegetation. The late Holocene is frequently separated into different periods, which characterize general climate trends (table 3-3). During the Neoglacial period, climate conditions across the Great Basin began to cool and moisture increased, especially in the northern Great Basin (fig. 3-1; Davis 1982; Grayson 2011; Wigand 2017; Woolfenden 1996). The Great Salt Lake reached its highest level 3,400 and 2,000 years ago since the early Holocene 21 feet above its present-day average (Grayson 2011).

The timing of transition from the Neoglacial to the drier and warmer post-Neoglacial period (2,500 to 2,000 years ago) varied regionally across the Great Basin and Colorado Plateau—but increasingly arid conditions became widespread by 2,000 years ago (fig. 3-1; Chatters and Hoover 1992; Eddy and Bradley 1991; Miller et al. 2001). Data indicate that climate became increasingly variable after 2,600 years ago and that episodes of drought occurred in some areas in the Great Basin, while other areas were wetter than current conditions. Water levels in the Carson Sink in Nevada, Diamond Pond in southeast Oregon, and Pahranagat Lake in southern Nevada were higher 2,300 years and 2,000 years ago than today (Grayson 2011; Wigand 1987; Wigand et al. 1995).

But evidence suggests the middle reaches of the Humboldt River were dry 2,400 years ago (Miller et al. 2001) and chenopod pollen (species adapted to arid conditions) increased 2,500 to 2,000 years ago in the Pyramid Lake area (Mensing et al. 2004).

Table 3-3—General climate conditions during different periods in the late Holocene (derived from Grayson 2011; Wigand et al. 1995). Numbers in parentheses are other reported estimates, typically varying with geographic region.

Period	Time	Climate
Early-Late Holocene	5,500–4,000 years ago	Drought ends but stays warm
Neoglacial	4,000–2,500 (2,000) years ago	Wet and cool
Late Holocene Dry	2,500 (2,000) to 1,300 years ago	Dry and warming
Medieval Warm	A.D. 900–1350	Dry and warm
Little Ice Age	1350–1850	Cooled off with variable moisture availability
Early postsettlement	1850–1920	Generally wet
Late postsettlement	1920–present	Generally warmer and drier
Temperate desert scrub	3,900–5,900	Present
Hot desert scrub	< 3,900	Present
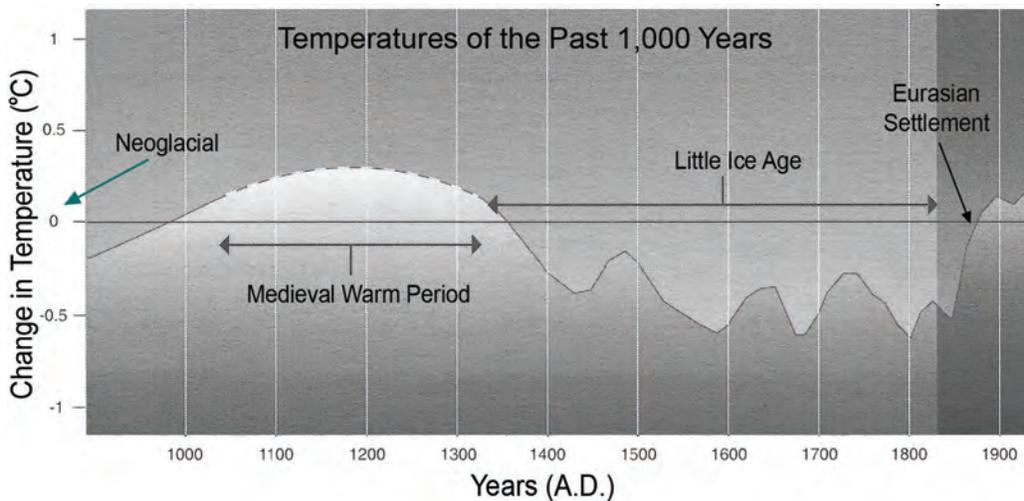
Figure 3-9—Global surface air temperature changes from A.D. 1000–1900. Highlighted is a possible protracted warming through the Medieval period followed by much colder conditions, loosely termed the Little Ice Age. Temperatures estimated from a variety of sources including tree rings and written records (Eddy and Bradley 1991).

Wigand and Rhode (2002) suggested an increase in summer precipitation about 2,000 years to 1,900 years ago may have favored the expansion of pinyon pine in the western Great Basin. Increasing temperatures and continued dry conditions characterized the shift to the Medieval Warm Period between 900 and 1350 CE (common era). Although this period was relatively warm and dry, it was not as severe or as long in duration as the mid Holocene (figs. 3-1, 3-9; Eddy and Bradley 1991; Grayson 2011). The 4 most severe droughts in the past 1,200 years occurred during the Medieval Warm Period and were drier than any 20th century droughts (Cook et al. 2004).

Tree-ring studies in the southern Sierras also revealed temperatures exceeded those of the late 20th century (Graumlich et al. 1995). The Medieval Warm Period ended abruptly around 600 years ago followed by the Little Ice Age. The Little Ice Age was a relatively cooler and initially wetter period than current conditions. Glaciers expanded in the Sierras and did not retreat until the early 1900s (fig. 3-9; Eddy and Bradley 1991; Graumlich 1993; Grayson 2011; Stine 1996). Upper treelines were at their lowest since the end of the early Holocene (Stine 1996). The first half of this period was generally

cool and moist while the second half was cool and dry (LaMarche 1974; Grissino-Mayer and Swetnam 2000; Wigand et al. 1995). Researchers attribute the cool temperatures during this period to 16 different volcanic eruptions between 1630 and 1850, including the Tambora eruption in 1815, which resulted in the year without summer in 1816 (Bradley and Jones 1993).

Vegetation in Holocene Period

Vegetation: Early Holocene 10,000–7,500 Years Ago

Just prior to the onset of the early Holocene, pinyon pines retreated south during the Younger Dryas cold snap—evidence that the cold temperatures restricted movement northward (Cole et al. 2013; Peter Wigand, Graduate Faculty, Department of Geography & Research Faculty, Nevada, Reno, 2018). During the initial portion of the early Holocene, both pinyon species resumed their northern migration.

In the western portion of the Great Basin, the northern boundary of singleleaf pinyon was the south end of the Inyo Mountains (fig. 3-5e; Cole et al. 2013; Grayson 2011). By 9,000 years ago, singleleaf pinyon entered the area of the White Mountains but remained a minor component of the tree canopy, which was dominated by Utah juniper (fig. 3-10; Jennings 1988, 1995). Today pinyon is the dominant woodland tree. In the central Great Basin, singleleaf pinyon migrated rapidly north along the Nevada and Utah border during the transition from the early to mid Holocene (fig. 3-F) (Cole 1985; Cole et al. 2013). Western juniper first appeared in the Great Basin around 12,070 years ago on the shrinking northwest shores of Winnemucca Lake just north of Reno, Nevada (Nowak et al. 1994a; Thompson et al. 1986). This is well south of its current range, where it remained absent throughout the early Holocene. In the Colorado Plateau, twoneedle pinyon's northern boundary remained south of the Utah and Colorado borders as late at 6,300 years ago (fig. 3-5e, f; Anderson et al. 2000; Cole et al. 2013). Relatively cooler temperatures during the early Holocene likely limited the northward migration of both pinyon species (Grayson 2011; Nowak et al. 1994a).

Vegetation: Mid Holocene 7,500–4,500 Years Ago

The driest and warmest period of the mid Holocene resulted in the upward movement of pinyon and juniper woodlands by as much as 1,500 feet in the Great Basin and Colorado Plateau (fig. 3-1; Cook et al. 2004; Grayson 2011; Jennings and Elliot-Fisk 1993; LaMarche 1973; Wigand et al. 1995; Wigand and Nowak 1992; Woolfenden 1996). In the northwestern Great Basin, Utah juniper retreated upslope but remained near its current latitudinal range. Sagebrush replaced Utah juniper at its lower boundary (Wigand et al. 1994b). However, in the Colorado Plateau and central Great Basin, Utah juniper, which occurred 400 miles south of its current range during the Glacial Maximum, moved north during the mid Holocene (Anderson et al. 2000; Lyford et al. 2003).

There was very limited northward migration of singleleaf pinyon along the east slopes of the Sierras that appeared to move and stop, responding to variable climate (fig. 3-10; Wigand 2017). But migration north continued rapidly into the central Great Basin along the Nevada and Utah border (fig. 3-5e, f; Cole et al. 2013; Grayson 2011; Nowak et al. 1994a). By 6,800 years ago, singleleaf pinyon moved into the Schell Creek and Pequop Mountains but remained absent 30 miles west in the Ruby Mountains. Movement of singleleaf pinyon northward along the Nevada and Utah border may have resulted from milder winter temperatures. Avoidance of more arid conditions during this period may have led to migration northward along higher elevations. Several researchers have reported that cold winter conditions likely limited movement of pinyon species northward (Grayson 2011; Nowak et al. 1994a; Peter Wigand, Graduate Faculty, Department of Geography & Research Faculty, University of Nevada, Reno, 2018).
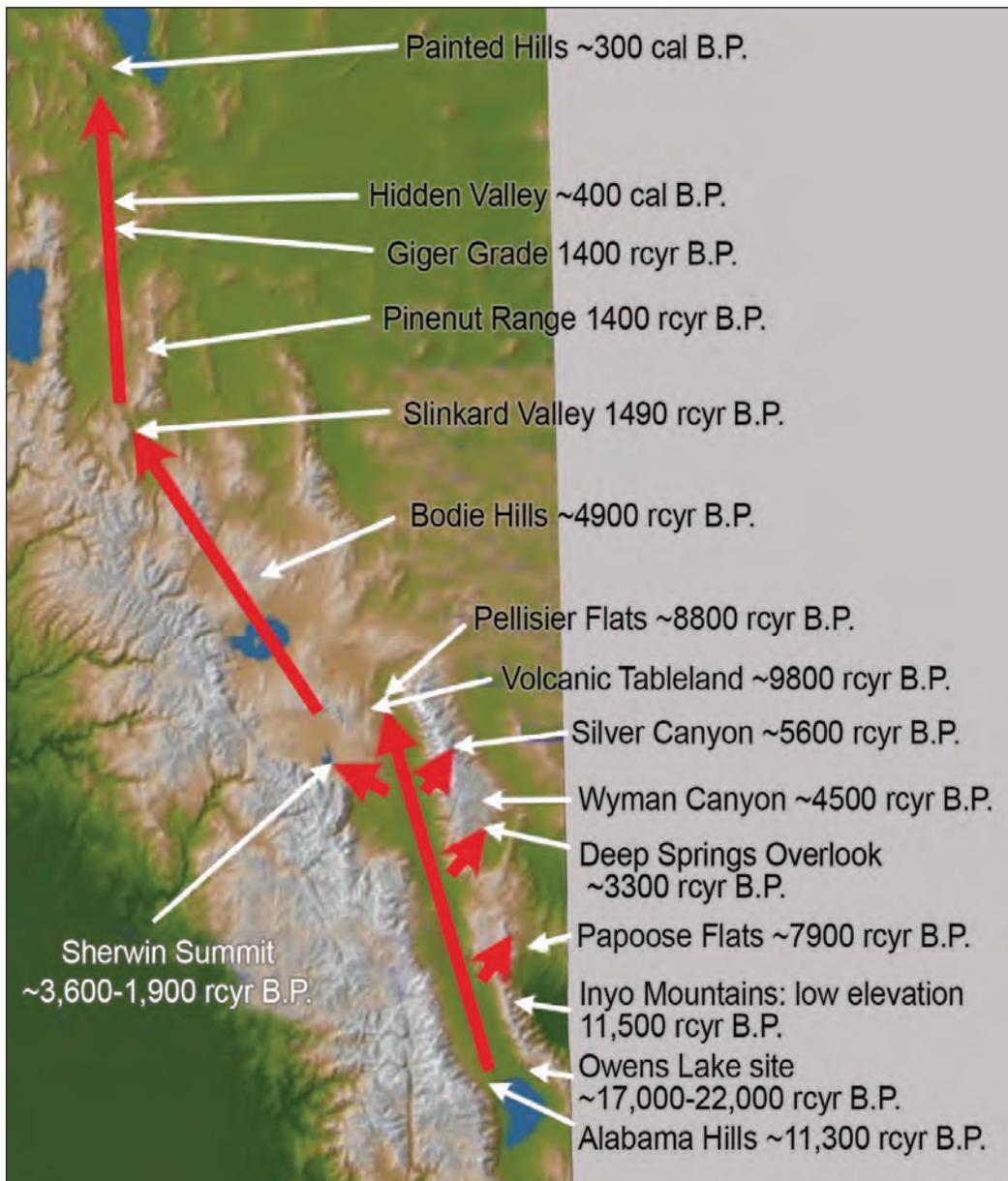
Figure 3-10—Northern movement of singleleaf pinyon along the east slopes of the Sierra Nevada range during the late Pleistocene and Holocene as a result of temperature changes. Note that evidence of the arrival of pinyon in this portion of the Great Basin does not show up until 4,900 years ago in the Bodie Hills near Mono Lake, and only arrives near Pyramid Lake 300 years ago (derived from Wigand 2017).

The northward expansion of singleleaf pine was considerably slower along the east slopes of the Sierras than the migration of singleleaf pinyon in eastern Nevada during this period, remaining south of the Utah and Colorado borders. Migration north along the Sierras was probably limited by variable climate conditions, particularly during dry cold periods. (Peter Wigand, Graduate Faculty, Department of Geography & Research Faculty, Graduate Program of Hydrological Sciences, University of Nevada, Reno, personal communication, 2019).

Vegetation: Late Holocene 5,000 Years Ago to 1850

The late Holocene was generally cooler and wetter than the mid Holocene, but there were several long-term cool and moist, warm and dry (table 3-3; fig. 3-1), and cool and dry periods, which caused vegetation movement (Cole et al. 2013; Grayson 2011; Nowak et al. 1994a,b). Woodlands reached their late Holocene presettlement maximum in expanse and abundance during the cool and wet Neoglacial period—2,000–4,000 years ago (Wigand et al. 1995; Wigand and Nowak 1992). Climate conditions during this period favored tree growth, cone production, and seedling establishment (Johnsen 1962; Fritts and Ziangdig 1986; Redmond et al. 2012; Smith et al. 1975). The shift toward increasing dominance of pinyon over juniper likely began during the end of the Neoglacial with warming temperatures. The Medieval Warm Period—A.D. 900 to A.D. 1300— (figs. 3-1, 3-9) followed the Neoglacial and included the 4 driest periods in the past 1,300 years (Cook et al. 2004). It resulted in a significant decline in areas occupied by pinyon and juniper during the Neoglacial (Davis 1982; Holmes et al. 1886; Stine 1990; Wigand 1987; Wigand and Nowak 1992; Wigand and Rose 1990).

However, both woodrat middens and the pollen record from Lead Lake in the Carson Sink indicate that pinyon pine increased markedly with respect to juniper during this period (Wigand and Rhode 2002). This corresponds with a drop in elevational distribution and northward latitudinal expansion of pinyon in woodrat middens of the area, suggesting the increase in summer precipitation may have been a more significant factor in the expansion of pinyon at that time. The combination of severe drought and increased fire, based on charcoal evidence, often occurred during these transition periods from wet to dry. The transition from cool and moist to warm and dry during the late Holocene coincided with dramatic reductions in juniper pollen in the north (Wigand 1987) and both juniper and pinyon pollen in the south (Wigand and Rose 1990).

The beginning of the Little Ice Age, around A.D. 1300, marks the onset of the re-expansion of pinyon and juniper woodlands and the continued shift toward the increasing proportion of pinyon over juniper (Gray et al. 2006; Wigand and Nowak 1992). But woodlands declined in the second half of the 1500s during a megadrought which influenced much of western North America (Emily Heyerdahl Dendrochronologist, USDA Forest Service, Rocky Mountain Research Station, Missoula, Montana, personal communication, 2014; Swetnam and Betancourt 1998). Expansion and infill (see Glossary) of woodlands resumed following the megadrought and continued during the 1600s through the mid-1800s (Floyd et al. 2017; Miller et al. 2008; Soulé and Knapp 2000; Waichler et al. 2001; Emily Heyerdahl, Dendrochronologist, USDA Forest Service, Rocky Mountain Research Station, Missoula, Montana, personal communication, 2016). In areas where pinyon and juniper grew in mixed stands, increases were greatest for pinyon during the past few centuries (Biondi and Bradley 2013; Despain and Mosley 1990; Wigand et al. 1995).

Both singleleaf and twoneedle pinyons continued to moved northward during the late Holocene, approaching their current northern distributions during the last hundred years. By the late Holocene, between 6,300–1,700 years ago, singleleaf pinyon approached its northern boundary in the central Great Basin along the Nevada and Utah border, but remained absent north of Mono Lake on the east slope of the Sierras and in central Nevada (figs. 3-5g, 3-10; Cole et al. 2013). The earliest evidence of singleleaf pinyon entering the western Great Basin was 4,900 years ago in Bodie Hills, just north of Mono Lake, California. Its continued progression north stalled until around 1,500 years ago (Nowak et al. 1994a), arriving at its present-day northwestern boundary near the southwest shores of Pyramid Lake about 300 years ago. Singleleaf pinyon finally appeared in central Nevada less than 2,000 years ago (Cole et al. 2013). Twoneedle pinyon moved northward into Utah, Colorado, and Wyoming between 6,300 and 1,700
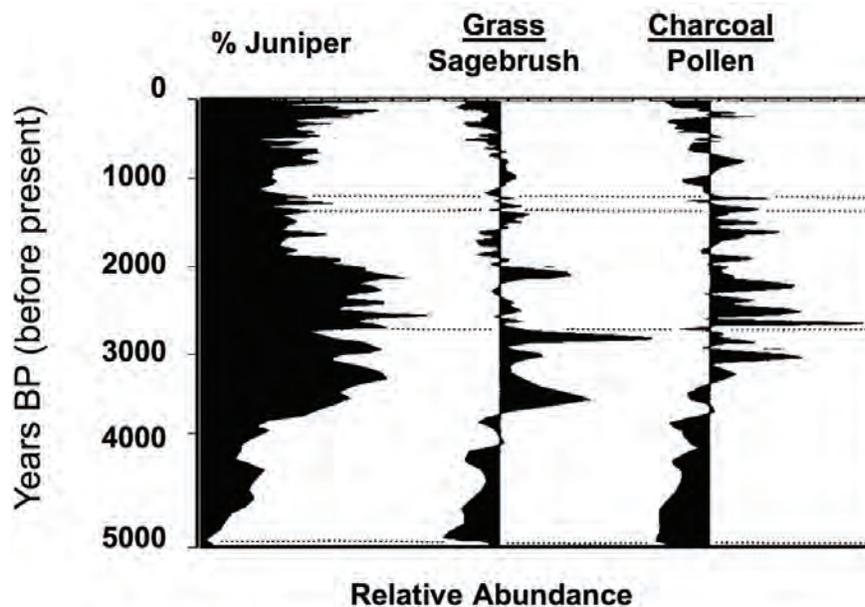
Figure 3-11—Measurements from sediment cores collected at Diamond Craters in southeastern Oregon include juniper pollen percentages, ratios of grass to sagebrush pollen, and ratios of juniper pollen to charcoal. Shifts to the right represent increasing relative abundance of juniper, grass pollen in relation to sagebrush pollen, and charcoal in relation to juniper pollen brought about by increased incidence/frequency of fire (Mehringer and Wigand 1990).

years ago (Cole et al. 2013). It arrived at its current northern boundary in the Dutch John Mountains of northeastern Utah near the Wyoming border about 500 years ago (Gray et al. 2006). Its movement northward stalled during the Medieval Warm Period around A.D. 1250 but rapidly resumed during the Little Ice Age. The current northern geographic boundaries of both singleleaf and twoneedle pinyons are thought to be related to late winter and early spring temperatures (Nowak et al. 1994a; West et al. 1978). Rapid temperature fluctuations during late winter and early spring in this region can cause dormancy to break early in pinyon, making them susceptible to frost damage.

Western juniper first arrived in its current geographic range in central and southeastern Oregon (Bedwell 1973; Wigand 1987) and northeastern California (Mehringer and Wigand 1984) during the mid Holocene, between 4,800–6,600 years ago. It rapidly expanded during the cool and moist Neoglacial period between 4,000–3,000 years ago, reaching its prehistoric maximum and extending across most of its present-day range (fig. 3-11; Mehringer and Wigand 1990; Wigand et al. 1995). However, severe droughts and major fires during the transition between the post-Neoglacial and Medieval Warm periods 2,500–2,000 years ago resulted in significant regional declines in western juniper (Mehringer and Wigand 1990; Wigand et al. 1995). As conditions cooled and moisture increased entering into the Little Ice Age, western juniper began to gradually increase based on pollen records (Mehringer and Wigand 1990).

Recent Historic Patterns

Late Presettlement Woodland Expansion and Infill

"In the past 150 to 300 years, expansion of juniper and pinyon from their original distribution and densities has presented problems in classifying present vegetation in relation to the historic climax plant community, and in correlating this plant community to specific environmental factors" (Vasek and Thorne 1977).

Climate conditions in the Great Basin and Colorado Plateau during the middle of the last millennium were generally cooler and wetter than currently (fig. 3-9) with the thermal minimum occurring around A.D. 1700 (Eddy and Bradley 1991; Lowell 2000). However, drought persisted in much of the northwestern Great Basin until the last phase of the Little Ice Age, which began about 300 years ago. Following the end of the Little Ice Age, temperatures warmed, but conditions stayed relatively moist with a persistent wet period between 1905 and 1917 across the West (Woodhouse et al. 2005). After 1920, climate in the Great Basin and Colorado Plateau became generally warmer (Eddy and Bradley 1991; Ghil and Vautgard 1991; Woolfenden 1996) and drier (Cook et al. 2004) and was marked by severe droughts. In the Great Basin one of the most severe droughts began in the 1920s, peaking in the 1930s (fig. 3-12a; Miller et al. 2011). However, in the Colorado Plateau this drought was not as severe (fig. 3-12b). Droughts again occurred across both regions in the 1950s, late 1980s, and early 2000s, resulting in considerably greater tree mortality in the Colorado Plateau than the Great Basin (Biondi and Bradley 2013; Breshears et al. 2005; Shaw et al. 2005).
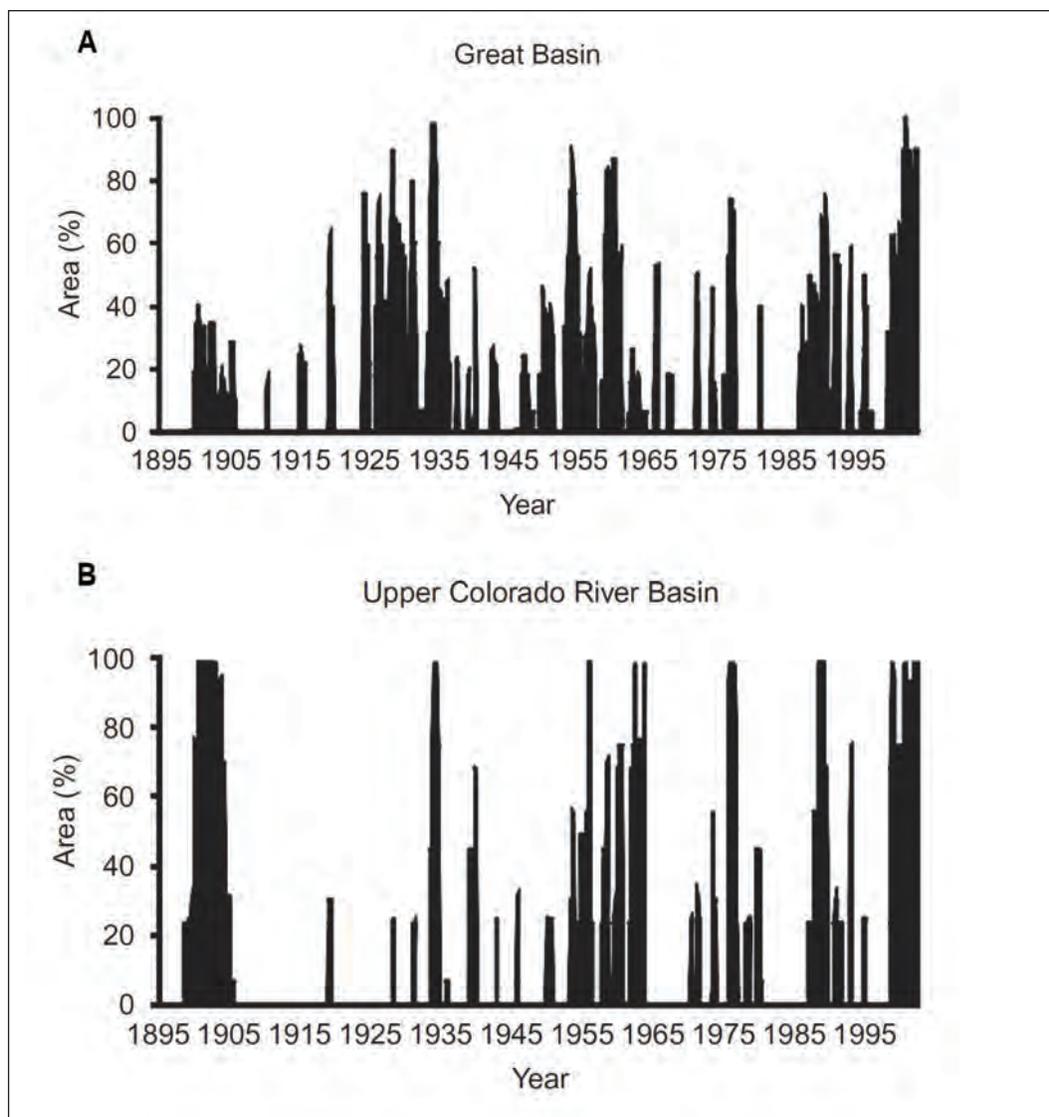
Figure 3-12—Percent area of major river basins in the (A) Great Basin and (B) Upper Colorado River Basin experiencing drought conditions from 1895–2004. Note the wet periods in the early 1900s for both regions. Graphs depict the Palmer Drought Severity Index (1965), a measure of the extent of departure from long-term average precipitation, temperature, and available water capacity (Miller et al. 2011).

USDA Forest Service RMRS-GTR-403. 2019.

Figure 3-13—Remnant wood in old growth stands can persist for centuries, providing clues to past woodland stand structure and climate. Horse Ridge, central Oregon. (Photo by Rick Miller, Oregon State University.)

Between the mid-1800s to the mid-1900s, packrat middens, pinyon and juniper pollen, and tree-ring chronologies strongly suggest an unprecedented increase in the rate of expansion and infill (increased density) of pinyon and juniper woodlands (table 3-4) (Biondi and Bradley 2013; Cole et al. 1997; Landis and Bailey 2005; Mehringer and Wigand 1990; Miller and Wigand 1994b). Questions addressed in this section include:

1) What were the structure and composition of mature woodlands prior to Eurasian settlement?

2) Were the distribution and density of pinyon and juniper woodlands stable just prior to Eurasian settlement?

3) What is the magnitude of expansion into sagebrush and associated plant communities?

4) Are there regional differences in woodland dynamics and disturbance regimes?

5) What were the contributing factors to woodland expansion and infill across the Intermountain West during the late 19th and 20th centuries?

Presettlement Woodlands Just Prior to Settlement

Evidence of change in woodland structure, infill, and expansion since the 1850s comes largely from tree-ring chronologies, remnant wood, sediment cores, charcoal, fire-scars, soil characteristics, and early observations (fig. 3-13). Most historical photos were taken several decades following postsettlement tree harvest of thousands of acres during the Comstock period between 1863 and 1908 across Nevada and Utah, and they must be interpreted carefully (Creque et al. 1999). Distant, casual observations of woodland structure must also be interpreted cautiously as illustrated by Woodbury (1947). He said that, "From a distance, the forest (pinyon and juniper woodland) appears to be a solid dark belt, but upon closer inspection, it is found that the stand is open with much space between the trees."

Table 3-4—Tree chronology studies across Great Basin and Colorado Plateau. Infill (I) occurs where old trees (mature prior to 1850) were present and expansion (E) where there was no evidence of old trees having been present on the site. Post-1950 trend in tree establishment compared to the first half of the 20th century.

Reference	Location	Period of chronology	Expansion (E) or Infill (I)	Tree species	Plant community	Peak establishment	Post-1950 trend	Comments
Adams 1975+A2:J21	central OR	1890–1970	NA	JUOC	ARTRv-PUTR	1920–1930	Decline	Postsettlement woodland of which 71 percent of the trees established between 1900 and 1935; increase started 1900
Barger & others 2009	Grand Staircase SC UT	1440–1940	I	PIED JUOS	Shallow sandy loams; ppt 14"	1910–1940 (1920–1929)	Na	Little evidence of fire; no difference in tree density between grazed and nongrazed although tree basal area was greater in grazed; herbaceous understory cover was 5–6 percent, not adequate to support a surface fire
Biondi & Bradley 2013	Mt Irish SE NV	1400–2000	I	PIMO	Upper moisture end of pinyon	1900–1940 1930–1960	Decline	Mt Irish: PIMO dominant tree more than doubled since 1800; mortality less than 10 percent; primarily infill; Mt Clover: PIMO more than doubled since 1800; mortality less than 10 percent; primarily infill; tree establishment primarily PIMO
	Clover Mts SE NV	1400–2000	I	PIMO	Upper moisture end of pinyon	1880–1930	Decline	
Blackburn & Tueller 1970	4 locations south of Ely, NV	1700–1970	E?	PIMO-JUOS	ARNO/PSSP	1920	Decline	No presettlement trees present; did not measure large dead wood; site south of Ely, likely harvested in the late 1800s and is regenerating
		1700–1970	I	PIMO	ARNO/PSSP	1920	Decline	Low density of old trees may indicate this site was not harvested in the late 1800s
Bolsinger 1989	NE CA	<1780–1980	E & I	JUOC	Mixed	1880–1930		73 percent of woodlands < 100 yr old; trees in 50-yr age classes
	E CA	<1780–1980	E & I	JUOS PIMO	Mixed	1880–1930		59 percent of woodlands <100 yr old; trees in 50-yr age classes
Burkhardt & Tisdale 1976	SW ID	1870–1970	E & I	JUOC	ARTRv/FEID-PSSP & ARAR	1920–1940	Decline	JUOC actively invading mountain big sagebrush; mature stands primarily ridges and rim rocks
Despain & Mosely 1990	AZ near Flagstaff		I	PIED JUOS	Elevation 6,700 ft; ppt 18" adjacent to PIPO	1880–1920 on > 50 percent plots	Decline	10 locations across 740 acres; 20 to 40 trees cored in each macro plot. Estimated age structure from diameters, which relationships were generally weak. Stand structure and lack of dead and downed wood suggest tree density has increased significantly within the past 200 yrs.
Eisenhart 2004	Uncompahgre Plateau W CO	1300–2000	E & I	PIED JUOS	Mixed	1880–1920	Decline	4 locations across the Colorado Plateau; shows fairly steady expansion based on stand age between 1500 to 1950; peaking around 1900 on 4 of the 3 sites. Rapid tree ring growth around 1910–1920.
Floyd-Hanna et al. 2004	Mesa Verde SW CO	1600–2000	I	PIED	Mixed	1910–1940	Decline	CP had longer wet period in first half of 1900s than the n GB; all stands showed an increase from 1900-1940.

(Continued)

Table 3-4—(Continued).

Reference	Location	Period of chronology	Expansion (E) or Infill (I)	Tree species	Plant community	Peak establishment	Post-1950 trend	Comments
Floyd-Hanna et al. 2008	N AZ	na	I	PIED JUOS	ARTR-PUTR hot-dry; ppt 12"	early 1900s	Na	Measured age of stands by sampling oldest trees; most stands had a large cohort of trees establishing in the early 1900s
Hattori et al.	W & C NV	NA	NA	PIMO JUOS	Mixed	NA	N	Chronology for depicting timing of cut during late 1800s mining. Upper elevation woodlands composed of larger but less dense canopies then lower reestablished (cutting for charcoal) dense younger woodlands below.
Heyerdahl et al.; unpublished data Horse Ridge	C OR Horse Ridge	800–2000	I	JUOC	ARTRv, ARTRw, PSSP	1600–1800		Relatively stable old-growth woodland; pulse occurs around 1700
Johnson & Miller 2008	SW ID	< 1850	E & I	JUOC	mixed	1900–1920	Decline	OR > 9-fold increase; ID > 2-fold
Landis & Bailey 2005	AZ Mogollon Rim	1400–2000	I	PIED JUOS	Basalt	1880–1960	No	Tree establishment declined around 1950 on sandstone derived soils but not occurring on basalt or limestone derived soils. Canopy cover in 1860 and 2002 was 4.5 and 30 percent (670 percent increase) on basalt, 11.5 and 33.8 percent (300 percent increase) on limestone, and 14.7 and 44.3 percent (300 percent increase) on sandstone. Since the 1950s drought, recruitment on limestone and sandstone dominated by JUOS and on basalt PIED.
					Sandstone	1880–1920	Decline	
			I		Limestone 6,300–6,800 ft	1860–1960	No	
Margolis 2014	NC NM			PIED JUMO JUSC	BOGR savanna 6,400–8,000 ft	1900–1930	Decline	Tree density has increased 600 percent since the late 1800s.
	UT	1600–2000	E & I	PIMO JUOS		1940–1970	Decline	Transects from the lower to upper woodland boundary covering Artrw, Artrv, and Arar; 9-fold increase since 1900
	NV	1800–2000	E & I	PIMO, JUOS		1920–1950	Decline	
	ID	1800–2000	E & I	JUOC		1920	Decline	5–10 percent cover; 5–11 trees/ac
Miller et al. 2008	OR	1800–2000	E & I	JUOC		NA	Decline	Study showed substantial infill and expansion across the Great Basin

(Continued)

Table 3-4—(Continued).

Reference	Location	Period of chronology	Expansion (E) or Infill (I)	Tree species	Plant community	Peak establishment	Post-1950 trend	Comments
Miller & Heyerdahl 2008		1750–2000	I	JUOC	ACOC	1880–1900	Decline	Infill: about double; pulse not consistent across communities e.g., sage-fescue or sage bluebunch
		1870–2000	E		ARTRv/PSSP	NA	Decline	Last fire across sagebrush sites in 1941, no evidence of old charred trees
	NE CA	1880–2000	E	JUOC	ARTRv/FEID	NA	Decline	Frequent fire regime
Miller & Rose 1995	Steens	1700–1990	E	JUOC	ARTRv, ARAR	NA	Decline	expansion into ARTRv; scattered old trees present on ARAR; few on south aspects; none on north slopes or POTR
			E	JUOC	ARTRv/FEID	1900–1920	Decline	
Miller & Rose 1999	C OR	1840–1995	I	JUOC	ARAR & south	1900–1920	Decline	OG very open ARAR sites; trees > 2-fold increase; ARAR old trees only accounted for 1.5 percent with a canopy of < 5 percent; OG very scarce in ARTRv plots
Miller unpublished data	Horse Ridge E OR	1587–1997	I	JUOC	ARTRv, FEID	1700	No	Very minimal anthropogenic disturbance
Soule & Knapp 2000	I land INRA E OR	1827–1997	yes	JUOC	ARTRv, PSSP	1978–1996	No	Old relict stand (island) with minimal disturbance
Tausch et al. 1981	NV	1400–1980	reestablish	PIMO JUOS		1905–1930	Decline	48 percent of the juniper and 71 percent of the pinyon plots were dominated by trees < 150 yrs old; really a study of expansion based on fig 5 not number of trees establishing.
Tausch & West 1988	SW UT	1835–1976	N/A	PIMO, JUOS		1920–1945	Decline	Reestablishment following mid-1800s fire was primarily pinyon but 38 percent juniper postfire survival and only 0.6 percent pinyon; decline after 1945 but surge in 1975 (based on seedling count)
Wall et al. 2001	E OR, NE CA, NW NV	1880–1990	E	JUOC	Aspen	1910–1919	Decline	Chronology dates back to oldest tree measured
Young & Evans 1981	NE CA	1850–1960	I	JUOC	Big sagebrush	1880–1920	Decline	Few scattered old trees present
		1600–1960	I	JUOC	Low sagebrush	1750 & 1880–1915	Decline	Peaks 1650–1750; 1850–1900
Waichler et al. 2001	C OR	1300–1990	I	JUOC	ARTRv, ARTRw, PSSP	1700–1850	Decline	Chronology for stable old-growth stand about 20 percent trees < 150 yrs; 11–33 percent tree cover and 30 trees/ac; infill very slow

ACOC = western needlegrass, ARAR = low sagebrush, ARNO = black sagebrush, ARTRv = mountain big sagebrush, ARTRw = Wyoming big sagebrush, FEID = Idaho fescue, JUOC = western juniper, JUOS = Utah juniper, PIED = twoneedle pinyon, PIMO = singleleaf pinyon, PSSP = bluebunch wheatgrass, PUTR = bitterbrush.

Early observations can often appear contradictory but seem to reflect the heterogeneity of pinyon and juniper woodlands in the Great Basin and Colorado Plateau at the time of settlement. When John Muir traveled across 11 mountain ranges in Nevada in 1878, he wrote about pinyon:

"In the number of individual trees and extent of range this curious little conifer surpasses all the others combined. Nearly every mountain in the state is planted with it, from near the base to a height of from eight thousand to nine thousand feet above the sea. Some are covered from base to summit by this one species, with only a sparse growth of juniper on the lower slopes to break the continuity of these curious woods. ... Tens of thousands of acres occur in one continuous belt. Indeed, the entire state seems to be pretty evenly divided into mountain ranges covered with nut pines and plains covered with sage—now a swath of pines stretching from north to south, now a swath of sage; the one black, the other gray; one severely level, the other sweeping on complacently over ridge and valley and lofty crowning dome." (From early writings of John Muir, reprinted in Steep Trails, *1994.)*

However, reports by John C. Fremont in 1842 and Parley P. Pratt in 1851 described vegetation in the Mountain Meadows of southwestern Utah very differently. Their separate accounts summarized by Cottam and Stewart (1940) provided a very different picture.

"The entire valley as seen by Fremont in 1842 and Pratt 1851 was grass-covered. ... Pratt stated the surrounding hills were abrupt, but rounded off, presenting a variety of beauteous landscapes and everywhere richly clothed with the choicest kind of bunch-grass, and bordered in their higher eminences with cedar and nut pine sufficient for fuel."

In the same valley, Cottam and Stewart conducted an extensive study in 1930 and reported:

"For a number of years young junipers in great numbers have been appearing in areas hitherto unoccupied by them" (Cottam and Stewart 1940).

In the Crooked River Canyon, upriver from Prineville, Oregon, Peter Skene Odgen wrote in his journal in 1825 that *"as we headed up the Crooked River we saw the occasional cedar tree up on the rimrock."*

Today the river canyon is densely populated with western juniper (fig. 3-14;, Rick Miller Professor emeritus Range Ecology, Oregon State University, Corvallis, Oregon, personal observation, 2016). In 1901, Griffith (1902) observed only scattered stands of juniper on Steens Mountain in southeastern Oregon where little cutting had occurred. And in Arizona, Leopold (1924) observed young trees encroaching on grassland parks after the introduction of livestock.

Structure of Persistent Woodlands Prior to Settlement

The 1850s marked the end of the Little Ice Age and the early 1860s a significant rise in modern anthropogenic impacts—including the introduction of large numbers of livestock throughout the Intermountain West (fig. 3-9). In the literature, this period is frequently used to separate presettlement and postsettlement pinyon and juniper woodlands. Tree-age ratios and the presence and densities of snags, remnant logs, and stumps in old-growth woodlands strongly suggest tree densities were lower prior to 1850 than today (table 3-4; figs. 3-15, 3-16). Studies indicate tree densities in old-growth woodlands increased two- to ninefold since 1860, which exceeds expected compensation for natural mortality (Baisan and Swetnam 1990; Barger et al. 2009; Biondi et al. 2013; Floyd et al. 2017; Floyd-Hanna et al. 2004; Hattori and Thompson 1987; Landis and Bailey 2005; Miller et al. 2008; Miller and Rose 1999, 2005; Waichler et al. 2001).

The large difference between low mortality of mature trees and rapid tree establishment since 1850 has resulted in significant infill in many old-growth (persistent)

Figure 3-14—Peter Skene Ogden observed only scattered "cedar trees" up on the rimrock as he traveled along the Crooked River in the 1820s. Crooked River Canyon, central Oregon. (Photo by Rick Miller, Oregon State University.)
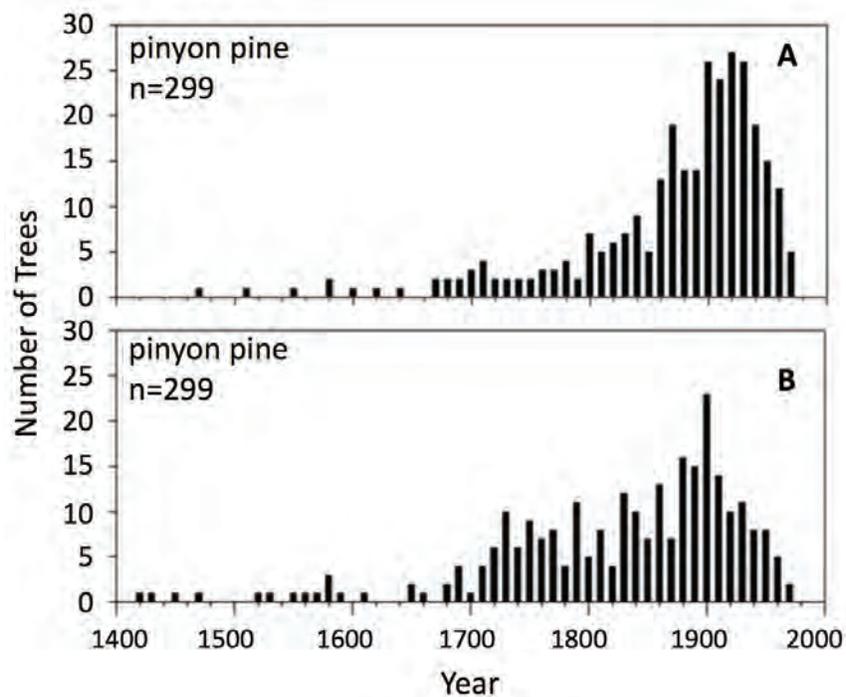
Figure 3-15—Age structure of pinyon pine sampled across (A) Mount Irish (B) and Clover Mountain in Nevada. Both study areas represent persistent woodlands and illustrate the pulse of tree establishment (infill) during the early 1900s (Biondi and Bradley 2013).
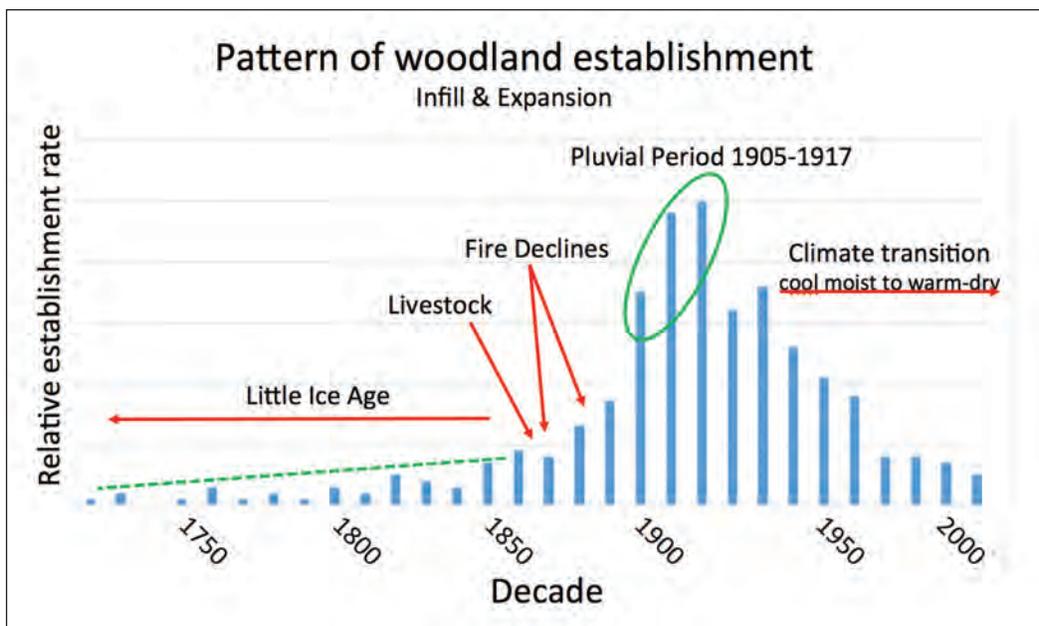
Figure 3-16—Decadal establishment of pinyon and juniper, which represents the general pattern shown by many pinyon-juniper chronologies conducted throughout the Great Basin and Colorado Plateau since the late 1800s.

woodlands across the Intermountain Region. Densities of dead trees, stumps, and logs, which can persist for centuries, suggest natural mortality of mature trees in largely undisturbed (in the absence of episodic events) old-growth woodlands is very low, less than 1 percent per century (Landis and Bailey 2005; Waichler et al. 2001). In relatively undisturbed old-growth woodlands in central Oregon, trees less than 150 years old made up 22 percent of the total population (85 trees per acre), well exceeding mortality rates that were less than 1 percent over multiple centuries (fig. 3-13; Waichler et al. 2001). The more open pre-1850s woodlands across the Great Basin and Colorado Plateau were often characterized by understory vegetation of shrubs, grasses, and forbs, frequently mentioned in paleobotany literature. However, there are persistent woodlands that have experienced heavy mortality from past periods of severe drought and insect infestation, especially the Colorado Plateau, and infrequent fires resulting in persistent woodlands exhibiting less old-growth characteristics. Also, considering the large spatial heterogeneity in climate, soils, and topography across the Great Basin and Colorado Plateau, closed stands with little understory cover, such as those observed by Muir in 1878 and Phillips (1909) at the turn of the century, also existed.

Based on estimated presettlement tree densities from tree core evidence, tree canopy cover was relatively open throughout many pinyon and juniper woodlands throughout the Great Basin and Colorado Plateau. In Arizona and Utah, presettlement tree canopy cover across three separate old-growth woodlands was estimated at 4.5–14.7 percent in 1860 (Landis and Bailey 2005). Before 1860 in the northwestern Great Basin, tree canopy cover across an extensive old-growth woodland varied from less than 5–33 percent (Waichler et al. 2001). These open stands likely supported high understory cover where climate and soils were suitable. Pollen and packrat midden evidence also suggests woodlands with well-developed sagebrush and grass understories (Anderson et al. 2000; Spaulding 1990; Wigand et al. 1994a,b; Woolfenden 1996).

Environmental factors that limited stand closure of presettlement woodlands are unknown. There is limited evidence that reoccurring low-intensity surface fires

maintained the open structure of presettlement woodlands (Baker and Shinneman 2004; Romme et al. 2009; Waichler et al. 2001). But low-severity fires were reported as common in pinyon and juniper woodlands of east Texas, influencing tree densities (Poulos et al. 2009). In the central Great Basin, frequent low-severity fires were also attributed to relatively lower density tree canopies, but in the upper elevation belt where pinyon was intermingling with mixed conifer (Biondi et al. 2011). On a nearby mountain range, the fire regime was characterized by infrequent fires with occasional patchy high-severity events. In the absence of reoccurring low-intensity fires, openness of pinyon and juniper woodlands, frequently reported in the paleobotany literature, would have had to be the result of limited tree-seedling establishment and/or natural thinning processes, especially of young trees. Establishment of tree seedlings resulting in infill and thickening of presettlement woodlands may have been limited or slowed down by competition from herbaceous vegetation (Guenther et al. 2004; Madany and West 1983; Soulé et al. 2004). Thinning may have also been caused by droughts and pests (Eisenhart 2004; Floyd et al. 2009; Greenwood and Weisberg 2008; Koepke and Kolb 2013; Koepke et al. 2010). We have little information as to how Native Americans may have managed woodland structure prior to Eurasian settlement.

Presettlement Fire Regimes

Persistent woodlands occur where disturbance regimes, soils, and climate are inherently favorable for pinyon and/or juniper (Romme et al. 2009) from establishment through maturation to late succession. Persistent woodlands exhibiting old-growth characteristics occur across a wide range of soils and parent materials but develop where stand-replacement disturbances are rare, extending over centuries (Harper and Davis 1999; Leonard et al. 1987; Pearson 1931; Springfield 1976; Stringham et al. 2015; West et al. 1998). Old-growth woodlands or trees are most often found on rock outcrops, knolls, and ridges with soils that are shallow, coarse, rocky, and often high in clay or sand (Barney and Frischknecht 1974; Bauer and Weisberg 2009; Cottam and Stewart 1940; Holmes et al. 1986; Miller and Rose 1999; Nicol 1937; Stringham et al. 2015; Woodbury 1947). These soils are often nutrient limited and support sparse and discontinuous fine surface fuels, resulting in fire return intervals measured in centuries (Barger et al. 2009; Floyd et al. 2004; Waichler et al. 2001). However, on more productive sites with moderately deep to deep soils, where cool and moist sagebrush communities were persistent, fire has been an important disturbance limiting expansion of pinyon and juniper. In general, fires typically increased (as evidenced by charcoal abundance and size) during extended wetter periods punctuated by droughts and declined during extended periods of drier climate (Wigand 2017).

Fire regimes characterize the nature of fire for a specific area over an extended period of time. Characteristics of fire regimes include frequency of occurrence, severity, intensity, size, complexity, and season for a specific area or plant community. Fire frequency, often reported as fire return interval, is an important factor determining the dynamics and persistence of plant communities. Fire return interval is sometimes confused with fire rotation. Fire rotation is the time required for a cumulative area burned to equal the size of the defined area of interest (Sugihara et al 2006). Fire rotation does not account for the heterogeneity of fire in time and space across a landscape or at local scales (Agee 1993; Heinselman 1973; Reed 2006). But the term is useful in describing fires in relatively homogenous vegetation and at regional scales when it is not realistic to evaluate landscapes at fine scales as a result of limited data. Fire return interval is the length of time between fires for a specific area. Mean fire return interval is most commonly reported in the literature. It represents the arithmetic average of all fire intervals over a designated period for a specific site or defined area. However, variation

in fire return intervals is a more important determining factor influencing long-term vegetation dynamics and persistence.

> *"Means are abstract values that we often falsely interpret as a concrete thing. Variability is the universal reality"* (Gould 1997).

The variation of fire return intervals in both time and space can describe the fine-scale complexity of fire across landscapes. The heterogeneity of fire occurrences at finer scales across landscapes have important ecological consequences related to processes, functions, and habitat configurations within landscapes (Miller et al. 2011). However, a primary limitation of calculating fire return intervals across semiarid communities is often the scarcity of fire scar data. In plant communities where trees persist or have recently expanded, tree-age structure analysis can supplement fire scar data as a proxy for estimating fire return intervals, especially where fires are stand replacing (Agee 1993).

Historic fire regimes were highly heterogeneous at local and regional scales in the Great Basin and Colorado Plateau due to variation in ignition, fuel, climate, topography, and vegetation (fig. 3-17; Marlon et al. 2012; Miller et al. 2011). Recent fire records across this region show that distinct regional patterns in present-day fire occurrences are largely determined by climate (fig. 3-18; Board et al. 2018; Littell et al. 2009), especially as climate affects ignitions, fuel characteristics, and timing and amounts of precipitation. Fires are significantly more common in the northern Intermountain Region and along the Wasatch Mountains and Mogollon Rim than the southern Great Basin and Colorado Plateau. While abundance and distribution of invasive annual grasses are likely influencing present-day regional fire occurrences in the northwestern Great Basin, Columbia Basin, and Snake River Basin, they do not represent the only reason for increased fire frequency in these regions. Seasonal patterns of lightning ignition in the northwestern Great Basin coincide with the season of peak surface fuel abundance and low fuel moisture content (West 1983).

Figure 3-17—Topography can be linked to very patchy fires, especially when burning under less severe weather conditions. Fairview Range, Nevada. (Photo by Rick Miller, Oregon State University.)
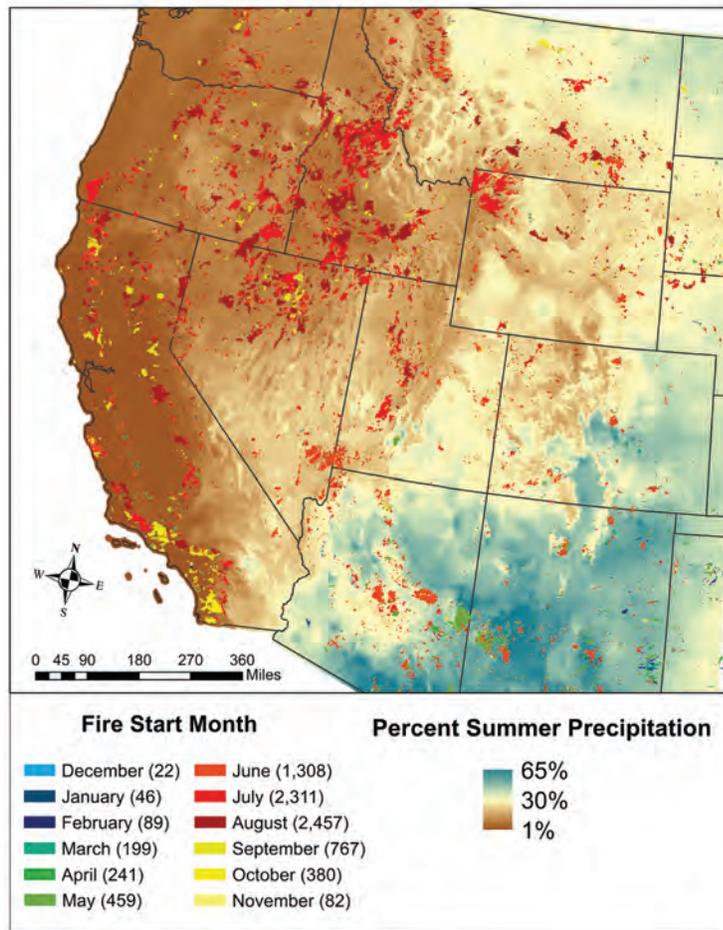
Figure 3-18—Large western fire events (more than 1,000 acres) between 2000–2014. Depicted are fire event timing (month of start) and percentage of summer precipitation (derived by David Board).

Fire Start Month

December (22)   June (1,308)
January (46)   July (2,311)
February (89)   August (2,457)
March (199)   September (767)
April (241)   October (380)
May (459)   November (82)

Percent Summer Precipitation

65%
30%
1%

Fire regimes can also vary abruptly among adjacent sites, as soils or aspects change, resulting in complex spatial patterns in fire return intervals. In the northwestern Great Basin, historic mean fire return intervals ranged from 10 to over 100 years across different plant communities occupying different aspects and soils (Miller and Heyerdahl 2008). Relatively short fire return intervals (less than 15 years) were reported on the more productive soils and northerly aspects where fire intervals were relatively short, fine surface fuels were 2,000–2,400 pounds per acre, and cover was 30–45 percent (predominately Idaho fescue and bluebunch wheatgrass grassland with less than 3 percent ponderosa pine cover and a few old junipers growing on rock protected microsites). In an immediately adjacent plant community (western juniper-western needlegrass) with little to no aspect and coarse-low productive soils, fine surface fuel loads were 100 pounds per acre with less than 10 percent cover. Fire return intervals here were more than 100 years. On the Colorado Plateau, mosaics of big sagebrush were typically found on the deeper soils (fig. 2-38) immediately adjacent to old-growth pinyon and juniper woodlands, which often occupied the shallow soils (fig. 2-7; Floyd et al. 2008). The combination of fuels and competition on these deeper soil sites may have limited the conversion to persistent woodlands.

Presettlement Fire Regimes: Pinyon and Juniper Woodlands

Fire return intervals are long, often centuries long, in persistent pinyon and juniper woodlands (Baker and Shinneman 2004; Barger et al. 2009; Board et al. 2018; Floyd et al. 2000, 2004, 2017; Huffman et al. 2008; Waichler et al. 2001). This allows woodlands to reach late successional stage and develop old-growth structural characteristics. Fire regimes in persistent or old-growth woodlands have changed little in the past several

centuries. But there appears to be an increasing trend of stand-replacement fires in persistent woodlands over the last 30 years (fig. 3-19; Barger et al. 2009; Board et al. 2018; Floyd et al. 2000, 2004, 2017; Waichler et al. 2001). Sparse surface fuels are likely the reason for infrequent fire. Fire spread in closed-canopy woodlands lacking an understory requires severe weather conditions and often results in stand-replacement events (Blackburn and Bruner 1975; Schroeder 1966). Low-severity fires in persistent woodlands are rare (Baker and Shinneman 2004; Romme et al. 2009; Waichler et al. 2001). Limited surface fuels in persistent woodlands are often a function of low productivity and shallow, very coarse-textured soils (Leonard et al. 1987; Stringham et al. 2015; USDA NRCS Harney County, Soil Survey of Harney County Area, Oregon 1997; Randy Lewis, Soil Scientist, USDA NRCS, Tremonton, Utah, personal communication, 2015). On more productive sites, competition from trees also limits understory productivity (Roundy et al. 2014a). Minimum surface fuels sufficiently able to carry fire under moderate weather conditions are estimated at 740 pounds per acre (Bunting et al. 1987; Evers et al. 2013; Gruell et al. 1986).

We have little information as to the role of fire in open persistent woodlands with abundant understory vegetation (Romme et al. 2009). Although there is limited evidence that fire thinned some persistent open woodlands (fig. 3-20; Miller and Rose 1999), there is little evidence of reoccurring fires in the majority of these open old-growth stands (Robin Tausch, Retired Range Scientist, USDA Forest Service, Rocky Mountain Research Station, Reno, Nevada; and Rick Miller, Professor Emeritus, Range Ecology, Oregon State University, Corvallis, Oregon, personal observations, 2017). And many of these once open stands have exhibited significant recent infill, especially during the first half of the 20th century.

Figure 3-19—Fire return intervals in old-growth woodlands can typically be measured in centuries. However, in the past 30 years, there has been a significant increase in stand replacement fires—such as in Mesa Verde where a large portion of the monument's woodlands have burned since the 1990s. Mesa Verde 2000 Pony Fire, Colorado. (Photo by Rick Miller, Oregon State University.)

Figure 3-20—Presettlement woodlands were often considered relatively open, based on midden data and age structure of live and dead trees. There is little evidence of reoccurring low intensity fires that maintained this open structure. With limited mortality of mature trees, successful establishment of trees resulting in infill was likely very low. Horse Ridge, central Oregon. (Photo by Rick Miller, Oregon State University.)

In the northern Great Basin, low sagebrush-Sandberg bluegrass communities (which occur on shallow, heavy clay soils) support only low levels of fine surface fuels, resulting in infrequent fires. However, in eastern Oregon, following 2 or more wet years (Holmes et al. 1986), mixed severity fires in 1717 and 1855 resulted in nearly 100 percent mortality of small trees and around 25 percent mortality of old mature trees (Miller and Rose 1999). Many semiarid communities are fuel limited, thus the accumulation of fine fuel during wet years increases fuel continuity at the landscape level and has been closely linked to large and widespread fires (Abatzoglou and Kolden 2013; Allen et al. 1995; Baisan and Swetnam 1997; Brown et al. 2008; Grissino-Mayer and Swetnam 2000; Miller and Rose 1999; Swetnam and Betancourt 1998).

Presettlement Fire Regimes: Associated Sagebrush Communities

In sagebrush and grassland ecosystems for which environmental conditions were suitable for tree establishment and growth—but trees were absent—woodland expansion was likely limited or slowed by fire and competition from perennial grasses in presettlement times (Wright et al. 1979; see the competition and grazing subsections). Fire return intervals in sagebrush and grassland communities—especially on cool and moist sites with moderately deep soils—contained more contiguous surface fuels, resulting in shorter fire return intervals than most persistent woodlands (Bauer and Weisberg 2009; Miller and Heyerdahl 2008). But fire return intervals from 10 to more than 100 years varied in time and space across heterogeneous landscapes that supported different sagebrush and grassland communities (Lesica et al. 2007; Mensing et al. 2006; Miller and Heyerdahl 2008; Miller and Rose 1999; Wright and Bailey 1982). Within a landscape or watershed, historical fire return intervals often varied at relatively fine scales, from tens to hundreds to thousands of acres (Heyerdahl et al. 2006; Miller and Heyerdahl 2008).

Data on fire return intervals is very limited for the warm, dry sagebrush and semi-desert communities (for example, in Wyoming, big sagebrush and black sagebrush with less than 12 inches precipitation) at the lower elevation woodland boundary. Fires were likely infrequent, occurring only every hundred years or more (Mensing et al. 2006; Wright and Bailey 1982) with a high degree of variation around the mean return interval (Miller et al. 2001b). Low abundance and continuity of fine surface fuels limited fire occurrences and spread. In addition to occasional fires, environmental constraints—including drought, cold-air inversions in the spring during budding, pests, and competition from perennial grasses—limited woodland expansion in the lower elevation warm and dry shrublands. Moisture and temperature conditions in the warm and dry sagebrush and semi-desert communities would have been especially harsh on seedling establishment and young tree survival.

Mountain big sagebrush communities characterized by cool and moist conditions (12–16 inches precipitation) on moderately deep to deep mollic soils are much more likely to burn than drier sagebrush communities (e.g., Wyoming big, low, and black sagebrush), supporting significantly greater fine surface fuels (Heyerdahl et al. 2006; Lesica et al. 2007; Miller and Heyerdahl 2008; Miller and Rose 1999). Mountain big sagebrush communities, which often intermingle with and occur at the mid and upper elevation boundary of pinyon and juniper woodlands, were considerably more extensive prior to 1850 than they are today (Miller et al. 2008; Tausch and Nowak 1999). Along the ecotone between mountain big sagebrush and ponderosa pine or Douglas-fir communities, fire return intervals were relatively short, ranging from 10 to 30 years (Biondi and Bradley 2013; Heyerdahl et al. 2006; Lesica et al. 2007; Miller and Heyerdahl 2008; Miller and Rose 1999).

However, intervals in drier mountain big sagebrush communities (approximately 12 inches of precipitation) were likely longer (fig. 3-21). Both rates of establishment and growth of pinyon and juniper can be high in these cool and moist, deep soil sites (Barney and Frischknecht 1974; Campbell 2016; Johnson and Miller 2006). Fire and competition from perennial herbs were the most likely factors in limiting woodland expansion into these cool and moist mountain big sagebrush and mountain brush communities. And competition from undisturbed herbaceous layers may have slowed the rates of tree establishment, allowing for less frequent fire return intervals to maintain a persistent shrub-steppe community (Wright et al. 1979).
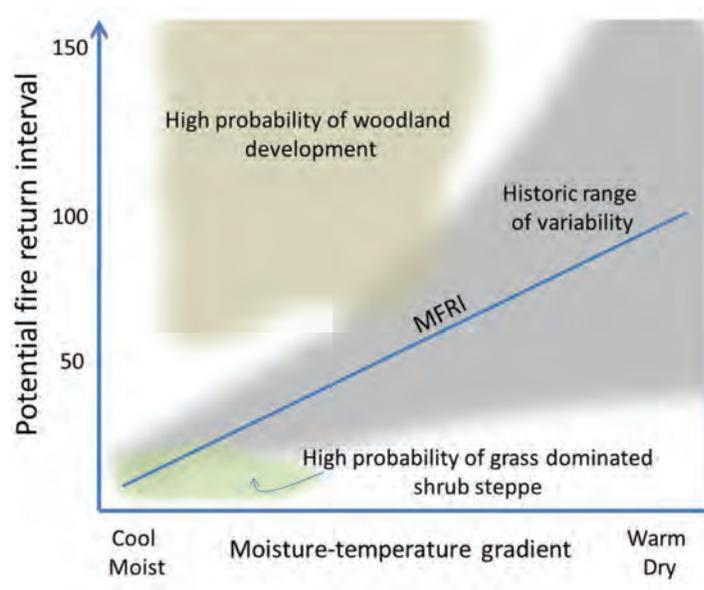
Figure 3-21—Conceptual model illustrating the range of historic potential mean fire return interval (MFRI) and historic range of variability (HRV) as it changes across a temperature and moisture gradient for sagebrush steppe (gray), persistent woodland (brown), and grassland (green). Fire frequency is a function of moisture and temperature, which influences fuel structure, composition, and biomass in addition to ignition. Persistent vegetation occupying the gray area would likely be a sagebrush herbaceous mix with the relative species abundance depending on time since fire (from Miller et al. 2011).

Presettlement Fire Regimes: Pinyon and Juniper Savanna

In the Colorado Plateau, where the proportion of warm season bunch and sod grasses can increase in composition in relation to the shrub layer as a result of increased summer precipitation, pinyon and juniper savannas and highland grasslands became more common (fig. 3-22; Nicol 1937; West 1999). In northern Arizona, early observations indicated that pinyon and juniper were expanding into grassland communities with the presettlement trees often occupying the rocky knolls and ridges (Leopold 1924; Nicol 1937). Fire history studies in pinyon and juniper savannas are limited (Romme et al. 2009). But a recent study in Arizona pinyon and juniper savannas reported presettlement mean fire return intervals of 7.8 years (Margolis 2014). The decline of both fire and perennial grass competition, which corresponded with the regional introduction of livestock, likely triggered the rapid increase in tree establishment across these savannas in the Colorado Plateau and Southwest (Margolis 2014; Nicol 1937; West 1984).

Postsettlement Woodland Expansion and Infill

Over the past 200 and more years, but prior to Eurasian settlement, evidence suggests pinyon and juniper woodlands were slowly expanding and infilling (Cole et al. 1997; Eisenhart 2004; Landis and Bailey 2008; Mehringer and Wigand 1990; Wigand et al. 1995; P.J. Mehringer, Retired Paleobotanist, Washington State University, Pullman, Washington, personal communication, 1990). This rate of infill increased shortly after Eurasian settlement, with rates of tree expansion into adjacent sagebrush ecosystems (fig. 3-23) and infill into established woodlands, peaking in the early 1900s across the Great Basin and Colorado Plateau (figs. 3-15, 3-16, 3-24; table 3-4; Barger et al. 2009; Biondi and Bradley 2013; Floyd-Hanna et al. 2004; Margolis 2014; Miller et al. 2008). Recent expansion rates appear to be greater than those occurring in the past (Cole et al. 1997; Mehringer and Wigand 1990; Miller and Wigand 1994). The rapid increase in tree expansion and infill is often attributed to climate, grazing, and reduced fire occurrence where surface fuels were once adequate to support surface fires.

Figure 3-22—Fire history studies in pinyon and juniper savannas are limited, but a recent study in Arizona pinyon and juniper savannas reported presettlement mean fire return intervals as relatively short. Northern Arizona. (Photo by Rick Miller, Oregon State University.)
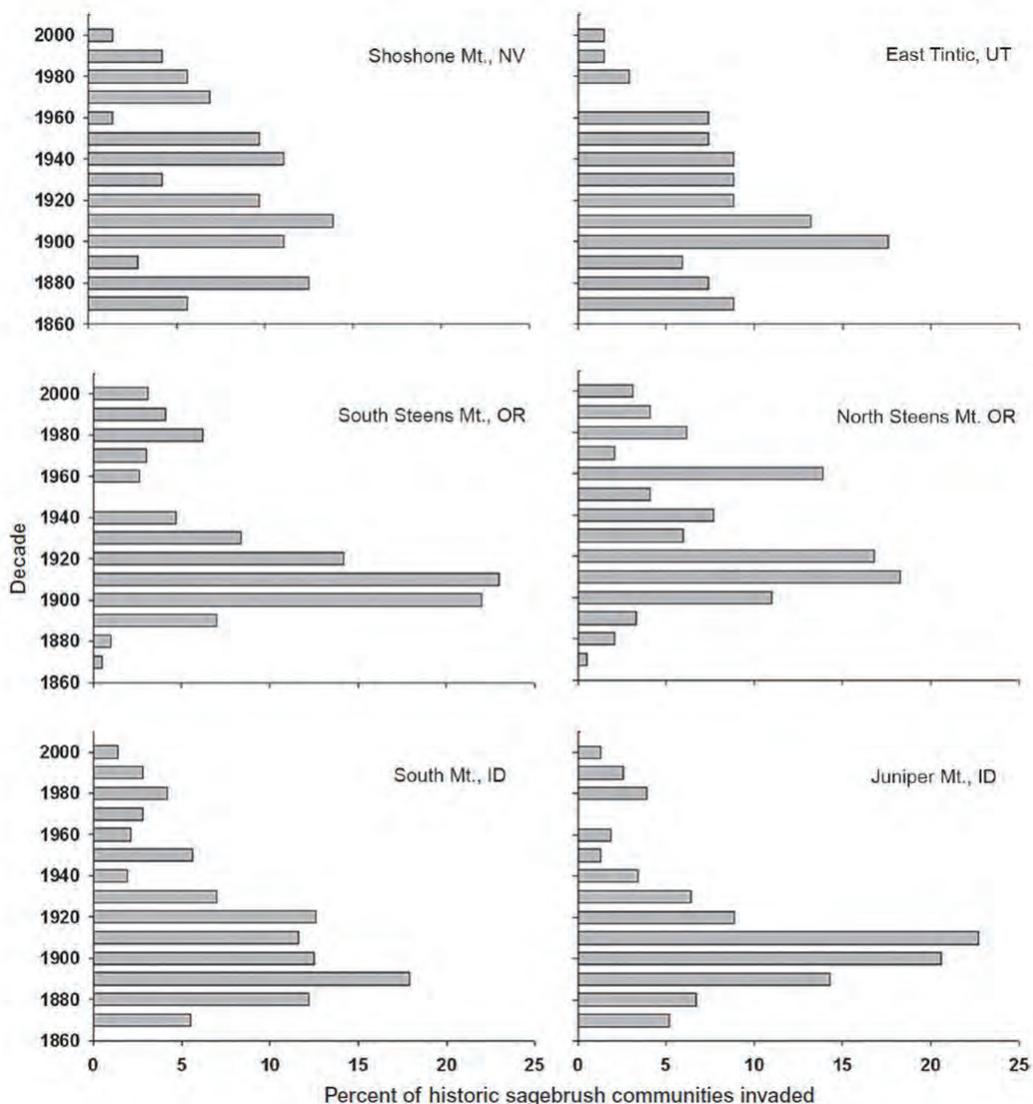
Figure 3-23—The relative proportion of pinyon and juniper expansion into historic sagebrush steppe stands lacking evidence of presettlement trees by decade between 1860 and 2000 (adapted from Miller et al. 2008).

However, consensus is lacking as to the relative importance of grazing, fire, and climate in recent expansions. Some attribute expansion primarily to natural climate phenomenon (Barger et al. 2009; Biondi et al. 2013; Floyd et al. 2004). Some attribute apparent woodland expansion to reestablishment following stand removal events, including extensive tree harvests through the Comstock period in the late 1800s (Lanner 1976; Young and Budy 1987). And others have attributed the recent increase to a combination of climate during the late 1800s and early 1900s and grazing—resulting in the reduction of fire occurrence (Baisan and Swetnam 1997; Heyerdahl et al. 2006; Miller and Rose 1999; Savage and Swetnam 1990; Swetnam et al. 2001; Touchan et al. 1995). Woodland expansion has also been linked to rising atmospheric CO_2 levels in the second half of the 20th century (Johnson et al. 1993; Soulé and Knapp 1999). Increased CO_2 levels can increase water-use efficiency in conifers, resulting in denser tree canopies (Knapp et al. 2001; Soulé and Knapp 1999).

Climate has always been a strong driver of woodland dynamics over time and space—but the effects of climate on woodland infill and expansion cannot be separated from fire,
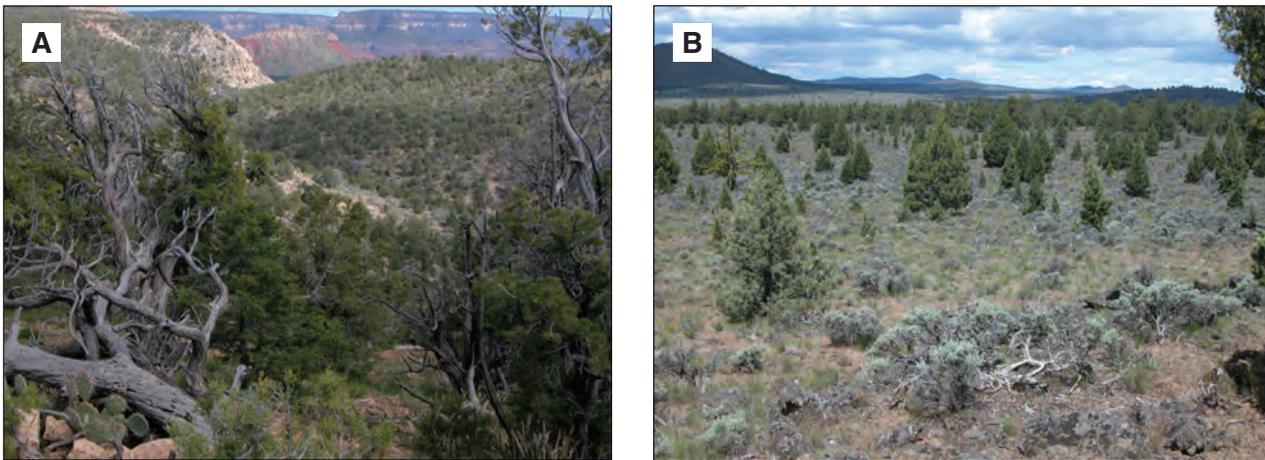
Figure 3-24—Many of the postsettlement trees we see today across the Colorado Plateau and Great Basin were established during a wet period between 1890 and 1920. (A) There has been a significant increase in tree density in this old-growth woodland on the Kaibab Plateau since the 1900s. (B) The mature trees on this site established during the first part of the 20th century expanding into mountain big sagebrush and Idaho fescue in central Oregon on moderately deep sandy loams. Presettlement trees, logs, and stumps are present but less than 1 per acre. (Photos by Rick Miller, Oregon State University.)

grazing (Caracciolo et al. 2016; Leopold 1924; Wright et al. 1979), insect infestations, and disease (Swetnam and Betancourt 1998). And variation in climate can increase or decrease anthropogenic effects on ecosystems (Swetnam and Betancourt 1998). With the exception of climate, the relative importance of the other attributing factors will vary within watersheds and across geographical regions.

Packrat middens, pollen, and tree-ring chronology evidence suggest an unprecedented increase in the expansion and infill of pinyon and juniper woodlands since the late 1800s (table 3-4; Cole et al. 1997; Landis and Bailey 2005; Mehringer and Wigand 1990; Miller and Wigand 1994). The presence or absence of stumps, snags, and logs, and the ratio of old-growth to young woodlands and trees across the Intermountain West, strongly suggest both infill and expansion significantly increased in the late 1800s and early 1900s (postsettlement) before declining to slower rates (figs. 3-15, 3-16). In the northwest, Mehringer and Wigand (1990) reported a fivefold increase in pinyon and juniper pollen from the late 1800s to 1980. In the Colorado Plateau, Cole et al. (1997) reported that changes in woodlands during the Holocene were minor compared to the last 200 years. Expansion and infill across the Great Basin and Colorado Plateau occurred as a gradual increase in the 1700s and early 1800s but as a rapid increase in the late 1800s, with a peak frequently occurring in the early 1900s. The majority of dendrochronology studies conducted across the Great Basin and Colorado Plateau reported increased tree abundance along this timeline.

Expansion of postsettlement woodlands into presettlement sagebrush ecosystems in Oregon, Nevada, Idaho, and Utah (figs. 3-23, 3-24b; Miller et al. 2008; Strand et al. 2008; West 1984), aspen communities in southeastern Oregon, northeastern California, and northwestern Nevada (Wall et al. 2001), and riparian and meadow vegetation in southwestern Utah (Cottam and Stewart 1940) peaked between 1890 and 1920 (fig. 3-16). In many postsettlement woodlands, the existence of presettlement trees, logs, snags, and stumps are missing, or they are only found at low densities on ecological sites that were likely persistent sagebrush communities prior to the late 1800s (Johnson and Miller 2008; Miller and Heyerdahl 2008; Miller and Rose 1995, 1999; Miller et al. 2008; Nicol 1937; Weisberg et al. 2008).

There is also no evidence that mature pinyon or juniper were common in meadow, riparian, or aspen communities, prior to 1880 (Cottam and Stewart 1940; Wall et al. 2001). Expansion of pinyon and juniper occurred both up and down slope (Bradley and Fleischman 2008; Cottam and Stewart 1940; Hattori and Thompson 1987; West 1984; Woodbury 1947). However, Lanner and Frazier (2011) found no evidence of downslope movement of pinyon and juniper woodlands based on current boundaries with early observations of Coville (1893) and Wilson (1941). But these early observations did not clearly define the lower limits of woodlands. The greatest rates of expansion have occurred in the productive cool and moist mountain shrublands with mollic soils (Campbell, USDA NRCS data 2015; Johnson and Miller 2006; West 1984).

By the end of the 20th century, tree densities were more than 600 percent of the historic estimate in sagebrush ecosystems (Cottam and Stewart 1940; Gedney et al. 1999) and in desert grasslands once dominated by tobosa and blue grama grass (Margolis 2014). Since the 1920s, the rate of tree establishment has declined, but expansion and infill are still occurring in a generally warming and more arid climate. This appears to be a contradiction from the past when woodlands usually declined during shifts from cool and moist to warm and dry climates (Wigand 1987) and suggests that increasing atmospheric CO_2 concentrations are an important factor in current woodland expansion.

Tree Expansion vs. Reestablishment From Harvest in the Late 1800s

Tree expansion into previously nonwoodland communities across Nevada and Utah is somewhat clouded by the reestablishment in persistent woodlands harvested during the Comstock Mining era between 1863 and 1908, peaking between 1884 and 1891 (Hattori and Thompson 1987; Strachan et al. 2013). For example, tree-age chronology studies by Blackburn and Tueller (1970) occurred near historic mining sites and likely included both harvested and nonharvested areas, based on the presence or absence of presettlement trees within their plots (sidebar 5). Estimates of the amount of pinyon and juniper cut for charcoal fuel in the Great Basin during this Comstock era range from 600,000–750,000 acres (Lanner 1981; Young and Budy 1979), which represents 3.4–4.25 percent of the total acreage of woodlands in Utah and Nevada (fig. 3-25; Tueller et al. 1979). Modeling results and historical harvesting evidence indicate that heavy harvesting of pinyon and juniper primarily occurred close to the mining districts, and quickly decreased with distance (Ko et al. 2011). For example, in the Eureka, Nevada, mining district, half of the presettlement woodlands were not harvested 10 miles from the mine. Steepness of terrain also limited tree harvesting. Harvesting was greatest on slopes less than 15 percent and rapidly declined on slopes greater than 25 percent. Based on the tree-ring records, old stumps, and mining construction materials, Hattori and Thompson (1987b) concluded that severe historic deforestation depicted by historic accounts and photographs may have been exaggerated because of their proximity to mining settlements.

The proportion of young versus presettlement woodlands strongly supports significant pinyon and juniper expansion and infill beginning since the late 1800s. A USDA Forest Service inventory in Utah and Nevada estimated that woodlands between 40–120 years old accounted for 67 percent of all pinyon and juniper stands (O'Brien and Woudenberg 1999). In a recent inventory of Nevada's forests, 62

Sidebar 5

What happened to the stumps?
When old trees die or when stand-replacing fires occur, a considerable number of stumps, logs, snags, and large charcoal pieces remain on site and persist for centuries. But stumps are rarely found on many areas where trees were cut during the Comstock era. Lanner (1976) suggests trees were cut to ground level, but this is highly unlikely since this method quickly dulls a saw blade and is extremely hard on the back and knuckles. Another extremely labor-intensive possibility is that stumps were excavated. This may have occurred immediately adjacent to the mining sites (Hattori and Thompson 1987b) but is doubtful across the hundreds of thousands of acres where stumps are missing. Another possibility is rapid stump decomposition. Kearns and others (2005) reported pinyon snags only persisted up to 25 years. However, Hattori and Thompson (1987b) crossdated (see Glossary) intact pinyon stumps near the Cortez Mining District cut between 1864 and 1932. Wood density, ratio of sapwood/heartwood, and aridity of the ecological site influence the rate of decomposition as does the age of the tree. The disappearance of these stumps remains a mystery.

Figure 3-25—Between 600,000 to 750,000 acres of pinyon were harvested for charcoal in Utah and Nevada in the late 1900s for extracting precious minerals. This represents approximately 3.4 to 4.25 percent of the total acres of woodlands in these two states. Panamint Range, eastern California. (Photo by Rick Miller, Oregon State University.)

percent of pinyon and juniper woodlands were reported to be less than 150 years old and 71 percent of pre- and postsettlement woodlands were experiencing infill (Menlove et al. 2016). Young stands in the Shoshone Mountains in central Nevada accounted for 57 percent, mixed age stands 33 percent, and old-growth 10 percent of woodlands (Weisberg et al. 2008). And, old-growth stands had considerable infill. In eastern California, young woodlands (less than 150 years old) accounted for 80 percent of the total area occupied by woodlands (Bolsinger 1989). Other studies reported that over 50 percent of woodlands (fig. 3-23) and trees (fig. 3-16) in Oregon, Idaho, Nevada, and Utah were relatively young—less than 150 years (Bradley and Fleischman 2008; Miller et al. 2008; Weisberg et al. 2007, 2008).

Comparisons of recent and early surveys and records estimate increases in postsettlement woodland area of more than 600 percent in Arizona (Margolis 2014) and similar increases between 1938–1988 in Oregon (Gedney et al. 1999) and 1864 and 1930 in southwestern Utah (Cottam and Stewart 1940). Based on these estimates, young woodlands accounted for well over half of the more than 20 million acres in eastern Oregon, eastern California, Nevada, and Utah. This exceeds the estimates of harvested acres in the late 1800s by 27 times. The limited distribution and abundance of charred snags, stumps, logs, and large charcoal across the Great Basin and Colorado Plateau in many young woodlands suggest the majority are not the result of tree replacement across millions of acres of persistent woodlands after stand-replacing events, but instead represent widespread expansion through new tree establishment in recent centuries. However, considerable variation in the ratio of pre- and postsettlement woodlands occurs geographically and regionally, with a higher proportion of presettlement woodlands in the Colorado Plateau where fire occurrence is considerably less (fig. 3-18; Board et al. 2018; Littell et al. 2009).

Potential Factors Influencing Postsettlement Expansion and Infill

The accelerated rate of woodland infill and expansion in the late 1800s and early 1900s was primarily driven by climate (fig. 3-16). But in addition to climate's direct effects on seed production (Redmond et al. 2012), seedling establishment (Chambers 2000; Johnsen 1962; Smith et al. 1975), and mortality (Breshears et al. 2005), the changes are also closely linked to climates effects on disturbance such as fire, competition, insects, and disease. Disturbance plays an important role in limiting the distribution of pinyon and juniper woodlands in nonwoodland ecological sites suitable for tree establishment and growth (fig. 3-24b; Leonard et al. 1987; Wright et al. 1979). Where soils and climate are suitable for tree establishment and growth, the frequency and severity of these disturbances influence the distribution and persistence of pinyon and juniper, as well as sagebrush, riparian vegetation, aspen, and meadow communities.

Climate

The accelerated rate of tree infill and woodland expansion in the late 1800s and early 1900s (table 3-4; figs. 3-15, 3-16, 3-23) is largely attributed to climate (Barger et al. 2009; LaMarche 1974). Following the Little Ice Age, the climate generally became milder and remained wetter into the early portion of the 1900s (Antevs 1938; Biondi et al. 2011; Gray et al. 2004; Graumlich 1987; LaMarche 1973; Wahl and Lawson 1970). In the early 1900s, between 1905 and 1917, moist conditions were persistent and widespread across the nine western states and southwestern Canada (Woodhouse et al. 2005). During the second half of the 20th century, conditions have generally become warmer and drier (fig. 3-26; Ghil and Vautgard 1991; Mann et al. 2009; Marlon et al. 2012; Mote et al. 2013; Woolfenden 1996).

These climate changes in the second half of the 20th century coincided with a decline in the rate of tree expansion and infill after 1920 across much of the Great Basin and Colorado Plateau (figs. 3-16, 3-23; table 3-4). The initiation of increasingly drier conditions by the 1920s occurred earlier in the northern Great Basin than the Upper Colorado River Basin (fig. 3-12a,b; Schwinning et al. 2008; Miller et al. 2011). The Southwest was generally wetter than the Great Basin during the 1930s, experiencing sporadic drought years until a persistent dry period during the 1950s and early 1960s (Grissino-Mayer 1995; Miller et al. 2011).

However, with generally drier conditions in the latter half of the 1900s in both the Great Basin and Colorado Plateau, tree chronology studies (table 3-4) and satellite imagery (Bradley and Fleischman 2008) show continued trends in expansion of pinyon and juniper woodlands, which defies past periods where warming and drying resulted in woodland declines (Wigand 1987). The continued woodland infilling and expansion in relatively warmer and drier conditions in the second half of the 20th century may be attributed to reduced competition from perennial grasses and higher levels of CO_2, which likely improve water use efficiency in pinyon and juniper trees (Ivans et al. 2006; Knapp et al. 2001).

However, the rate and extent of 20th century pinyon and juniper expansion are spatially variable across the Colorado Plateau and Great Basin, as is recent woodland decline. In western Colorado, widespread tree canopy decline occurred in pinyon and juniper woodlands between 7,500–8,500 feet, and a widespread increase between 5,900–7,400 feet (Manier et al. 2005). In addition, significantly greater tree mortality from drought-related causes has occurred across the Colorado Plateau of twoneedle pinyon compared to limited losses of singleleaf pinyon in the Great Basin (Biondi and Bradley 2013; Clifford et al. 2008).
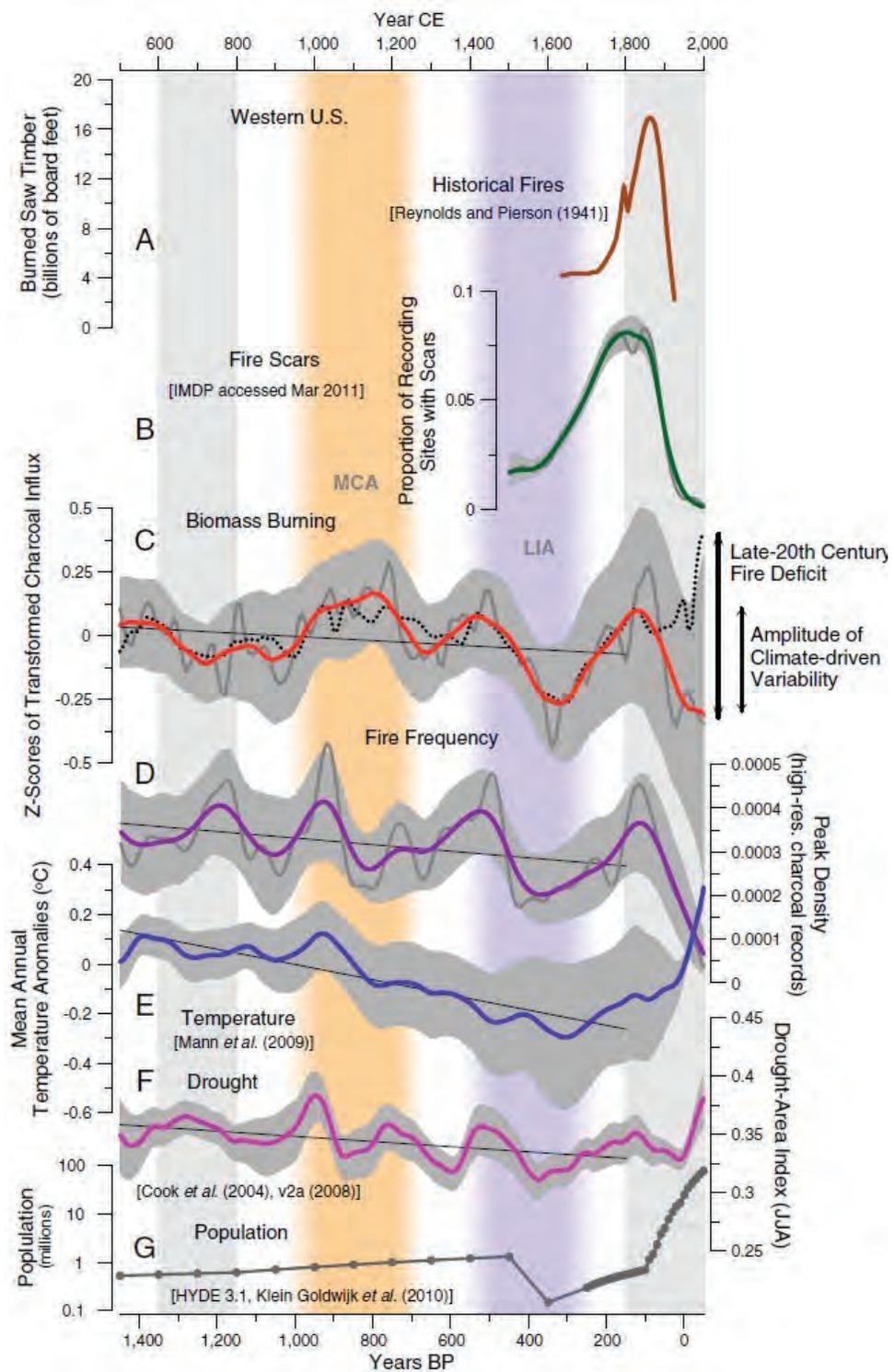
Figure 3-26—(A) Estimated historical saw timber affected by fires. (B) Smoothed proportions of dendrochronological sites recording fire scars (the green curve is based on locally fitting nearest-neighbor parameter of 0.25, while the gray curve is based on a parameter value of 0.10. (C) Smoothed and standardized 25-year (gray) and 100-year (red) trend line through standardized biomass burning records along with predicted biomass burning based on a GAM (black dashed line) fit to the 100-year biomass burning records. (D) Smoothed peak density (inferred fire frequency) from charcoal values (E) Smoothed gridded temperature anomalies for the western United States. (F) Smoothed Palmer Drought Severity Index for the western United States. (G) Population estimates for the western United States. All smoothed curves are plotted with 95 percent bootstrap confidence intervals (from Marlon et al. 2012).

Fire, Climate, and Grazing

Scientists disagree on the role and importance of fire and grazing in recent woodland expansion and infill across the Great Basin and Colorado Plateau (Baker 2011; Miller et al. 2011).

This is largely a result of:

1) Difficulties related to separating the direct and indirect effects of climate.

2) Unclear distinction often made between presettlement or persistent woodlands and recently expanded woodlands into historic nonwoodland communities.

3) Unclear distinction between recently expanded woodlands and reestablishment in persistent woodlands following stand-replacement events.

4) Geographic, regional, and local differences in disturbance regimes acting to control woodland expansion or contraction.

5) A wide range of variation in woodland structure, composition, and net-primary productivity (the rate at which all the plants in an ecosystem produce net useful chemical energy) in both time and space.

6) Issues of scale and landscape heterogeneity, especially as they relate to fire rotation and fire return interval interpretations.

7) Limited evidence of historical fire regimes in both time and space across semiarid plant communities, especially shrubland ecosystems.

Climate conditions in the late 1800s and early 1900s were ideal for cone production, tree-seedling establishment, and rapid growth rates (Eisenhart 2004; Fritts and Ziangdig 1986; LaMarche 1974; Redmond et al. 2012), especially during the persistently wet period in the West from 1905–1917 (Woodhouse et al. 2005). This wet period coincided with peak pinyon and juniper establishment reported in numerous chronological studies in the Great Basin and Colorado Plateau (table 3-4). But increased moisture would also have resulted in the accumulation of fine surface fuels and increased fuel continuity in the absence of heavy livestock grazing in the early 1900s. Antecedent wet years are especially important in increasing fuel continuity in communities with limited surface fuels and relatively long fire return intervals (Miller and Heyerdahl 2008; Miller and Rose 1999). Fine fuel accumulation in wet years has been closely linked to widespread large fires (Abatzoglou and Kolden 2013; Allen et al. 1995; Baisan and Swetnam 1990, 1997; Brown et al. 2008; Grissino-Mayer and Swetnam 2000; Miller and Rose 1999; Swetnam and Betancourt 1998). Unlike forested systems, fine fuel production was found to be more important than drought conditions for widespread fires in the semiarid West (Crimmins and Comrie 2004).

The majority of fire scar chronologies and tree-ring data indicate a decline in widespread surface fires in the West beginning in the late 1800s, which suggests postsettlement fires were largely eliminated (Allen et al. 2008a; Baisan and Swetnam 1990; Biondi et al. 2011). Fine fuel reductions by heavy livestock grazing during the late 1800s and early 1900s coincided with this dramatic decline in fire occurrence (Allen et al. 2008a; Baisan and Swetnam 1997; Gruell et al. 1994; Heyerdahl et al. 2006; Miller and Rose 1999; Savage and Swetnam 1990; Swetnam et al. 2001; Touchan et al. 1995). Climate changes alone were not likely sufficient to completely eliminate widespread surface fires (Swetnam et al. 2001). In addition to fine fuel reductions from grazing, fire suppression efforts and the near elimination of burning by Indians also likely contributed to the unprecedented decline in fire in the late 1800s across the Intermountain West. The sudden decline in fire occurrences in the late 1800s was like no other in the last 3,000 years (fig. 3-26; Marlon et al. 2012). The magnitude and importance of historical Native

American burning may have had implications from moderate to significant (Griffen 2002; Kay 2007; Keeley 2002; Pyne 1982; Robbins 1999; Stewart 2002) to minimal (Baker 2002; Vale 1975). Likely the degree of burning varied regionally (Swetnam et al. 2001) with tribal culture, fuels and types of vegetation, and food resources.

The impact of grazing on historic fire regimes also varied across the landscape, having little effect where surface fine fuels were insufficient to support fire spread, resulting in historically long fire return intervals (fig. 3-27; Barger et al. 2009; Huffman et al. 2008; Miller and Heyerdahl 2008; Waichler et al. 2001). The amount of woodland infill between grazed and ungrazed mesas in Utah was no different where fine surface fuels were already inadequate to support surface fires (5–6 percent cover) (fig. 3-27; Barger et al. 2009). In northern California, biomass of fine surface fuels was 100 pounds per acre in a presettlement woodland where the fire return interval exceeded 100 years (Miller and Heyerdahl 2008). In adjacent pine-fescue and mountain big sagebrush-fescue communities supporting 1,200 to 2,350 pounds per acre of fine surface fuels, mean fire return intervals were less than 15 years (Miller and Heyerdahl 2008). Fine fuel loads in mountain big sagebrush communities generally range between 1,000–3,000 pounds per acre (Mahalovich and McArthur 2004; Miller and Heyerdahl 2008; Passey et al. 1982; Shiflet 1994), which is more than adequate to support fire spread (Bunting et al. 1987; Evers et al. 2013; Gruell et al. 1986).

Grazing and Plant Competition

In the early 1900s, during his time with the USDA Forest Service in Arizona, Leopold (1924) wrote, "These ranges had never been grazed and they grazed them to death … the removal of the grass competition and fire damage brought in the reproduction [of trees]." Others also observed a decline in perennial grass cover from heavy grazing in the early 1900s, which coincided with the expansion of young woodlands (Emerson
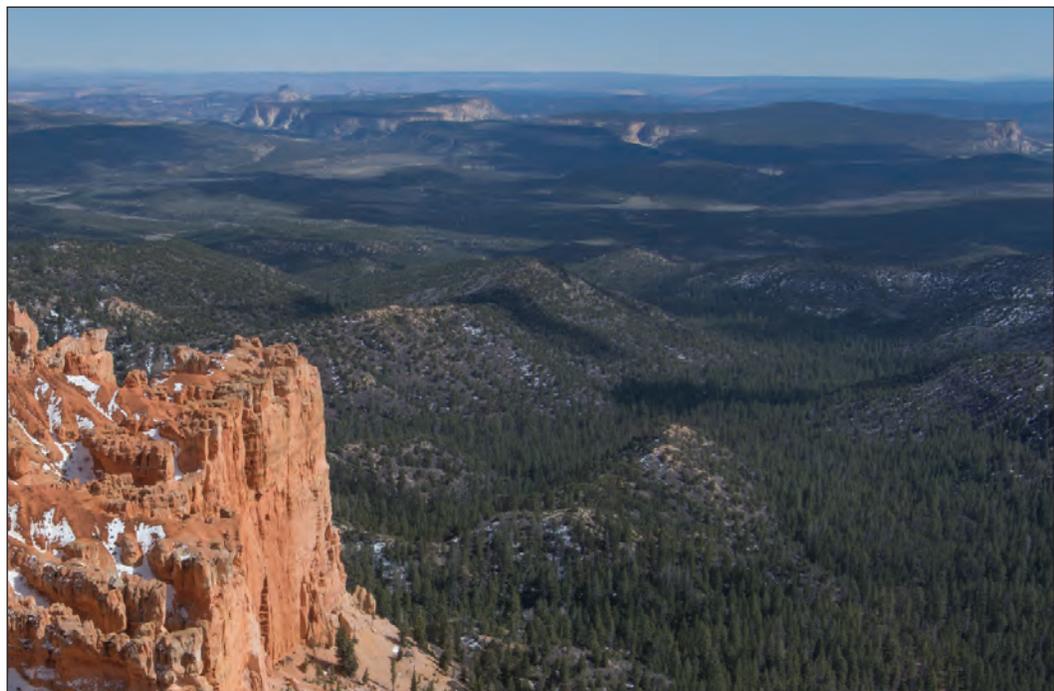
Figure 3-27—Just below the horizon are the ungrazed No-Man's mesa (left) and the grazed Deer mesa (right), where stand structures are similar. Both mesas historically had insufficient surface fuels to support fire spread, resulting in long fire return intervals, so grazing had little impact on the fire regime. Bryce Canyon looking south, Utah. (Photo by Rick Miller, Oregon State University.)

1932; Nicol 1937; Woodbury 1947). In several studies comparing historically grazed and ungrazed areas, pinyon and juniper density and canopy cover were greater in grazed areas (Guenther et al. 2004; Madany and West 1983; Shinneman and Baker 2009; Soulé et al. 2004). In the Grand Staircase Escalante National Monument, Utah, there was no difference in tree densities between grazed and ungrazed sites, but tree basal area was greater, and there was a significant reduction in perennial grasses on grazed compared to ungrazed high mesa sites (Barger et al. 2009; Harris et al. 2003).

Studies in ponderosa pine forests found that cool season grasses were very competitive with pine seedling establishment (Pearson 1934, 1942; Rummell 1951). Few small pines occurred in ungrazed areas, with a dense sod of perennial grasses compared to areas with depleted grass cover (Rummell 1951). Simulation modeling in woodlands across the Intermountain West identified grazing as a key factor in juniper encroachment by the way it reduced perennial grass and shrub cover, reduced fire occurrence, and disseminated pinyon and juniper seed (Caracciolo et al. 2016). Wright et al. (1979) proposed that the combined effects of presettlement droughts and perennial grass competition slowed the rate of woodland expansion and infill, which allowed longer fire return intervals to limit tree establishment. Depletion of perennial grasses and increased shrub canopy also augment woodland expansion by increasing the availability of safe sites for tree-seedling establishment. Shrubs provide safe sites for tree-seedling establishment by mediating temperatures and reducing competition from grasses (Burkhardt and Tisdale 1976; Johnsen 1962; Miller and Rose 1995; Soulé and Knapp 2000; Soulé et al. 2004).

Changing Fire Regimes Since 1980

Since 1980, when consistent and reliable reporting of wildfire ignitions and size began, total area burned and average size of individual fires across the West have increased (Brooks et al. 2004; Dennison et al. 2014; Littell et al. 2009; Miller et al. 2011; Morton et al. 2013; Westerling et al. 2006). This short record and high interannual variability make interpreting potential changes in fire regimes difficult in persistent woodlands where very long fire return intervals are typical. Still, there are indications that fire size and frequency have recently increased in pinyon and juniper woodlands (Board et al. 2018). Recent large fires in Mesa Verde covered 10 times the area they did in the past 150 years, and possibly the last 400 years (fig. 3-19; Floyd-Hanna et al. 2004). And the trend of larger areas burned in pinyon and juniper land cover types across the Intermountain West has significantly increased over the past 30 years (Board et al. 2018; Floyd et al. 2017). The increase is likely a result of increasingly warmer and drier conditions, 20th century increases in tree densities and canopies, and the increase in invasive annual grasses (Brooks et al. 2004; Westerling et al. 2006). The increase in erratic annual weather patterns influencing fuels and ignition can also influence fire regimes.

Postsettlement Woodland Decline

In the absence of stand-replacement disturbances, mortality in persistent pinyon and juniper woodlands can be less than 1 percent per century (Landis and Bailey 2005; Menlove et al. 2016; O'Brien 1999, 2002; Waichler et al. 2001). However, episodic stand-replacement or substantial thinning events at local and mesoscales do occur and are often linked to drought and drought-related disturbances (Furniss and Carolin 1977; Swetnam and Betancourt 1998). These episodic events are often connected to variations in climate, which are often synchronized with fire and insect infestations resulting in widespread tree die-offs (Breshears et al. 2005; Gaylord et al. 2013; Shaw 2006; Swetnam and Betancourt 1998). There have been 5 major droughts in the West during the 20th and 21st centuries (Betancourt et al. 1993; Breshears et al. 2005; Herford et al. 2002; Koepke et al. 2010).

The drought between 1942 and 1957 (fig. 3-12b) was the most severe in the past 400 years (Herford et al. 2002; Swetnam and Betancourt 1998). And the beginning of the 21st century began with a severe drought (from 2000 to 2003), resulting in 40-80 percent tree mortality across the Colorado Plateau (Breshears et al. 2005). A very recent report suggests the Southwest has been in a severe drought for the past 19 years (1999-2018) and much of the rest of the West for nearly the same time (Meyer 2018).

Although mortality can be relatively widespread, it typically differs among geographical regions and between pinyon and juniper. Die-offs during the drought of the early 2000s resulted in less than 10 percent mortality of singleleaf pinyon in the Great Basin (Biondi and Bradley 2013) compared to more than 50 percent in twoneedle pinyon across the Colorado Plateau (Breshears et al. 2005; Mueller et al. 2005a; Shaw et al. 2005). During this time, the large proportion of tree mortality was pinyon—with Utah, Rocky Mountain, and oneseed junipers suffering considerably less (Shaw et al. 2005). Mortality of mature reproductive twoneedle pinyon was 60 percent in Colorado, 74 percent in Arizona, and 94 percent in New Mexico—compared to less than 10 percent mortality in mature juniper (Floyd et al. 2009). The majority of pinyon trees were infested with bark beetles. Juniper had little infestation of insects and fungi. Dissimilarities in drought-related mortality between pinyon and juniper is largely attributed to differences in drought avoidance (Breshears et al. 2008a; Linton et al. 1998; Miller and Shultz 1987) and vulnerability to insect infestations (Floyd et al. 2009).

The relationship between drought-related mortality and tree density has been reported to range from minimal to significant (Clifford et al. 2008; Floyd et al. 2013; Greenwood and Weisberg 2008; McMillin et al. 2008; Negrón and Wilson 2008). In evaluating this relationship across spatially diverse woodlands, one must consider the capacity of the site to support trees—which varies considerably (Miller et al. 2000). Drought-related mortality in the Southwest may have been intensified by infill that has occurred across the majority of pinyon and juniper woodlands across the West during the 20th century (fig. 3-16; Barger et al. 2009; Biondi and Bradley 2013; Floyd-Hanna et al. 2004; Margolis 2014; Miller et al. 2008). The effect of elevation on drought-related mortality is also unclear. Greater mortality of twoneedle pinyon during the 2000-2003 drought was reported at lower elevations in Arizona, Utah, and Colorado (Cole et al. 2008a; McMillin et al. 2008). However, Breshears et al. (2008b) found elevation and mortality of twoneedle pinyon was positively correlated with elevation. Other site characteristics such as soil, seasonal patterns in precipitation, and temperature may be confounding the relationship between mortality and elevation.

SECTION 4: ECOHYDROLOGY OF PINYON AND JUNIPER WOODLANDS

Summary

The changes occurring throughout pinyon and juniper woodlands pose important ecohydrologic ramifications for plant communities (Pierson and Williams 2016; Wilcox and Breshears 1995). The broad variety of climate, soils, and topography in these ecosystems result in a wide range of hydrologic function. Woodlands can be generally classified either as "resource conserving" or "nonconserving" (Wilcox et al. 2003a). Resource conserving communities occur when ample vegetation and groundcover limit runoff and soil loss, promoting ecohydrologic resilience. Nonconserving or "leaky"

communities typically have extensive bare ground connectivity, which promotes runoff and soil loss. Woodlands intermediate to these two endpoints may transition from one conserving state to another following disturbance or drought events that changes the amount and structure of surface cover.

The resilience to remain or return to resource conserving is closely linked to an area's soils, climate, and topography. Throughout much of the semiarid Intermountain Region, pinyon and juniper are expanding into sagebrush ecosystems and infilling into persistent woodlands. Tree expansion and infill can result in reduced understory vegetation, increased connectivity of bare ground, and amplified runoff and soil loss.

Increased frequency of droughts and heavy fuel loads increase the threat of high-severity wildfires, which have significant adverse impacts on ecohydrological process. Initial ecohydrologic and erosion impacts of tree reduction by fire, mechanical treatments, or drought depend on: (1) the degree to which the vegetation and ground cover structure are altered, (2) initial site conditions, and (3) ecological site attributes.

In rain- to snow-dominated climate regimes, plot- and hillslope-scale studies suggest that burning commonly increases the risk of runoff and soil erosion-generating events by reducing surface cover structure and increasing connectivity of bare ground. This risk is likely greatest for sites commonly subjected to intense summer thunderstorms. Risk declines over time with vegetation and ground recovery (commonly within 5 years).

Topography and soils also influence potential erosion. The literature spanning all regions indicates that mechanical tree-removal treatments can initially improve infiltration and limit hillslope runoff and erosion if tree debris is sufficiently distributed into bare patches and in contact with the soil surface (Williams et al. 2019). Historic and recent research on drought-related vegetation transitions has documented transformative changes in ecohydrologic function that facilitate site degradation. These studies highlight the need to identify sites approaching the tipping points and management practices that increase resilience ahead of drought, fire, beetle infestations, invasive weeds, and other disturbance events. When ecological site characteristics are considered, successful tree-reduction treatments on woodland-encroached sagebrush rangelands can improve vegetation structure and ecohydrologic function. Improved ecohydrologic function on these landscapes further enhances the vegetation and ground cover structure and improves ecosystem resistance and resilience to invasive plants and disturbances.

Historical and current literature is inconclusive regarding tree-removal impacts on groundwater availability (Deboodt 2008; Kormos et al. 2017a; Ochoa et al 2018; Niemeyer et al. 2016). Studies conducted in the Great Basin and Colorado Plateau indicate tree removal can increase soil water along a hillslope over a broad range in annual precipitation. However, the additional amount of available water typically declines in the first few to 4 years after tree removal. The literature suggests the annual precipitation requirement for such enhancements in available groundwater with tree reductions likely occurs at ranges from 8 to over 16 inches (200 to over 400 mm) for predominately cold-season precipitation regions in the Great Basin. In the Southwest, very limited work would suggest 16 inches or more is required to see a response in soil water. The literature is also limited and inconclusive regarding tree-reduction impacts on streamflow.

Great Basin studies indicate tree reduction can affect patterns of snow accumulation and melt and in doing so, influence the timing of streamflow (Kormos et al. 2017b). Other studies from the Great Basin suggest tree reductions may have little impact on streamflow but can temporarily increase groundwater. Literature on impacts of tree-removal treatments to increase streamflow for the southern Colorado Plateau and southwestern United States have reported mixed results, and there is no clear indication that tree removal in pinyon and juniper woodlands on sites with rain-dominated precipitation regimes will foster long-term increases in streamflow. Recent studies of drought-induced tree die-off in woodlands and

forests of the Colorado Plateau have reported reductions to no change in streamflow. To date, there is little evidence that drought-related changes to vegetation in pinyon and juniper woodlands significantly alter water availability at the annual time scale, particularly for the rainfall-dominated southwestern United States. In the Great Basin and northwestern Colorado Plateau, large-scale die-offs have not been reported and therefore ecohydrologic impacts of drought cannot be assessed for those regions.

Introduction

Ecohydrology processes in pinyon and juniper woodlands are based on the same principles as other semiarid communities where water and soil retention or losses are governed by the structure, amount and cover of vegetation, inherent soil and topographic attributes, and climate (fig. 4-1; Davenport et al. 1998; Kormos et al. 2017a; Ludwig et al. 2005; Niemeyer et al. 2016, 2017; Petersen et al. 2009; Pierson et al. 2007, 2010; Reid et al. 1999; Roundy et al. 2014a, 2014b; Urgeghe et al. 2010; Wilcox et al. 1988, 1996a, 2003a; Williams et al. 2014a, 2016a,b, 2019). Vegetation and groundcover affect the soil microclimate (Breshears et al. 1997a, 1998; Lebron et al. 2007), enhance infiltration (Cline et al. 2010; Pierson et al. 2010, 2013, 2014; Reid et al. 1999; Roundy et al. 2017; Wilcox et al. 2003a; Williams et al. 2014a, 2016a, 2019) and soil water recharge and storage (Bates et al. 2000; Mollnau et al. 2014; Ochoa et al. 2018; Roundy et al. 2014a; Young et al. 2013a,b), and protect the soil surface from raindrop impact and the erosive energy of overland flow (Cline et al. 2010; Pierson et al. 2007, 2010, 2013, 2014, 2015; Reid et al. 1999; Roundy et al. 2017; Williams et al. 2014a, 2016a,b, 2019). In contrast, bare patches on these landscapes rapidly shed water and erode during intense rainfall events (Cline et al. 2010; Petersen and Stringham 2008; Petersen et al. 2009; Pierson et al. 2007, 2010; Reid et al. 1999; Wilcox 1994; Wilcox et al. 1996a; Williams et al.

Figure 4-1—Photographs from a Utah juniper woodland showing the patchy structure of tree-covered area and bare intercanopy between trees (A), a bare intercanopy runoff plot with concentrated overland flow and rilling (B), and shrub (C), under tree canopy (D), grass covered interspace (E), and bare interspace (F) microsites. The red arrow in photograph A depicts the general direction of concentrated overland flow that commonly occurs during runoff generating events on these landscapes. (Figure modified from Pierson and Williams 2016.)

2014a). Bare patches exhibit different seasonality in wetting, drying, and evaporative losses than vegetated patches, which can also affect overall landscape productivity (Breshears et al. 1997a, 1998; Newman et al. 2010).

The amount and structure of vegetation, inherent soil properties, and topography regulate responses to water input at hillslope to watershed scales (Davenport et al. 1998; Kormos et al. 2017a; Roundy et al. 2014a, 2017; Wilcox et al. 1996a, 2003a; Williams et al. 2014b, 2016a,b). Wilcox et al. (2003a) characterize these semiarid woodland landscapes as either "resource conserving" (fig. 4-2), where the vegetation structure and conditions capture water and buffer runoff and soil loss, or "nonconserving" (fig. 4-3) where the vegetation structure and conditions promote loss of water and soil—and lead to site degradation. In this section, we discuss these relationships for pinyon and juniper woodlands in the Great Basin and Colorado Plateau, including some coverage of fundamental hydrology and erosion processes for rangelands. This synthesis focuses on the annual water budget and associated hydrologic components of woodlands and includes erosion concepts in concert with overland flow and runoff processes. We do not separate content for presettlement and postsettlement woodlands, as the literature addressing hydrologic processes is too limited for such a separation in the context of broad coverage of pinyon and juniper hydrologic and erosion processes. We do, however, provide a brief discussion of woodland hydrologic and erosion responses to tree-removal practices (Brockway et al. 2002; Cline et al. 2010; Hastings et al. 2003; Jacobs 2015; Nouwakpo et al. in review; Ochoa et al. 2018; Owen et al. 2009; Pierson et al. 2007, 2013, 2014, 2015; Roundy et al. 2014a, 2017; Williams et al. 2014a, 2016a,b, 2019)—aimed at restoring woodland-encroached landscapes—which is also discussed in Section 5 (Bates and Davies 2016; Bates and Svejcar 2009; Bates et al. 2017b; Havrilla et al. 2017b; McIver and Brunson 2014; McIver et al. 2014; Redmond et al. 2013).

Figure 4-2—Phase II, which still maintains an intact understory is an example of resource conserving. However, as trees continue to fill the community, the understory will decline, increasing bare ground and becoming nonresource conserving. Modoc Plateau, northeastern California. (Photo by Rick Miller, Oregon State University.)

Figure 4-3—Phase III, with depleted herbaceous vegetation, declining shrub canopy, and high level of bare ground, is nonresource conserving. The high amount and connectivity of bare ground limits water capture, resulting in increased runoff, sediment loss and loss of nutrients. Schell Creek Mountains, eastern Nevada. (Photo by Rick Miller, Oregon State University.)

Water Budget in Pinyon and Juniper Woodlands

The water budget—the amount of water a plant community captures, stores, and releases—is input minus output and equals the changes in water storage. The ability of a plant community or landscape to capture and store water is closely linked to ecosystem resilience. Generally, the water budget is determined for a hydrologic water year. Water years are commonly defined as October 1 to September 30, as this time period marks the end of the growing season and the start of water input for the next growing season.

Precipitation

Pinyon and juniper woodlands extend across a large region of the Great Basin and Colorado Plateau where geology, soils, and climate—especially amounts and seasonality of precipitation—are highly variable (table 4-1; figs. 2-1, 2-2, 4-4, 4-5). Pinyon and juniper generally occur in areas that receive 10–16 inches of precipitation in the Intermountain Region (see Section 2 on woodland distribution and ecological site characteristics). However, there are some notable exceptions in the Southwest.

Table 4-1—Locations plotted in figure 4-1 and figure 4-4 and respective references.

Site name	State	Latitude	Longitude	Reference
Camp Creek (CC)	OR	43.96° N	-120.34° W	Deboodt et al. 2008; Ochoa et al. 2018
Reynolds Creek (RC)	ID	43.05° N	-116.43° W	Niemeyer et al. 2016
South Mountain (SM	ID	42.67° N	-116.90° W	Kormos et al. 2017b
Porter Canyon (PC)	NV	39.46° N	-117.62° W	Stringham et al. 2018
Canyonlands (CL)	UT	38.83° N	-109.84° W	West et al. 2007, 2008
Beaver Creek (BC)	AZ	34.37° N	-111.42° W	Clary et al. 1974; Zou et al. 2010
Cibeque Ridge (CR)	AZ	34.01° N	-110.20° W	Ffolliott and Gottfried 2012
Sevilleta (SV)	NM	34.23° N	-106.31° W	Limousin et al. 2013; Plaut et al.2012

USDA Forest Service RMRS-GTR-403. 2019.
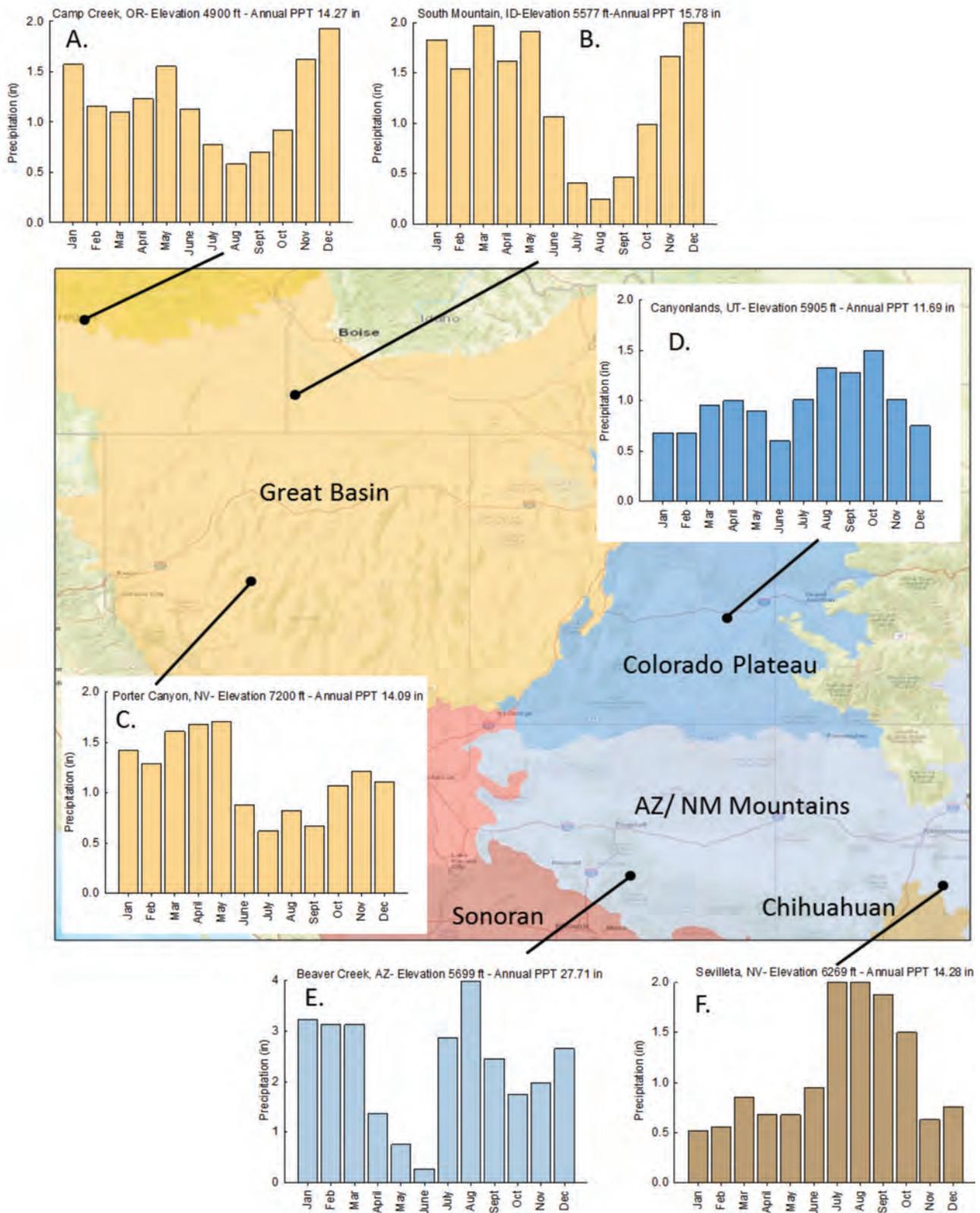
Figure 4-4—Mean monthly precipitation for six selected locations listed in table 4-1, inclusive of sites with a snow-dominated (A-C), mixed phase (D), and rainfall-dominated (E-F) precipitation regimes. Precipitation from 30-year monthly PRISM data, 800 m resolution (PRISM Climate Group 2018). (Figure modified from Williams et al. 2018b.)
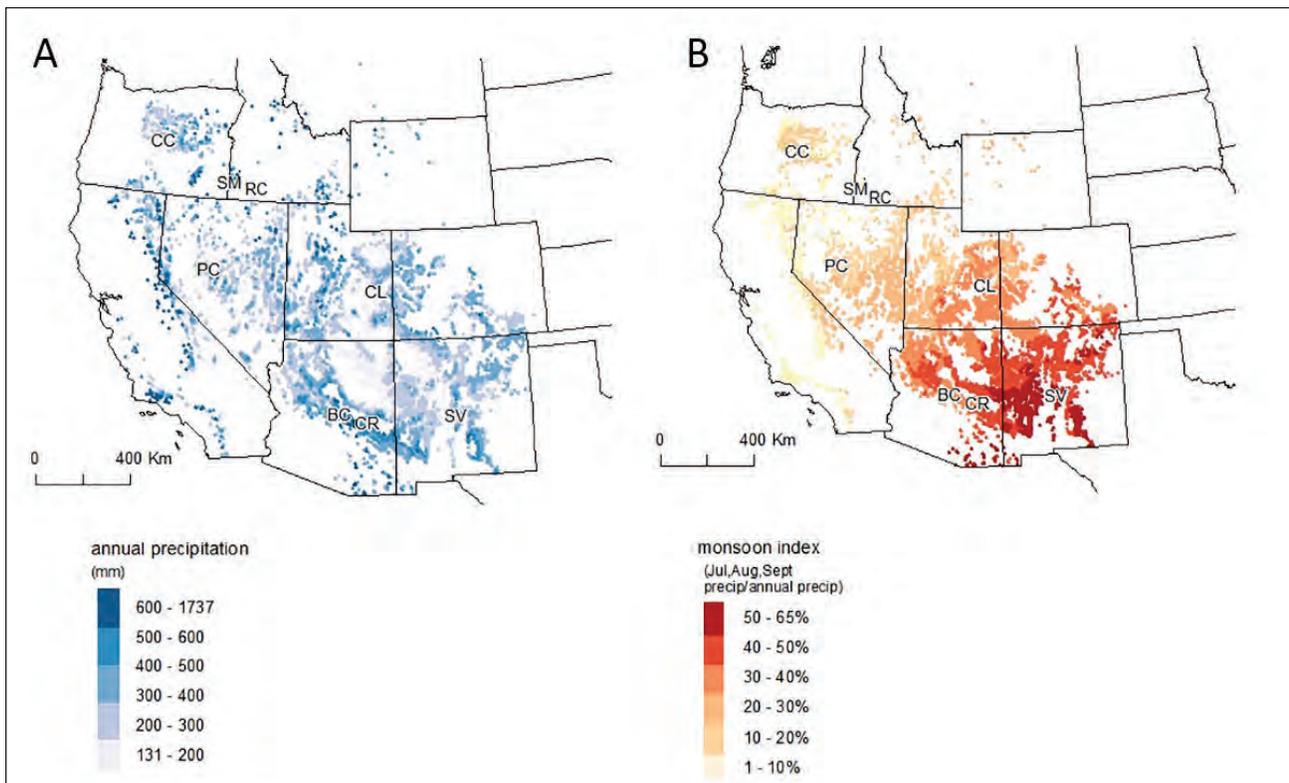
Figure 4-5—Maps of (A) average mean annual precipitation (shown in mm, 25.4 mm = 1 in) and (B) the monsoon index, which is the fraction of the annual precipitation that occurs in July, August, and September, across the pinyon and juniper domain in the western United States. Study sites from table 4-1 are depicted with site abbreviations. (Modified from Niemeyer et al. 2017; figure drawn by Tom Dilts, Great Basin Landscape Ecology Lab, University of Nevada-Reno.)

Precipitation in figure 4-4 is scaled the same for five locations on the y-axis, but Beaver Creek, Arizona (fig. 4-4e), receives nearly twice as much average annual rainfall. The seasonality of precipitation has been postulated to have important ecohydrological ramifications on woody plant expansion and treatments to remove woody plant species (Huxman et al. 2005). Specifically, it has been proposed that in regions where winter precipitation is the dominant input to the water budget (Mediterranean and cold-dominated climates), there is a greater potential to redistribute limited water resources with woody plant control. In these areas the growing season is largely decoupled from precipitation inputs so melting snowpack or winter rainfall are able to infiltrate and percolate to greater depths. In contrast, in summer-dominated rainfall systems, plants actively transpire shallow soil moisture delivered as rainfall (Huxman et al. 2005; Niemeyer et al, 2017). Alternatively, in these water-limited environments where pinyon and juniper exist, all water may be lost as evapotranspiration, regardless of seasonality.

This chapter reviews studies that address the ecohydrological implications of pinyon and juniper, which span these seasonally different regions. A map of the distribution of pinyon and juniper in these regions was redrawn from Niemeyer et al. (2017), which shows the range of mean annual precipitation (MAP) and percentage of rainfall that was delivered in July, August, and September (i.e., the monsoon index) across the regions (fig. 4-5).

Surficial Hydrology and Erosion Processes

Infiltration on Woodlands

Soil infiltration rates in woodlands are influenced by the amount and arrival rate of water at the ground surface; the ability of the soil to conduct water into and through

the soil; the slope, the roughness; and the chemical characteristics of the soil surface (Branson et al. 1981; Dingman 2014; Dunne and Leopold 1978; Hillel 1998). Infiltration is dynamic during an event, varying with changes in soil wetting and soil properties. In general, infiltration is more rapid in the early stages of water input into dry soil, decreases as the surface soil wets up, and approaches a relatively steady state (steady state infiltration rate) as soil becomes saturated (fig. 4-6).

Based on infiltrometer and rainfall simulation studies, infiltration rates and hydraulic conductivities for woodland soils vary at multiple scales within woodlands and across geographical regions (table 4-2; Cline et al. 2010; Lebron et al. 2007; Madsen et al. 2008; Petersen and Stringham 2008; Pierson et al. 2010, 2013, 2014; Roundy et al. 1978; Wilcox et al. 2003c; Williams et al. 2014a, 2019). Wilcox et al. (2003c) assessed unsaturated and saturated hydraulic conductivities of tuff-derived sandy loam to loam soils in canopy areas underneath trees and in vegetated and bare interspace areas. The assessments took place within the intercanopy between trees of twoneedle pinyon and oneseed juniper woodlands using infiltrometer (3-inch diameter) methodologies.
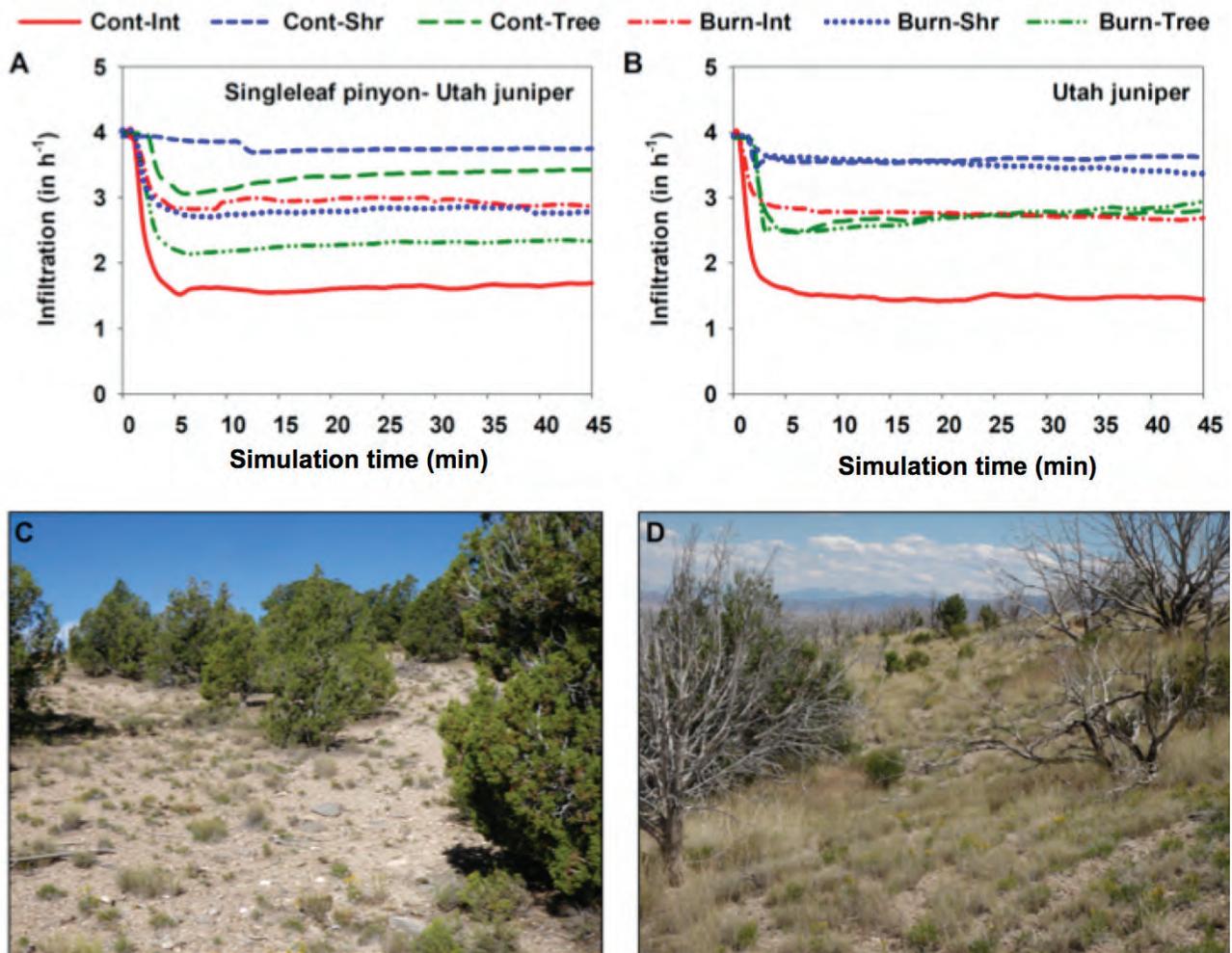
Figure 4-6—Infiltration (A and B) from rainfall simulation experiments (4.02 in h⁻¹, 45 min, 5.4 ft² plots) in untreated interspaces between shrubs and trees (Cont-Int), untreated shrub (Cont-Shr) and tree (Cont-Tree) canopy areas, burned interspaces (Burn-Int), and burned shrub (Burn-Shr) and tree (Burn-Tree) canopy areas in a singleleaf pinyon-Utah juniper woodland (A) and a Utah juniper woodland (B) 9 years following prescribed fire. Photographs at bottom of figure show the untreated Utah juniper woodland with extensive intercanopy area comprised largely of bare interspace and limited shrubs (C) and the same site 9 years after burning with good herbaceous cover within the intercanopy (D). Data from Williams et al. 2018a.

Table 4-2—Site characteristics, infiltration, runoff coefficients, and sediment yields reported for rainfall simulation and natural rainfall experiments in the western United States. Modified from Pierson and Williams (2016).

Study	Region (location) - community type	Microsite (plant)	Treatment /burn severity	Plot size (ft²)	Slope (%)	Time since trmt. (mth)	Rain rate (in h⁻¹) /duration (min)	Rain type	WDPT (s)[a]	Soil water (%)[b] /conditions	Bare soil (%)	Canopy cover (%)	Ground cover (%)	Avg. Infiltration (in h-1)	Runoff coef. (%)[c]	Sed. yield (t ac⁻¹)
Michaelides et al. 2009	Hot desert (Chihuahuan Desert, New Mexico) – mixed	Shrub canopy area (*Larrea tridentata*)	None	16	7	-	4.9/16–30	Artificial	-	7	79	65	21	-	55	0.43
O'Dea & Guertin 2003	Hot desert (Sonoran Desert, Arizona) – perennial grassland	Mixed grasses	Unb	323	3-Jan	-[d]	Variable	Natural	-	Dry	-	21	-	-	10	7.58[d]
		Mixed grasses	Mod	323	3-Jan	-[d]	Variable	Natural	-[e]	Dry	-	11	-	-	11	12.49[d]
		Mixed grasses	Unb	323	3-Jan	-[f]	Variable	Natural	-	Dry	-	37	-	-	14	8.03[f]
		Mixed grasses	Mod	323	1–3	-[f]	Variable	Natural	-[e]	Dry	-	29	-	-	14	9.81[f]
Craddock & Pearse 1938	Cold desert Great Basin, Idaho) - shrub steppe	Perennial bunchgrasses	None	217	30	-	1.8/60	Artificial	-	Dry	-	-	~ 35	-	1	~ 0.01
			None	217	30	-	3.6/30	Artificial	-	Dry	-	-	~ 35	-	1	~ 0.01
			None	217	40	-	1.8/60	Artificial	-	Dry	-	-	~ 35	-	< 1	~ 0.01
			None	217	40	-	3.6/30	Artificial	-	Dry	-	-	~ 35	-	< 1	~ 0.01
Craddock & Pearse 1938	Cold desert Great Basin, Idaho) - shrub steppe	Cheatgrass (*Bromus tectorum*)	None	217	30	-	1.8/60	Artificial	-	Dry	-	-	~ 20	-	10	0.14
			None	217	30	-	3.6/30	Artificial	-	Dry	-	-	~ 20	-	5	0.05
			None	217	40	-	1.8/60	Artificial	-	Dry	-	-	~ 20	-	26	0.22
			None	217	40	-	3.63	Artificial	-	Dry	-	-	~ 20	-	53	1.48
Craddock & Pearse 1938	Cold desert Great Basin, Idaho) - shrub steppe	Mixed grasses (*Lupinus* spp./*Stipa lettermanii*)	None	217	30	-	1.8/60	Artificial	-	Dry	-	-	~ 25	-	44	1.22
			None	217	30	-	3.6/30	Artificial	-	Dry	-	-	~ 25	-	57	2.28
			None	217	40	-	1.8/60	Artificial	-	Dry	-	-	~ 25	-	44	1.13
			None	217	40	-	3.6/30	Artificial	-	Dry	-	-	~ 25	-	55	2.09
Johnson & Blackburn 1989[g]	Cold desert Great Basin, Idaho) - shrub steppe	Sagebrush (*A. tridentata*)	None	351	10	-	2.5/60	Artificial	-	Dry	16	39	84	-	2	0.03
			None	351	6	-	2.5/60	Artificial	-	Dry	19	12	81	-	11	0.05
			None	351	9	-	2.5/60	Artificial	-	Dry	45	16	55	-	17	0.08
Pierson et al. 1994	Cold desert Great Basin, Idaho) - shrub steppe	Shrub canopy area (*A. tridentata*)	None	1	7	-	2.6/60	Artificial	-	4	0	-	100	-	2	~ 0.01
		Vegetated interspace	None	1	7	-	2.6/60	Artificial	-	4	4	-	96	-	23	0.1
		Bare interspace	None	1	7	-	2.6/60	Artificial	-	2	94	-	6	-	65	1.7
Pierson et al. 2002[h]	Cold desert Great Basin, Idaho) - shrub steppe	Shrub canopy area (*A. tridentata*)	Unb	5	35–60	12	2.6/60	Artificial	-	~14	7	88	93	-	11	~ 0.01
			Mod	5	35–60	12	2.6/60	Artificial	-	~ 5	97	11	3	-	34	0.13
			High	5	35–60	12	2.6/60	Artificial	-	~ 5	98	13	2	-	37	0.1
		Interspace	Unb	5	35–60	12	2.6/60	Artificial	-	~14	89	18	11	-	24	0.02
			Mod	5	35–60	12	2.6/60	Artificial	-	~ 5	95	16	5	-	26	0.05
			High	5	35–60	12	2.6/60	Artificial	-	~ 5	99	5	1	-	49	0.66

(Continued)

Table 4-2—(Continued).

Study	Region (location) - community type	Microsite (plant)	Treatment /burn severity	Plot size (ft²)	Slope (%)	Time since trmt. (mth)	Rain rate (in h⁻¹) /duration (min)	Rain type	WDPT (s)[a]	Soil water (%)[b] /conditions	Bare soil (%)	Canopy cover (%)	Ground cover (%)	Avg. Infiltration (in h-1)	Runoff coef. (%)[c]	Sed. yield (t ac⁻¹)
Pierson et al. 2001a, 2008b	Cold desert Great Basin, Nevada) - shrub steppe	Shrub canopy area (A. tridentata)	Unb	5	30–40	1	3.3/60	Artificial	200	7	1	100	99	-	30	0.05
			High	5	30–40	1	3.3/60	Artificial	102	1	99	1	1	-	37	0.18
		Interspace	Unb	5	30–40	1	3.3/60	Artificial	220	5	6	74	94	-	49	0.11
			High	5	30–40	1	3.3/60	Artificial	97	1	99	4	1	-	30	0.09
Reid et al. 1999	Temperate steppe mountains (Pajarito Plateau, New Mexico) – woodland	Vegetated interspace	None	22	6–12	-	Variable[i]	Natural	-	-	34	-	66	-	25[i]	1.33[i]
		Bare interspace	None	22	6–12	-	Variable[i]	Natural	-	-	84	-	16	-	37[i]	4.49[i]
		Tree canopy area (P. edulis or J. monosperma)	None	11	6–12	-	Variable[i]	Natural	-	-	0	-	100	-	8[i]	0.17[i]
Roundy et al. 1978	Cold desert (Great Basin, Nevada)– woodland	Shrub canopy area (A. tridentata)	Unb	9	5	-	3.3/60	Artificial	-	Dry	17	-	83	3.08	-	0.18
		Interspace	Unb	9	5	-	3.3/60	Artificial	-	Dry	-	-	-	2.27	-	0.42
		Tree canopy area (P. monophylla or J. osteosperma)	Unb	9	5	-	3.3/60	Artificial	Rep.	Dry	1	-	99	3.28	-	0.02
		Shrub canopy area (A. tridentata)	Burn	9	5	1–2	3.3/60	Artificial	-	Dry	80	-	20	3.06	-	0.28
		Interspace	Burn	9	5	1–2	3.3/60	Artificial	-	Dry	-	-	-	2.34	-	0.3
		Tree canopy area (P. monophylla or J. osteosperma)	Burn	9	5	1–2	3.3/60	Artificial	Rep.	Dry	25	-	75	3.2	-	0.08
		Shrub canopy area (A. tridentata)	Unb	9	5	-	3.3/60	Artificial	-	Wet	17	-	83	2.85	-	0.26
		Interspace	Unb	9	5	-	3.3/60	Artificial	-	Wet	-	-	-	1.48	-	0.38
		Tree canopy area (P. monophylla or J. osteosperma)	Unb	9	5	-	3.3/60	Artificial	Rep.	Wet	1	-	99	3.25	-	0.02
		Shrub canopy area (A. tridentata)	Burn	9	5	1–2	3.3/60	Artificial	-	Wet	80	-	20	2.48	-	0.47
		Interspace	Burn	9	5	1–2	3.3/60	Artificial	-	Wet	-	-	-	1.29	-	0.39
		Tree canopy area (P. monophylla or J. osteosperma)	Burn	9	5	1–2	3.3/60	Artificial	Rep.	Wet	25	-	75	2.94	-	0.09

(Continued)

Table 4-2—(Continued).

Study	Region (location) - community type	Microsite (plant)	Treatment /burn severity	Plot size (ft²)	Slope (%)	Time since trmt. (mth)	Rain rate (in h⁻¹) /duration (min)	Rain type	WDPT (s)[a]	Soil water (%)[b] /conditions	Bare soil (%)	Canopy cover (%)	Ground cover (%)	Avg. Infiltration (in h-1)	Runoff coef. (%)[c]	Sed. yield (t ac⁻¹)
		Shrub canopy area (*A. tridentata*)	Unb	9	8	-	3.3/60	Artificial	-	Dry	10	-	90	3.16	-	0.24
		Interspace	Unb	9	8	-	3.3/60	Artificial	-	Dry	-	-	-	1.62	-	0.65
		Tree canopy area (*P. monophylla or J. osteosperma*)	Unb	9	8	-	3.3/60	Artificial	Rep.	Dry	1	-	99	3.26	-	0.05
		Shrub canopy area (*A. tridentata*)	Burn	9	8	1–2	3.3/60	Artificial	-	Dry	45	-	55	3.14	-	0.35
		Interspace	Burn	9	8	1–2	3.3/60	Artificial	-	Dry	-	-	-	1.52	-	0.47
Roundy et al. 1978 *continued*	Cold desert (Great Basin, Nevada) – woodland *continued*	Tree canopy area (*P. monophylla or J. osteosperma*)	Burn	9	8	1–2	3.3/60	Artificial	Rep.	Dry	19	-	81	3.09	-	0.17
		Shrub canopy area (*A. tridentata*)	Unb	9	8	-	3.3/60	Artificial	-	Wet	10	-	90	2.92	-	0.36
		Interspace	Unb	9	8	-	3.3/60	Artificial	-	Wet	-	-	-	0.77	-	0.81
		Tree canopy area (*P. monophylla or J. osteosperma*)	Unb	9	8	-	3.3/60	Artificial	Rep.	Wet	1	-	99	3.24	-	0.04
		Shrub canopy area (*A. tridentata*)	Burn	9	8	1–2	3.3/60	Artificial	-	Wet	45	-	55	2.68	-	0.49
		Interspace	Burn	9	8	1–2	3.3/60	Artificial	-	Wet	-	-	-	0.99	-	0.98
		Tree canopy area (*P. monophylla or J. osteosperma*)	Burn	9	8	1–2	3.3/60	Artificial	Rep.	Wet	19	-	81	3.09	-	0.27
Pierson et al. 2007	Cold desert (Great Basin, Oregon) woodland	Intercanopy	None	350	19	-	2.2/60	Artificial	-	Wet	84	6	16	-	25	0.53
		Intercanopy – trees removed	Cut	350	19	-	2.2/60	Artificial	-	Wet	64	23	36	-	2	~0.01
Pierson et al. 2009	Cold desert (Great Basin, Idaho) shrubland-woodland	Shrub canopy area (*A. tridentata*)	Unb	5	35-50	1	3.3/60	Artificial	286	7	2	84	98	-	39	0.08
			Mod-High	5	35-50	1	3.3/60	Artificial	261	3	42	10	58	-	76	0.82
		Interspace	Unb	5	35-50	1	3.3/60	Artificial	110	3	25	31	75	-	63	0.87
			Mod-High	5	35-50	1	3.3/60	Artificial	117	4	84	0	16	-	55	3.15
Pierson et al. 2009	Cold desert (Great Basin, Idaho) shrubland-woodland	Intercanopy	Unb	350	35-50	1	3.3/60	Artificial	-	2	24	57	76	-	4	0.04
			Mod-High	350	35-50	1	3.3/60	Artificial	208	4	76	0	24	-	27	4.41

(Continued)

Table 4-2—(Continued).

Study	Region (location) - community type	Microsite (plant)	Treatment /burn severity	Plot size (ft²)	Slope (%)	Time since trmt. (mth)	Rain rate (in h⁻¹) /duration (min)	Rain type	WDPT (s)[a]	Soil water (%)[b] /conditions	Bare soil (%)	Canopy cover (%)	Ground cover (%)	Avg. Infiltration (in h-1)	Runoff coef. (%)[c]	Sed. yield (t ac⁻¹)
Pierson et al. 2013; Williams et al. 2014a	Cold desert (Great Basin, Idaho) woodland	Shrub canopy area (*Artemisia* spp.)	Unb	5	20	-	4.0/45	Artificial	<5	Wet	25	117	75	2.72	20	0.03
			High	5	18	11	4.0/45	Artificial	11	Wet	57	21	43	2.83	23	0.64
		Interspace	Unb	5	14	-	4.0/45	Artificial	<5	Wet	46[k]	20	54	1.5	63	0.16
			High	5	16	11	4.0/45	Artificial	<5	Wet	49[k]	21	51	1.97	51	0.6
		Tree canopy area (*J. occidentalis*)	Unb	5	21	-	4.0/45	Artificial	42	Wet	0	17[l]	100	3.03	23	0.03
			High	5	17	11	4.0/45	Artificial	54	Wet	50	5[l]	50	1.69	58	0.92
		Intercanopy	Unb	140	19	-	4.0/45	Artificial	-	Wet	28[m]	18	72	2.28	50	1.21
			High	140	16	11	4.0/45	Artificial	-	Wet	39[m]	32	61	2.28	50	2.55
		Tree canopy area (*J. occidentalis*)	Unb	140	16	-	4.0/45	Artificial	Rep.	Wet	7	26[l]	93	4.17	13	0.21
			High	140	18	11	4.0/45	Artificial	Rep.	Wet	25	15[l]	75	1.85	58	4.83
Pierson et al. 2010, 2015	Cold desert (Great Basin, Nevada) - woodland	Intercanopy	Unb	140	9	-	2.5/45	Artificial	-	6	26[n]	39	74	1.81	24	0.2
			High	140	9	10	2.5/45	Artificial	-	1	70[n]	23	30	2.37	6	0.11
		Tree canopy area (*P. monophylla or J.osteosperma*)[l]	Unb	140	9	-	2.5/45	Artificial	Rep.	11	3	27[l]	97	2.09	3	0.08
			High	140	9	10	2.5/45	Artificial	Rep.	2	19	6[l]	81	2.31	1	0.03
		Intercanopy	Unb	140	9	-	4.0/45	Artificial	-	Wet	26[n]	39	74	2.24	47	0.99
			High	140	9	10	4.0/45	Artificial	-	Wet	70[n]	23	30	2.4	40	1.54
		Tree canopy area (*P. monophylla or J. osteosperma*)[l]	Unb	140	9	-	4.0/45	Artificial	Rep.	Wet	3	27[l]	97	4.02	5	0.16
			High	140	9	10	4.0/45	Artificial	Rep.	Wet	19	6[l]	81	3.61	13	0.35
Pierson et al. 2010, 2015	Cold desert (Great Basin, Utah) - woodland	Intercanopy	Unb	140	14	-	2.5/45	Artificial	-	7	29[o]	19	71	2.28	10	0.17
			High	140	19	10	2.5/45	Artificial	-	7	43[o]	17	57	2.25	6	0.18
		Tree canopy area (*J. osteosperma*)[l]	Unb	140	15	-	2.5/45	Artificial	Rep.	9	8	21[l]	92	2.51	2	0.06
			High	140	19	10	2.5/45	Artificial	Rep.	6	41	3[l]	59	1.95	21	2
		Intercanopy	Unb	140	14	-	4.0/45	Artificial	-	Wet	29[o]	19	71	2.6	44	1.32
			High	140	19	10	4.0/45	Artificial	-	Wet	43[o]	17	57	2.18	41	2.19
		Tree canopy area (*J. osteosperma*)[l]	Unb	140	15	-	4.0/45	Artificial	Rep.	Wet	8	21[l]	92	4.41	10	0.29
			High	140	19	10	4.0/45	Artificial	Rep.	Wet	41	3[l]	59	2.18	52	8.44
Pierson et al. 2014	Cold desert (Great Basin, Nevada) - woodland	Shrub canopy area (*Artemisia* spp.)	Unb	5	11	-	4.0/45	Artificial	<5	Wet	12	93	88	3.66	4	0.03
			High	5	13	10	4.0/45	Artificial	<5	Wet	49	53	51	3.19	10	0.21
		Interspace	Unb	5	9	-	4.0/45	Artificial	<5	Wet	44[p]	33	56	2.36	41	0.1
			High	5	10	10	4.0/45	Artificial	<5	Wet	49[p]	30	51	2.13	46	0.18
		Tree canopy area (*P. monophylla or J. osteosperma*)	Unb	5	12	-	4.0/45	Artificial	48	Wet	0	7[q]	100	4.02	0	0
			High	5	15	10	4.0/45	Artificial	65	Wet	9	4[q]	91	2.68	28	0.21

(Continued)

Table 4-2—(Continued).

Study	Region (location) - community type	Microsite (plant)	Treatment /burn severity	Plot size (ft²)	Slope (%)	Time since trmt. (mth)	Rain rate (in h⁻¹) /duration (min)	Rain type	WDPT (s)[a]	Soil water (%)[b] /conditions	Bare soil (%)	Canopy cover (%)	Ground cover (%)	Avg. Infiltration (in h-1)	Runoff coef. (%)[c]	Sed. yield (t ac⁻¹)
Pierson et al. 2014	Cold desert (Great Basin, Utah) - woodland	Shrub canopy area (*Artemisia* spp.)	Unb	5	17	-	4.0/45	Artificial	< 5	Wet	18	69	82	3.58	8	0.15
			High	5	16	10	4.0/45	Artificial	< 5	Wet	44	28	56	2.48	29	0.98
		Interspace	Unb	5	19	-	4.0/45	Artificial	< 5	Wet	52[f]	19	48	1.77	56	1.04
			High	5	16	10	4.0/45	Artificial	< 5	Wet	38[f]	7	62	1.46	64	1.57
		Tree canopy area (*J. osteosperma*)	Unb	5	20	-	4.0/45	Artificial	88	Wet	7	22[q]	93	2.76	22	0.44
			High	5	21	10	4.0/45	Artificial	125	Wet	9	2[q]	91	2.99	18	1.31

[a]Water drop penetration time (WDPT) is an indicator of persistence and strength of soil water repellency as follows: < 5 s wettable, 5 to 60 s slightly repellent, 60 to 600 s strongly repellent (Bisdom et al. 1993). "Rep." indicates authors reported water repellent soil conditions but did not specifically provide a WDPT or other quantitative measure of soil water repellency for respective plots.

[b]Measured near the soil surface (< 5 cm depth).

[c]Runoff coefficient is equal to cumulative runoff divided by cumulative rainfall applied. Value is multiplied by 100 to obtain percentage.

[d]Cumulative runoff and sediment yield for period of 1 July 1998 to 1 October 1998 resulting from natural rainfall events (~ 3.9 in). Fire was in May 1998.

[e]Informal water drop tests showed no postfire soil water repellency at soil surface (O'Dea and Guertin 2003).

[f]Cumulative runoff and sediment yield for period of 1 July 1999 to 1 October 1999 resulting from natural rainfall events (4.2 in). Fire was in May 1998.

[g]Data shown are for experiments on initial dry conditions (dry run) at Coyote Butte, Nancy, and Summit sites, respectively, by row.

[h]Data presented from south-facing slopes solely.

[i]Runoff and erosion following natural rainfall events were monitored over a 26-month period.

[j]*J. occidentalis* trees removed from site by chainsaw cutting 10 years before rainfall simulation experiments.

[k]Bare ground was about 90 percent across unburned (42 percent rock and 46 percent bare soil) and burned (43 percent rock and 49 percent bare soil); ground cover column includes rock cover (and ash for burned plots).

[l]Trees removed from plot by chainsaw immediately before simulations.

[m]Bare ground was about 88 percent (60 percent rock and 28 percent bare soil) for unburned and 88 percent (45 percent rock, 4 percent ash, and 39 percent bare soil) for burned; ground cover column includes rock (and ash for burned plots).

[n]Bare ground was about 64 percent (38 percent rock and 26 percent bare soil) for unburned and about 86 percent (16 percent rock, < 1 percent ash, and 70 percent bare soil) for burned; ground cover column includes rock cover (and ash for burned plots).

[o]Bare ground was about 79 percent (50 percent rock and 29 percent bare soil) for unburned and about 81 percent (38 percent rock, < 1 percent ash, and 43 percent bare soil) for burned; ground cover column includes rock cover (and ash for burned plots).

[p]Bare ground was about 73 percent (29 percent cover and 44 percent bare soil) for unburned and 87 percent for burned (38 percent rock and 49 percent bare soil); ground cover column includes rock cover (and ash for burned plots).

[q]Trees removed from plot about 12 months before simulation as part of earlier study (Pierson et al. 2010).

[r]Bare ground ≥ 90 percent across unburned (38 percent rock cover and 52 percent bare soil) and burned (56 percent rock cover and 38 percent bare soil); ground cover column includes rock cover (and ash for burned plots).

The authors found that saturated hydraulic conductivity was statistically similar for undercanopy areas (5.91 in h^{-1}) and intercanopy areas (2.87 in h^{-1}) due to high variability in canopy area values, but that unsaturated conductivities were greater for canopy areas (0.20–1.50 in h^{-1}) than bare interspaces (0.08–0.39 in h^{-1}). Vegetated interspace unsaturated conductivities (0.12–0.71 in h^{-1}) were similar to both canopy areas and bare interspaces. The reported saturated and unsaturated hydraulic conductivities reflect wet and dry conditions, respectively, for the soil surface after removal of litter and duff, but with any soil crust intact.

In a similar infiltrometer (0.63-inch diameter) study on loamy sand soils of Utah juniper-twoneedle pinyon woodlands, Madsen et al. (2008) assessed unsaturated hydraulic conductivities and soil water repellency in litter-covered canopy areas underneath juniper and pinyon and in intercanopy areas with biological soil crusts. The study found soils underneath trees were hydrophobic and yielded an average unsaturated hydraulic conductivity of 0.68 in h^{-1}, with conductivity rates increasing with distance from the base of the tree (0.18–2.26 in h^{-1}). Unsaturated hydraulic conductivity varied substantially for the wettable or hydrophilic soil conditions in the intercanopy (~4.93 in h^{-1}) and was, on average, 3.52 in h^{-1} for a transitional zone between the tree canopy edge and the intercanopy and 5.98 in h^{-1} within the intercanopy beyond this transitional zone. Soil water content averaged 0.08 in^3 in^{-3} in canopy areas, 0.19 in^3 in^{-3} in the intercanopy, and 0.18 in^3 in the intermediate zone between tree canopies and the intercanopy.

As in the Wilcox et al. (2003c) infiltrometer study, litter and duff were removed before the infiltrometer experiments, but biological soil crusts were left in place. The trend of higher infiltration rates in canopy areas versus intercanopy areas in the Wilcox et al. (2003b) study reflect the effect of infiltration-inhibiting vesicular crusting (also see Blackburn 1975; Blackburn and Skau 1974; Pierson et al. 1994; Roundy et al. 1978) in the interspace. However, two other studies in Utah juniper reported greater infiltration in intercanopy than in canopy areas resulting from the effect of soil water repellency on infiltration into mineral soils beneath trees (Lebron et al. 2007; Madsen et al. 2008). Lebron et al. (2007) pointed out that the unsaturated infiltrometer experiments in the Wilcox et al. (2003c) study were conducted on wet soils immediately after the saturated infiltrometer experiments in that study and therefore may not capture the effect of soil water repellency on infiltration. Wilcox et al. (2003c) did not report on presence or absence of soil water repellency, but, as also noted by Lebron et al. (2007), soil water repellency is more the norm than exception under litter layers of juniper and pinyon trees (fig. 4-7; Madsen et al. 2008, 2011; Pierson et al. 2010, 2013, 2014; Williams et al. 2014a, 2016a, 2019). Neither the Lebron et al. (2007) nor Madsen et al. (2008) studies reflect the influences of interception and water storage by the canopy and litter layers in buffering soil water repellency effects on infiltration and runoff generation (Meeuwig 1971; Pierson et al. 2008a, 2009, 2010, 2013, 2014, 2015; Williams et al. 2014a, 2016a, 2019).

A number of rainfall simulation studies have provided infiltration estimates across diverse woodland conditions and demonstrate the partitioning effect of vegetation and groundcover on water availability for runoff on woodlands (table 4-2). Blackburn and Skau (1974) reported infiltration rates ranging 1.97–2.85 in h^{-1} for initially dry soil conditions and 1.71–2.79 in h^{-1} for initially wet soil conditions across singleleaf pinyon-Utah juniper and Utah juniper woodlands in Nevada. The rainfall application rate was 3.0 in h^{-1} over a 30-minute duration over a variety of plot sizes—spanning canopy and interspace areas—with most being 3 feet by 3 feet in size (Blackburn and Skau 1974).

Roundy et al. (1978) assessed infiltration into soils derived from volcanic parent material on alluvial fans with 5–8 percent slopes (table 4-2). The sites were dominated by singleleaf pinyon and Utah juniper. Rainfall was applied at a rate of 3.31 in h^{-1} for one
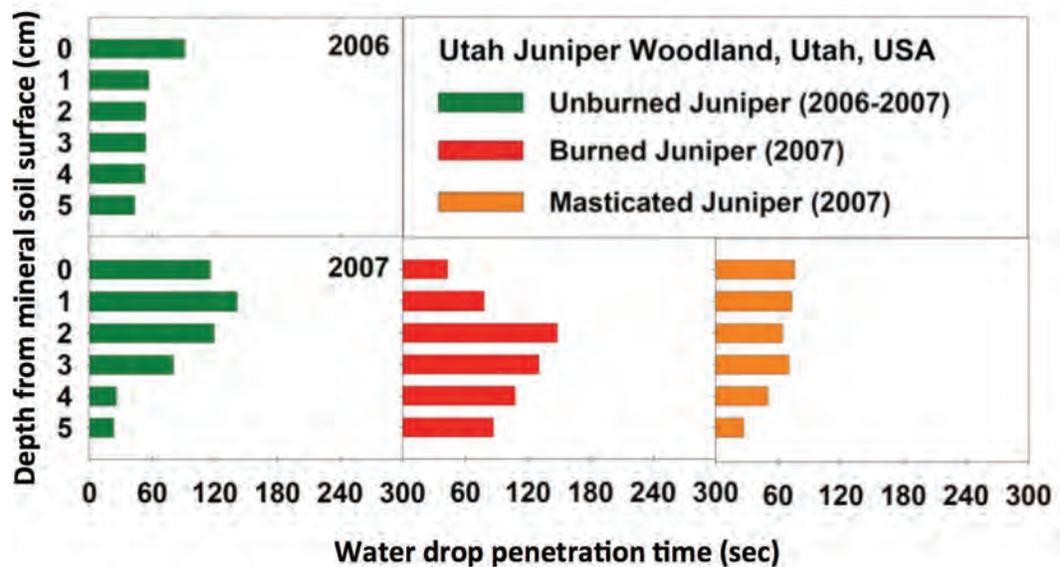
Figure 4-7—Persistence of soil water repellency in tree canopy areas of unburned, burned, and masticated Utah juniper as measured by the water drop penetration time (WDPT; Bisdom et al. 1993). Data from Pierson et al. (2010, 2014). Depth from mineral surface shown in cm (1 in = 2.54 cm), as presented by Pierson et al. (2010, 2014). Soils are water repellent where WDPT ≥ 5 s.

hour to plots approximately 3 feet by 3 feet in size and to variable-sized plots at dry and wet antecedent moisture conditions. Infiltration rates were generally similar for dry and wet soil conditions across tree canopy areas (3.27 in h^{-1}) and shrub canopy areas (2.99 in h^{-1}) respectively, but were substantially lower for interspaces under dry (1.93 in h^{-1}) and wet soil conditions (1.14 in h^{-1}) with respect to all canopy areas.

In a multi-year series of studies on singleleaf pinyon-Utah juniper, Utah juniper, and western juniper woodlands, Pierson et al. (2010, 2013, 2014) and Williams et al. (2014a, 2019) reported average infiltration rates under initially dry soil conditions ranging from 1.06–1.97 in h^{-1} for interspaces, 1.78–2.52 in h^{-1} for tree canopy areas, and 2.40–2.52 in h^{-1} for sagebrush canopy areas (table 4-2). The same studies reported average infiltration rates under initially wet soil conditions ranging from 1.06–2.36 in h^{-1} for interspaces, 2.68–4.02 in h^{-1} for tree canopy areas, and 2.72–4.02 in h^{-1} for sagebrush canopy areas. Rainfall was applied at 2.52 in h^{-1} for 45 minutes for the dry soil conditions and 4.02 in h^{-1} for 45 minutes for the wet soil conditions and plot sizes were approximately 5.4 ft^2. Soils in the Pierson et al. (2010, 2013, 2014) and Williams et al. (2014a, 2019) studies were strongly water repellent underneath pinyon and juniper trees and were wettable in interspaces and underneath shrub canopies.

In a companion study to Pierson et al. (2010, 2013, 2014) and Williams et al. (2014a, 2019), Cline et al. (2010) reported minimum and steady state infiltration rates were nearly threefold greater for vegetated (48 percent foliar cover) versus bare (3 percent foliar cover) interspace plots (approximately 5.4 ft^2), with rainfall applied on wet soil conditions at a rate of 4.02 in h^{-1} for 45 minutes. The above cited studies reflect the generally greater infiltration rates on litter-covered soils underneath trees relative to interspace soils on pinyon and juniper woodlands (table 4-2; fig. 4-6) and demonstrate the effect of the litter layer on buffering strong soil water repellency at the mineral soil surface underneath pinyon and juniper conifers. Likewise, the studies demonstrate that infiltration rates on woodlands decline with reduction of vegetation and groundcover, as reflected by the generally higher infiltration rates in canopy areas and vegetated interspaces relative to bare interspace areas (table 4-2).

This assertion is further supported by a rainfall simulation study by Petersen and Stringham (2008) on western juniper sites in Oregon. The study applied 4.02 in h^{-1} rainfall for 60 minutes under dry antecedent soil conditions to approximately ~2.7 ft^2 intercanopy plots on hillslopes with high (greater than 22 percent), moderate (13–16 percent), and low (less than 3 percent) juniper cover. Steady state infiltration for the low juniper cover hillslopes (3.54 in h^{-1}) was 34 percent and 68 percent greater, respectively, than measured on the moderate (2.36 in h^{-1}) and high (1.14 in h^{-1}) juniper cover hillslopes (Petersen and Stringham 2008). The combination of bare soil and rock cover in the study averaged 23 percent for low juniper cover, 63 percent for the moderate juniper cover, and 94 percent for the high juniper cover plots.

The studies herein clearly demonstrate water available for runoff processes on woodlands is concentrated primarily in the intercanopy between trees and is likely greatest in bare interspace areas. The infiltration rates presented above are likely largely affected by the rainfall application rates and durations, which vary within natural storms, but provide a basis for understanding the distribution of sink and source areas and the potential for runoff generation on woodlands.

Runoff From Woodlands

Plot-scale studies have demonstrated that the amount and type of runoff, as well as the connectedness of runoff sources, along a woodland hillslope are largely determined by magnitude of water input and the amount and continuity of bare ground (fig. 4-1a, 4-8; Pierson et al. 2007, 2010, 2013, 2015; Reid et al. 1999; Roundy et al. 2017; Wilcox et al. 1996a, 2003a; Williams et al. 2014a,b, 2016a,b, 2019). Table 4-2 summarizes results from rainfall experiments across the Intermountain West. Runoff from tree canopy
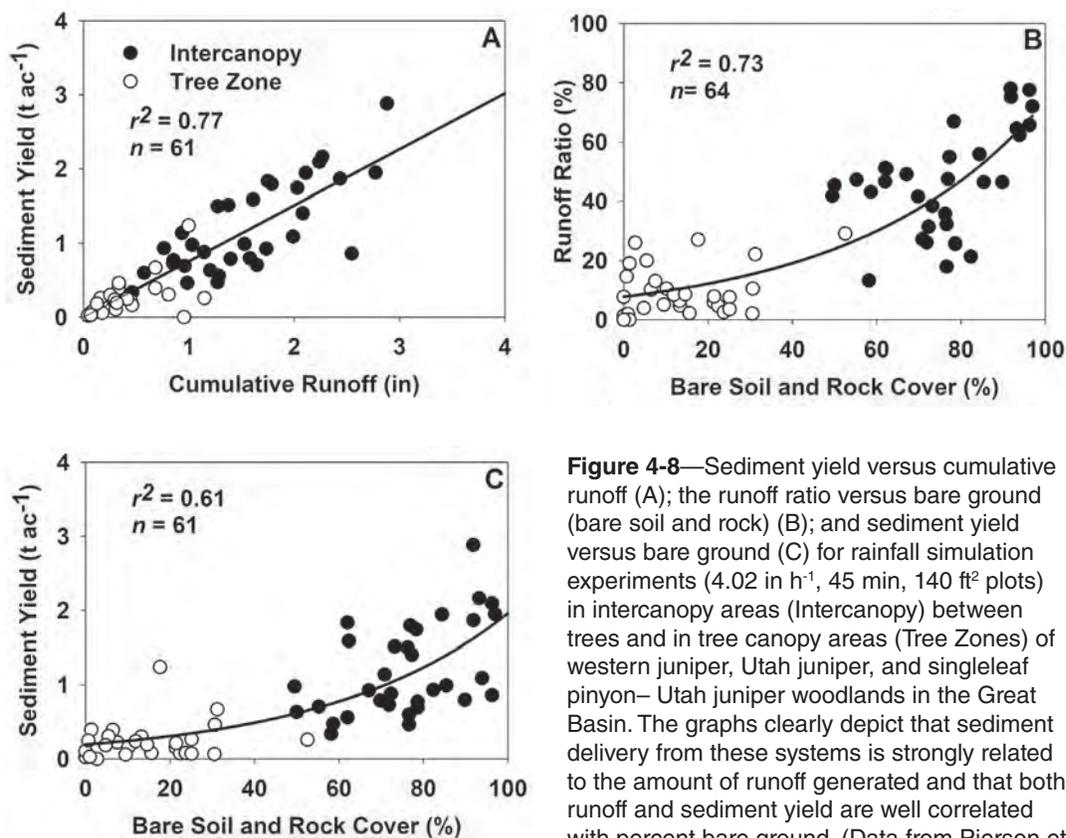
Figure 4-8—Sediment yield versus cumulative runoff (A); the runoff ratio versus bare ground (bare soil and rock) (B); and sediment yield versus bare ground (C) for rainfall simulation experiments (4.02 in h^{-1}, 45 min, 140 ft^2 plots) in intercanopy areas (Intercanopy) between trees and in tree canopy areas (Tree Zones) of western juniper, Utah juniper, and singleleaf pinyon– Utah juniper woodlands in the Great Basin. The graphs clearly depict that sediment delivery from these systems is strongly related to the amount of runoff generated and that both runoff and sediment yield are well correlated with percent bare ground. (Data from Pierson et al. 2010, 2013 and Williams et al. 2014a; figure modified from Williams et al. 2018b.)

areas in pinyon and juniper woodlands is often limited relative to that of the intercanopy between trees due to precipitation interception and water storage in the canopy and litter layers (table 4-2; fig. 4-8; Pierson et al. 2010, 2013, 2014, 2015; Reid et al. 1999; Wilcox et al. 2003a; Williams et al. 2014a;). In contrast, runoff generated in bare interspaces on woodlands is the primary source for runoff accumulation downslope unless captured by nearby vegetated or litter-covered patches (Davenport et al. 1998; Ludwig et al. 2005; Pierson et al. 2007, 2010, 2013; Reid et al. 1999; Wilcox et al. 2003a; Williams et al. 2014a, 2016a, 2019).

Wilcox (1994) found that 10–18 percent of the annual water budget over a 2-year period was converted to runoff, mostly during intense convective thunderstorms, in the intercanopy (approximately 325 ft^2 plots) of a gently sloping (5 percent) pinyon and juniper woodland in New Mexico. Wilcox (1994) also noted that both winter snowmelt and rain-on-snow events can generate runoff in pinyon and juniper woodlands, but that hillslope runoff risk is likely greatest during high intensity convective storms. Pierson et al. (2010) found that runoff from bare interspace areas on relatively large rainfall simulation plots within the intercanopy at 2 sloping (10–15 percent) Great Basin woodland sites contributed substantially to the formation of concentrated flow during high intensity rainfall application (4.02 in h^{-1}, 45 min; fig. 4-1b). The study found that cumulative runoff from the same simulated storm on small interspace plots was similar to the cumulative runoff measured on the large intercanopy plots, but that the runoff on the larger plots occurred mainly as high velocity concentrated overland flow.

Similar results were reported by Pierson et al. (2013) and Williams et al. (2014a) for a sloping (10–25 percent) western juniper woodland in which the same methodologies as Pierson et al. (2010) were applied. Collectively, the studies found that intercanopy runoff from applied high intensity storms was largely controlled by the combined amount of bare ground (bare soil and rock, averaged 64–89 percent) and increased where bare ground exceeded 50–60 percent (fig. 4-8). Pierson et al. (2010, 2013) and Williams et al. (2014a) measured mean intercanopy runoff rates of 0.28–0.55 in h^{-1} for 2.52 in h^{-1} rainfall intensity on dry soils and 1.97–2.20 in h^{-1} for 4.02 in h^{-1} rainfall intensity applied on wet soils over a 45-minute duration to the 140 ft^2 plots (table 4-2). In another rainfall simulation study, Pierson et al. (2007) found that large patches of bare interspace (averaged 91 percent bare ground) on approximately 350 ft^2 intercanopy plots within a sloping (19 percent) western juniper woodland facilitated formation of concentrated overland flow during 2.17 in h^{-1} applied rainfall over 60-minute duration (table 4-2). Runoff averaged approximately 0, 0.12, 0.20, 0.31, and 0.51 in h^{-1} at 5, 10, 15, 30, and 60-minutes into the simulations, representative of 2-, 4-, 8-, 50-, and 100-year storm events, respectively (Pierson et al. 2007).

Reid et al. (1999) measured seasonal runoff from natural rainfall in intercanopy and canopy areas of a sloping (approximately 10 percent) twoneedle pinyon-oneseed juniper woodland over a 26-month period (table 4-2). That study found that a substantial portion (37 percent) of the precipitation from rainfall events was converted to runoff in bare intercanopy patches and that 12 percent of the precipitation from these areas was recaptured as run-on in vegetated intercanopy patches downslope. Reid et al. (1999) estimated that tree canopy patches covered 50 percent of their study site and described intercanopy areas as a mosaic of patches devoid of vegetation and patches with relatively dense vegetation. The authors further noted that bare patch connectivity was limited at the site and that there was little indication of rilling (Reid et al. 1999).

Studies discussed above (Pierson et al. 2007, 2010, 2013; Reid et al. 1999; Williams et al. 2014a) contrast two different woodland intercanopy areas that function very differently hydrologically. Studies by Pierson and Williams document runoff responses for woodlands with extensive, well-connected intercanopy bare ground and high runoff—

and the third study by Reid documents substantial capture of runoff as run-on in an intercanopy with limited bare ground connectivity. Similar contrasting responses across multiple sites led Wilcox et al. (2003a) to characterize woodlands such as in the Pierson et al. (2007, 2010, 2013) and Williams et al. (2014a) studies with extensive bare ground connectivity and high runoff rates as "nonconserving" or "leaky" and those as in the Reid et al. (1999) study with patchy bare ground as "resource conserving." Suffice to say that woodlands of the Great Basin and Colorado Plateau exist across both conditions and therefore hillslope runoff behavior is quite variable across this vast domain. However, the general nonlinear trend of increasing patch-scale to hillslope runoff contributions with increasing bare ground and bare intercanopy connectivity is likely common for runoff generating storms on these landscapes (Davenport et al. 1998; Pierson et al. 2010, 2011, 2013; Urgeghe et al. 2010; Wilcox et al. 1996a, 2003a; Williams et al. 2014a,b).

Watershed scale studies on runoff from pinyon and juniper woodlands are more limited than plot-scale to hillslope scale studies. Wilcox (1994) provides a summary of numerous early (1960–1980 era) watershed scale (approximately 60 to more than 150,000 acres) runoff studies for southwestern U.S. pinyon and juniper woodlands with winter- and summer-dominated precipitation regimes. The summary reports that although runoff ranged from 2–23 percent of the annual water budget (11–21 inches) for the studies reviewed, runoff from southwestern U.S. pinyon and juniper woodlands generally amounts to less than 10 percent of the annual water budget (Wilcox 1994). Wilcox (1994) further noted that evapotranspiration is the dominant water loss mechanism on southwestern U.S. pinyon and juniper woodlands (fig. 4-9); that streamflow from these woodlands is typically ephemeral at the annual time scale; and that the seasonality of runoff for these landscapes is strongly related to the precipitation regime, with winter flows more common on snowy uplands and high summer flows occurring following intense thunderstorms.

More recently, Kormos et al. (2017b) summarized hydrologic data collected over a period of 6 years for 4 western juniper dominated (42–61 percent juniper canopy cover) experimental watersheds in southwestern Idaho. The watersheds range in size from
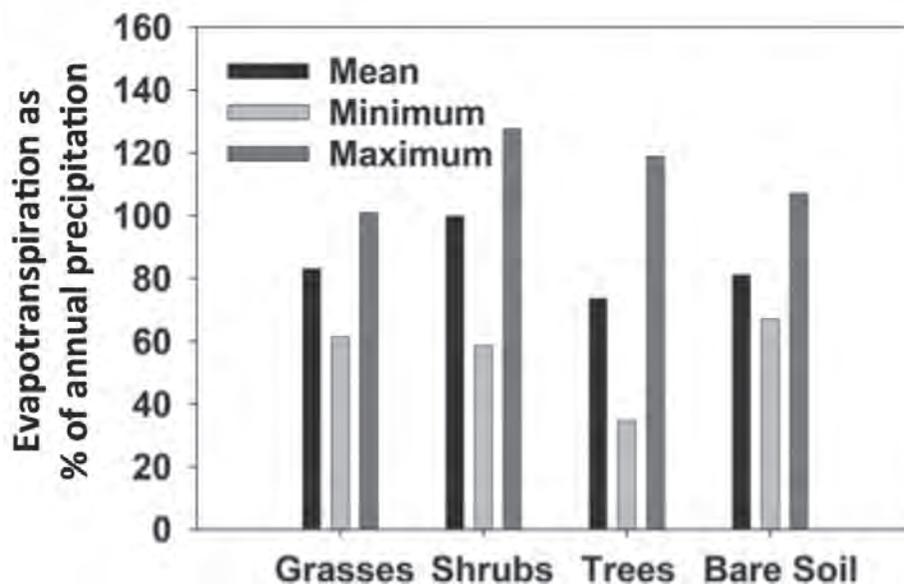
Figure 4-9—Estimated averages and ranges (as maximum and minimum) of annual evapotranspiration as a percentage of annual precipitation for rangeland plant types as estimated from the literature and reported in Pierson and Williams 2016.

approximately 50–175 acres and span hillslope gradients of approximately 20 percent. Annual precipitation at the sites (around 25 inches) occurs primarily during the winter and spring seasons and is therefore dominated (53–76 percent) by snowfall and mixed-phase events typical of inland northwest woodlands (Kormos et al. 2017b). Streamflow at the sites is ephemeral in response to snowmelt and rain-on-snow events, usually ceasing in late spring to mid-summer (Kormos et al. 2017b). Average annual streamflow across the four watersheds for the period of record was approximately 4.5 inches, or about 18 percent of the mean annual precipitation for the same period (Kormos et al. 2017b). Collectively, the Wilcox (1994) and Kormos et al. (2017b) studies characterize watershed scale runoff responses common to woodlands at the annual time scale spanning the rain-dominated, mixed-phase, and snow-dominated precipitation regimes in which pinyon and juniper woodlands occur (Caroll et al. 2016; Deboodt 2008; Kormos et al. 2017a,b; Niemeyer et al. 2017; Ochoa et al. 2018; Wilcox et al. 2003a). Although streamflow amounts to only a small portion of the annual water budget for these systems, the patchy structure of pinyon and juniper woodlands, particularly where degraded, exhibit limited buffering capacity to the most intense storms and can be subject to extreme runoff events (fig. 4-10a,b; Roundy and Vernon 1999).

Erosion From Woodlands

Rainfall simulation experiments provide reasonable estimates of pinyon and juniper woodland splash-sheet erosion (see Glossary for rainsplash and sheet erosion) occurring over fine spatial scales (table 4-2). Blackburn and Skau (1974) applied rainfall at 3 in h^{-1} over a 30-minute duration to a variety of plot sizes spanning canopy and interspace areas in pinyon and juniper woodlands in Nevada. They reported sediment yields ranging 0–0.42 t ac^{-1} for initially dry soil conditions and 0–0.62 t ac^{-1} for initially wet soil conditions across a diversity of soils. In another singleleaf pinyon-Utah juniper study, Roundy et al. (1978) found splash-sheet erosion from volcanic soils was substantially higher for interspaces (0.38–0.81 t ac^{-1}) relative to shrub canopy areas (0.18–0.36 t ac^{-1}) and tree canopy areas (0.02–0.05 t ac^{-1}) during 3.31 in h^{-1} rainfall simulations applied for one hour across dry and wet antecedent moisture conditions (table 4-2).

Rainfall simulation studies by Pierson et al. (2010, 2013, 2014) and Williams et al. (2014a, 2019) likewise reported higher splash-sheet erosion levels for interspaces than shrub and tree canopy areas across multiple sloping (10–25 percent) pinyon and juniper
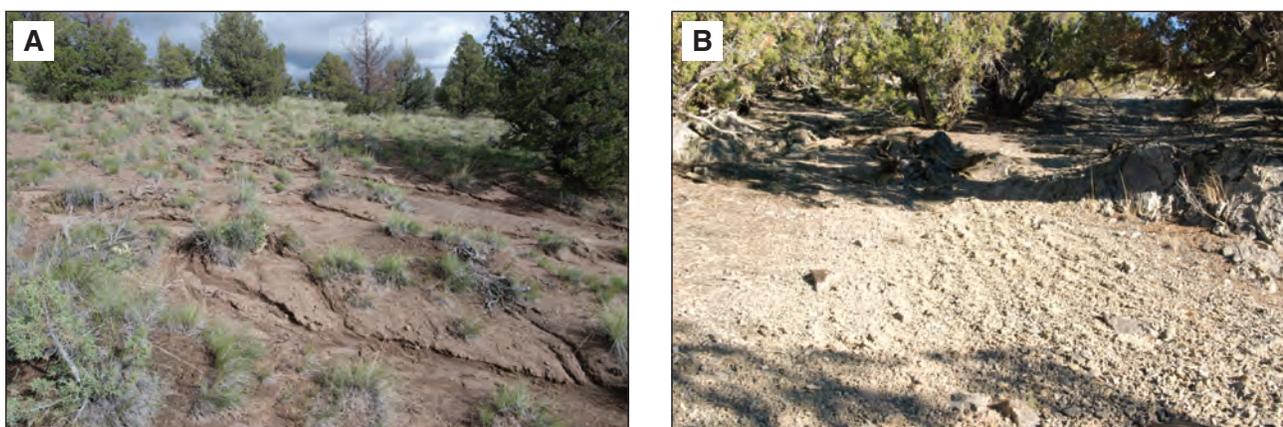
Figure 4-10—Sediment delivery on pinyon and juniper woodlands during runoff generating storms is largely controlled by the amount and intensity of precipitation and connectivity of bare intercanopy areas. (A) Although the understory vegetation is intact, a high-intensity storm still resulted in some soil loss and development of rills. (B) Increased connectivity of bare ground, especially where it exceeds 50 percent, results in high runoff and sediment loss. Some sites have lost the A horizon. Central Nevada. (Photos by Rick Miller, Oregon State University.)

woodlands sites in the Great Basin spanning soils from volcanic to sedimentary parent rocks (table 4-2). The studies applied rainfall rates on 5.4 ft^2 plots at 2.52 in h^{-1} for 45 minutes for dry soil conditions and 4.02 in h^{-1} for 45 minutes for the wet soil conditions. The authors reported average sediment yields for initially dry soil conditions ranging from 0.03 to 0.56 t ac^{-1} for interspaces, 0–0.24 t ac^{-1} for tree canopy areas, and 0–0.04 t ac^{-1} for sagebrush canopy areas.

The same studies reported sediment yields for initially wet soil conditions ranging from 0.16 to 1.70 t ac^{-1} for interspaces, 0–0.78 t ac^{-1} for tree canopy areas, and 0–0.21 t ac^{-1} for sagebrush canopy areas. The wide range in values for a given microsite (interspace, tree canopy, shrub canopy) and soil moisture condition (dry, wet) reflect different soil erodibilities across the study sites associated with the varying soil types, with soils derived of sedimentary rock (limestone and sandstone) having the highest erodibility (Pierson et al. 2010). The generally higher sediment yields from interspaces are due to higher bare soil exposure and runoff for interspace areas relative to the tree and shrub canopy areas (Pierson et al. 2010, 2013, 2014; Williams et al. 2014a, 2019). A western juniper woodland study by Petersen and Stringham (2008) provided limited specific data on sediment yields from intercanopy rainfall simulations on hillslopes representing a gradient of juniper dominance and bare conditions. That study found that 1-hour rainfall simulations on 2.7 ft^2 intercanopy plots at a 4.02 in h^{-1} intensity produced threefold more sediment from bare intercanopy plots (4.49 t ac^{-1}) than vegetated intercanopy plots (1.33 t ac^{-1}) on soils derived from volcanic parent rock.

Cline et al. (2010), using the same methodologies as described for Pierson et al. (2010) above, also reported a two- to threefold greater sediment yield for 2.52 in h^{-1} and 4.02 in h^{-1} rainfall simulations from bare (0.28 and 1.39 t ac^{-1}) than vegetated interspaces (0.07 and 0.59 t ac^{-1}) at Utah juniper woodland. The studies cited above clearly depict the typical distribution of sediment sources on pinyon and juniper woodlands, with sediment primarily generated from bare and vegetated interspaces and overall erodibility varying with soil type (Al-Hamdan et al. 2012b; Cline et al. 2010; Pierson et al. 2010, 2011; Roundy et al. 1978; Williams et al., 2014a, 2014b, 2016a, 2019).

Findings from plot-scale studies underscore that the potential for cross-scale sediment delivery on pinyon and juniper woodlands during runoff generating storms is largely controlled by the amount and connectivity of bare intercanopy area (table 4-2, fig. 4-8; Pierson et al. 2007, 2010, 2013, 2015; Reid et al. 1999; Wilcox et al. 2003a; Williams et al. 2014a, 2016a, 2019). As previously discussed, interspaces on woodlands are the primary sources for runoff and sediment, and, where well connected, these sources accumulate in concentrated flow paths (figs. 4-1b, 4-3) with high flow velocity, erosive energy, and sediment transport capacity (Al-Hamdan et al. 2012a, 2012b, 2013; Nouwakpo et al. 2016; Pierson et al. 2007, 2010; Williams et al. 2014a, 2016a, 2019).

Pierson et al. (2007, table 4-2) reported sediment yields of approximately 0, 0.02, 0.04, 0.13, and 0.52 t ac^{-1} at 5, 10, 15, 30, and 60 minutes (respectively) into rainfall simulations on approximately 350 ft^2 intercanopy plots at an application rate of 2.13 in h^{-1} in a western juniper study. The authors found that runoff from bare interspaces during concentrated flow experiments facilitated high flow velocities (approximately 0.33 foot s^{-1}) and sediment concentrations (approximately 2,200-2,600 ppm).

Similar studies of Utah juniper, singleleaf-Utah juniper, and western juniper woodlands by Pierson et al. (2010, 2013, 2015) and Williams et al. (2014a, 2016a) found intercanopy sediment yield from high intensity rainfall simulations (140 ft^2 plots) increased nonlinearly with bare ground where bare ground exceeded 50–60 percent due to formation of high velocity concentrated flow paths. Those studies reported intercanopy flow velocities ranging 0.20 to 0.66 ft s^{-1} for concentrated flow experiments with flow

releases of approximately 4, 8, and 12 gal min^{-1}. High runoff rates on intercanopy plots during rainfall simulation experiments in the study transported interspace generated sediment downslope in concentrated flow resulting in cumulative sediment yields ranging 0.16–0.20 t ac^{-1} and 0.99–1.79 t ac^{-1} for simulations at 2.52 in h^{-1} (dry conditions) and 4.02 in h^{-1} (wet conditions) intensities, 45-minute durations (Pierson et al. 2010, 2013, 2015; Williams et al. 2014a, 2016a).

In contrast, concentrated flow paths were limited on litter-covered tree canopy areas and sediment yield from the same simulations on these plots ranged 0.04–0.08 t ac^{-1} and 0.16–0.35 t ac^{-1} for the lower and higher intensity rates, respectively (Pierson et al. 2010, 2013, 2015; Williams et al. 2014a, 2016a). The Pierson et al. (2010, 2013) and Williams et al. (2014a, 2016a) studies found that intercanopy sediment yield increased with increase plot scale (from 5.4 ft^2–140 ft^2) even though runoff rates were similar across plot scales. The authors attributed the increased sediment delivery across spatial scales within the intercanopy to the concentration of overland flow at the larger plot scale, indicative of a "nonconserving" system as described for degraded woodlands in the southwestern United States (Davenport et al. 1998; Wilcox et al. 1996a, 2003a). Reid et al. (1999) and Wilcox et al. (2003a) describe conditions measured in "resource-conserving" woodlands whereby isolated runoff from bare patches is captured in downslope vegetated patches and hillslope sediment loss is limited.

Wilcox (1994) found that erosion from pinyon and juniper woodlands in the southwestern United States is generally higher in the summer in association with high intensity monsoonal thunderstorm events relative to the winter-season runoff from snowmelt with no raindrop impact. That study reported summer season sediment yields of approximately 0.01–1.28 t ac^{-1} and winter-season sediment yields of approximately 0.01–0.05 t ac^{-1} for approximately 325 ft^2 natural runoff plots (5 percent slope, 15 percent bare ground, undisturbed plots only) over a 2-year period (Wilcox 1994). Annual watershed scale erosion estimates for pinyon and juniper woodlands are largely absent from the literature and can vary greatly with land use and disturbance, topography, and soil type (0.08–4.46 t ac^{-1}; see Ffolliott and Gottfried 2012; Hastings et al. 2003; McAuliffe et al. 2014; Wilcox et al. 1996a, 1996b, 2003a), so current estimates are heavily biased by a few studies. Overall, hillslope contributions of sediment to channels and the watershed scale are dictated by the runoff and sediment source and sink structure and are greatest for high magnitude runoff events on sites with well-connected bare intercanopy area (50–60 percent bare ground) and an ample sediment supply (Davenport et al. 1998; Pierson et al. 2011; Wilcox 1994; Wilcox et al. 1996a, 1996b, 2003a; Williams et al. 2014a,b, 2016a).

Soil Water

Vegetation structure affects soil water availability by influencing soil water capture and moderating the soil microclimate through shading from solar radiation and insulating surface soils (Breshears et al. 1997a, 1997b, 1998; Lebron et al. 2007; Pierson and Wight 1991). For example, Pierson and Wight (1991) reported interspace locations in a sagebrush community had higher (by 41.4 ˚F) maximum and lower (by 34.7 ˚F) minimum near-surface (0 to 4-inch depth) soil temperatures than canopy locations during the spring season. Small grass and moss clumps within interspaces had little influence on near-surface soil temperatures, but shrub cover and the associated litter mounds insulated the soil surface from incoming solar radiation during daylight hours and from sensible heat loss at night (Pierson and Wight 1991).

Breshears et al. (1997a) found that interspaces between tree canopies of a twoneedle pinyon-oneseed juniper woodland exhibited greater (40–50 percent more) near-surface

solar radiation than tree canopies, and that preferential shading on the northern side of tree canopy areas significantly reduced near-surface solar radiation. Breshears et al. (1997a) also determined that snow water equivalent was greater in interspace locations than under tree canopies and that the differential accumulation resulted in temporal variability in the spatial arrangement of soil water. Soils in tree canopy areas were wetter than interspace soils in early winter following complete melt of canopy area snowpacks and during the monsoon season immediately after runoff events. Wetter soil conditions on the edges of canopy areas compared to interspaces following monsoon rain events were assumed to be partially related to runoff from interspace locations to canopy areas. Interspace soils were wetter than canopy area soils later in the winter and in early spring during the interspace snowmelt period. The differential snow accumulation and melt patterns, related in large part to canopy snow interception, exerted a greater influence on the spatial distribution of soil water than did the effects of preferential shading (Breshears et al. 1997a). The main effect of solar radiation on soil moisture patterns was observed within interspace patches, where north edges of the intercanopy with greater solar radiation were wetter than the shaded south edges during winter and spring. Breshears et al. (1998), working at the same site as Breshears et al. (1997a), found that maximum air temperature was as much as 50 °F greater on interspaces than tree canopy areas during late spring through summer and that the associated differences in spatial temperature produced differences in soil evaporation. The authors suggested that spatial differences in soil temperature affected soil evaporation only when soils were thawed and were amplified at lower soil water contents.

Several authors have postulated that soil water repellency underneath pinyon and juniper canopies may provide water conservation and increased plant productivity for these conifers (Lebron et al. 2007; Madsen et al. 2008; Robinson et al. 2010). Lebron et al. (2007) and Madsen et al. (2008) observed that surface water on water repellent soils under Utah juniper and twoneedle pinyon canopies was routed through the water repellent layer into deeper soil layers via bypass or preferential flow in isolated wet spots. Robinson et al. (2010) likewise found soil water repellency under Utah juniper and twoneedle pinyon concentrated infiltration of summer precipitation to undercanopy soils via preferential flow. Roundy et al. (1978) hypothesized similar behavior to explain rapid infiltration of simulated rainfall into water repellent soils of Utah juniper (table 4-2).

Water availability deep in the soil profile favors woody plant recruitment and increased plant productivity through greater water availability and transpiration rates and recruits surface plant and litter biomass associated with higher infiltration rates (Huxman et al. 2005; Ludwig et al. 2005; Pierson et al. 2010, 2013, 2014, 2015; Wilcox et al. 2003a; Williams et al. 2014a, 2016a,b). Therefore, surface flow routing by soil water repellency may function similar to the lateral surface transfers of overland flow (run-on) in maintaining shrub and woodland tree islands of higher biological activity and water retention (Bhark and Small 2003; Breshears et al. 1997a; Huxman et al. 2005; Ludwig et al. 2005; Reid et al. 1999; Robinson et al. 2010; Schlesinger et al. 1990; Wilcox et al. 2003a). Overall, soil water in pinyon and juniper woodlands is affected by above and belowground physical and biological attributes that regulate spatial variability in water input, storage, and use (Breshears et al. 1997a, 1997b, 1998; Deboodt 2008; Mollnau et al. 2014; Ochoa et al. 2008).

Evapotranspiration

Interception

Plant canopies physically intercept precipitation and the intercepted precipitation is either lost to the atmosphere by evaporation/sublimation or is subsequently transferred to the soil surface as throughfall or stemflow. Interception is strongly influenced by the precipitation intensity, duration, and frequency and by the type and structure of the

vegetative community (Branson et al. 1981; Owens et al. 2006; Dunkerley 2008). The percentage of event gross rainfall captured by cover elements generally decreases as rainfall intensity and duration increases (table 4-3; Branson et al. 1981; Carlyle-Moses 2004; Owens et al. 2006).

Literature on snowfall canopy interception specific to rangeland plants is extremely limited. Hull (1972) and Hull and Klomp (1974) found that dense shrub cover (2.2 plants per 10.8 ft^2) intercepted 37 percent of snowfall at a rangeland site in Idaho. In a twoneedle pinyon-oneseed juniper woodland site in New Mexico, Breshears et al. (1997a) found that snow accumulation during each of three winter seasons was much greater in intercanopy areas between tree canopies than underneath tree canopies. Snow water equivalent in the first year of the 3-year study was about 80 percent greater in the intercanopy openings between trees than in areas underneath canopies. Whole plant interception of wind-distributed snow by rangeland vegetation is paramount in retaining snow against wind scour (Flerchinger and Cooley 2000; Flerchinger et al. 1998; Kormos et al. 2017a; Seyfried and Flerchinger 1994; Seyfried and Wilcox 1995; Marks and Winstral 2001; Marks et al. 2001, 2002; Winstral and Marks 2002). Wind and topography interact to redistribute fallen snow on undulating terrain, while vegetation reduces wind velocities and facilitates deposition (Marks et al. 2001, 2002; Marks and Winstral 2001; Winstral and Marks 2002).

Deeper snow accumulations provide greater insulation for surface soils and plant productivity and prolong snow-covered periods (Liston et al. 2002; Sturm et al. 2001). The vegetation snow-holding capacity is a function of the vegetation height, density of plants, and snowpack conditions (Liston and Sturm 1998; Pomeroy and Gray 1995; Sturm et al. 2001). Hutchinson (1965) found that a shrub stand 20 inches in height stored 1 inch more water than an adjacent area void of shrubs. Flerchinger et al. (1998) reported that snow depth at a wind-driven rangeland site in Idaho typically varied by plant community from less than 24 inches in low sagebrush/grass to 40 inches in mountain big sagebrush-snowberry and 40–315 inches in an aspen-willow stand.

One of the reasons that pinyon and juniper expansion may lead to ecological degradation (Peterson and Stringham 2009) and loss of herbaceous and shrub cover both under tree canopies (Tausch et al. 1981; Miller et al. 2008) and in the intercanopy areas (Bates et al. 2000; Bybee et al. 2016; Davenport et al. 1998) is reduced precipitation inputs due to interception. Savenije (2004) makes a case for interception being the most overlooked component in rainfall and runoff analyses. At the whole tree level, liquid rainfall interception was measured using simulated rainstorms ranging from 0.09 inches–1.02 inches on singleleaf pinyon and Utah juniper (Stringham et al. 2018). Five storms targeting 0.08, 0.20, 0.39, 0.59 and 0.98 in h^{-1} were simulated on 19 individual trees of each species for a total of 130 storms. Actual storm sizes varied somewhat due to wind conditions, so the measured storm sizes were used in the analyses. There was no difference

Table 4-3—Estimated event and annual interception as compiled from the literature and summarized by Pierson and Williams 2016.

Cover type	Event interception as % of gross rainfall	Annual interception as % of gross rainfal
Individual conifer or shrub	50–60 for low intensity	5–50
	5–35 for high intensity	5–15 more common
Litter	2–20	2–20
Shrub or woodland community	5–50	5–25
Herbaceous community	15–80	10–55

in interception based on species, so results were combined for both species. Stringham et al. (2018) found a threshold of 0.20 inches of precipitation; interception was higher for storms of 0.20 inches or less and lower for storms greater than 0.20 inches.

Niemeyer et al. (2016) measured interception of both rain and snow under western juniper instrumented with lysimeters under two tree canopies and two interspaces outside of the canopy were instrumented with lysimeters. The ratio of the interspace to under canopy surface water input was greater for snow (79.4 percent) than for rain (54.8 percent), which was attributed to the redistribution of snow under tree canopies. In other words, a greater fraction of snow made it below the tree canopy while more rainfall was intercepted. In the Great Basin, using rainfall collectors below western juniper tree canopies, 42 percent of annual precipitation was intercepted and subsequently evaporated or sublimated back to the atmosphere (Young et al. 1984). On the Edwards Plateau in Texas, Ashe juniper was monitored with collection tubes under the canopy for 3 years at 10 sites and 2,700 rain events were recorded. Average total interception by tree canopies was 35 percent (Owens et al. 2006). The litter layer under trees can also absorb water that would have otherwise made it to the soil surface. Owens et al. (2006) recorded that on average 5 percent of precipitation is absorbed by the litter layer under Ashe juniper.

Transpiration

Expansion of pinyon and juniper woodlands into areas previously dominated by sagebrush communities introduces deeply rooted evergreen species, which can change the timing and magnitude of transpiration losses relative to shrub and grass dominated communities (Ryel et al. 2010). Pinyon and juniper have different water use strategies as demonstrated by studies at the leaf level through measured photosynthesis, stomatal conductance (in part determines transpiration), and leaf water potentials (a measure of plant water stress) (Angell and Miller 1994; Lajtha and Barnes 1991; Limousin et al. 2013; Williams and Ehleringer 2000). Pinyon is relatively isohydric, which means leaf water potentials are regulated at a fairly conservative minimum stomatal setpoint determined by their vulnerability to xylem cavitation (Meinzer et al. 2009), below which stomata will close and photosynthesis will cease.

Juniper is relatively anisohydric (keeping stomates open for longer periods of time under the presence of decreasing leaf water potentials) and is able to tolerate lower leaf water potentials and keep stomata open, which maintains stomatal conductance and photosynthesis (West et al. 2008). Juniper will therefore continue to transpire water to the atmosphere during drought conditions for a longer time period than pinyon. Minimum leaf water potential demonstrates the divergence in water use strategies of these two species. Minimum leaf water potentials reported for twoneedle pinyon and oneseed juniper in the Southwest and Colorado Plateau range from -2.0 to -2.5 MPa for pinyon and -3.7 to -6.9 MPa for juniper (Lajtha and Barnes 1991; Williams and Ehleringer 2000; Limousin et al. 2013). The ability of oneseed juniper to withstand drought longer than twoneedle pinyon has also been attributed to the ability of juniper to extract water from deeper in the soil profile. Stable isotope ratios in xylem water of these two species show that while both species are able to use summer rain, pinyon is more reliant and responsive to summer rain—which increases shallow soil water—while juniper extracts a greater proportion of water from deeper in the soil profile (Bates et al. 2000; Breshears et al. 1997a; Flanagan et al. 1992; West et al. 2007a,b, 2008).

Williams and Ehleringer (2000) proposed that along the summer monsoon cline (with the largest input of summer rainfall in the Southwest and decreasing monsoon inputs moving northwards) that these species should use more monsoon rainfall in areas where monsoon rainfall is a greater percentage of total rainfall. This idea was largely supported by the authors' findings, but it was a threshold response to a critical amount of

summer precipitation, not a linear response (Williams and Ehleringer 2000). The isotopic evidence contained in 40,000-year-old twoneedle pinyon needles in pack rat middens indicate the distribution of twoneedle pinyon was strongly tied to summer rainfall (Pendall et al. 1999). It is notable that singleleaf pinyon replaces twoneedle pinyon in the Great Basin, which receives less than one-third of its total annual precipitation as summer rain. In primarily winter-spring precipitation regime in the northern Great Basin, transpiration is strongly influenced by soil temperature and moisture availability (Angell and Miller 1994). Peak transpiration occurs during the spring. However, transpiration rates are relatively low compared to other conifers, partially a result of stomata covered by adjacent leaf scales (fig. 2-16a,b; Miller and Shultz 1987).

In a study in southern Utah, twoneedle pinyon and Utah juniper relied on similar shallow water sources in the early spring, and may be in direct competition with understory species. However, during the summer twoneedle pinyon was able to increase transpiration in response to summer rainfall events that created shallow soil water, while Utah juniper does not. Both species are able to obtain a considerable proportion of water from below the shallow soil depth (West et al. 2008) likely due to their ability to grow deep roots into cracks in bedrock and grow on rock outcrops without significant soil development (fig. 4-11; Foxx and Tierney 1987; Harper et al. 2003). In general it appears that twoneedle pinyon is more reliant on summer rainfall than is Utah juniper (Flanagan et al. 1992; Williams and Ehleringer 2000). This requires that twoneedle pinyon maintain active shallow roots, which experience greater temperature extremes that may limit this species in the future if temperatures continue to increase. Evidence for the presence of shallow roots demonstrate twoneedle pinyon can take up water from intercanopy spaces (Breshears et al. 1997b), which could make it competitive with understory shrubs and grasses.

Transpiration can be measured at the whole tree level by installing sensors to measure sap flow rates through xylem tissue (Granier 1987), but scaling such tree level measures

Figure 4-11—Juniper and pinyon roots can go deep into the cracks of bedrock, grow on rock outcrops and exist with very shallow soils (less than 10 inches in depth). Grand Staircase, southern Utah. (Photo by Rick Miller, Oregon State University.)

to the stand level is problematic. Three studies scaled sap flow velocity to the whole stand level using allometric measurements and sap flow rates (Mollnau et al. 2014; Pangle et al. 2015; West et al. 2008). West et al. (2008) measured maximum sap flow rates at 0.01 in d^{-1} and annual total transpiration between 6.5 percent–14.5 percent of total annual rainfall for a site in southern Utah with 9 inches of average annual precipitation. Approximately 88 percent of basal tree cover at the site was oneseed juniper and the remaining 12 percent was twoneedle pinyon (West et al. 2008). Pangle et al. (2015) also found total pinyon and juniper transpiration was low and 11 percent of annual precipitation over 5 years for a stand with similar basal area of oneseed juniper and twoneedle pinyon and located in the warm Chihuahuan Desert with average precipitation of 14 inches. In wet years or irrigated treatments, trees used more water, but still less than 18 percent of total precipitation (Pangle et al. 2015).

In the studies by West et al. (2008) and Pangle et al. (2015), understory coverage was minimal, indicating much of the annual precipitation is lost through other processes than plant transpiration, such as interception, evaporation, and/or runoff. The third study, Mollnau et al. (2014), was located in Oregon in the cold-dominated Great Basin with mean annual precipitation of 13 inches. This study estimated that transpiration for a stand of western juniper was 0.02 in d^{-1} during the summer months (Mollnau et al. 2014). The stand-level estimates from the studies cited herein are on the low end of the range calculated from a canopy diffusion model, which predicts 15–80 percent transpiration loss from pinyon and juniper woodland (Lane and Barnes 1987). Based on seasonal conductance for a Phase III western juniper stand, a transpiration model estimated transpiration accounted for 44 percent of the total precipitation received in 1 year (12.5 inches) (Angell and Miller 1994). This highlights the complexities of measuring the water budget and scaling up individual measurements to the stand-level (Wilcox et al. 2003a). Sap flow can underestimate transpiration due to limited numbers of probes, variations in actual sapwood area, failure to adequately characterize radial variation in sap flow rates, thermal gradients, and power supply problems (Čermák et al. 2004; Clearwater et al. 1999).

Total Evapotranspiration

Total evapotranspiration can be measured at the ecosystem level by mounting instruments on tall towers above the tree canopy to determine the outgoing flux of water vapor; the current method is referred to as eddy covariance estimates. Eddy covariance data for Utah juniper measured the highest evapotranspiration rates in March and April of 0.06 in d^{-1}; the remainder of the growing season rates were below 0.04 in d^{-1} (Leffler et al. 2002). However, there is a lack of studies on evapotranspiration of pinyon and juniper woodlands at this ecosystem scale.

Effects of Disturbances and Land Management Practices

Numerous empirical assessments and modeling studies have assessed the effects of pinyon and juniper die-off or removal on evaporation, transpiration, interception, snow ablation (the sum of snow evaporation and sublimation), snow accumulation, and streamflow timing and magnitude (see review by Adams et al. 2012). In these dryland regions where precipitation is usually below 20 inches, conflicting results have been reported in the literature. Adams et al. (2012) proposed a conceptual model for assessing potential hydrologic consequences of large landscape-scale tree die-offs. The authors' conceptual framework is also relevant for a variety of tree-removal conditions. Tree removal can alter shading and solar radiation inputs; directly diminish transpiration and interception; reduce surface roughness, and thereby increase wind speeds; and facilitate night-time cooling by elevating outgoing longwave radiation at night through loss of the

warm air trapping effect. The net effect of these opposing processes determines whether more or less water is available, as depicted in the authors' conceptual model. The results of these processes are generally more ambiguous in regions where annual precipitation is less than 20 inches because almost all water is lost to evapotranspiration regardless of vegetation type. The complexities lie in separating these terms into their component parts. In wetter regions, conversion of woody vegetation to shrubs and grasses is known to produce a reduction in evapotranspiration because of the removal of more deeply rooted species (Zhang et al. 2001).

Williams et al. (2014a) suggested that tree removal through fire may serve as an ecohydrologic threshold reversal mechanism over time on juniper-dominated sagebrush rangelands. The authors suggested that, over time, tree removal by fire or cutting potentially increases soil water availability (Bates et al. 2000; Roundy et al. 2014a) and thereby enhances understory vegetation (fig. 4-12; Roundy et al. 2014b) that improves infiltration (Williams et al. 2018a), reduces runoff and erosion (Pierson et al. 2007; Williams et al. 2018a), and promotes persistence of a sagebrush and native herbaceous plant community (Bates et al. 2014; Chambers et al. 2014; Miller et al. 2013, 2014; Roundy et al. 2014b; Williams et al. 2017). Roundy et al. (2017) further suggested tree removal by chaining combined with seeding, served in the same manner, and found that the treatment increased intercanopy vegetation and reduced runoff and erosion from natural rainfall events on a pinyon and juniper woodland in Utah. Williams et al. (2018) suggested that studies of pinyon and juniper woodland ecohydrologic responses to tree removal across the Great Basin and Colorado Plateau depend on effectiveness of treatments to enhance intercanopy vegetation. Overall, the impacts of disturbances and

Figure 4-12—The response of this site to fire was largely due to a good cover of bluebunch wheatgrass prior to the event. Increased herbaceous cover improves infiltration and reduces runoff and erosion. (Photo courtesy of Northwest Watershed Research Center, USDA Agricultural Research Service, Boise, Idaho.)

land management actions on water and erosion in woodlands then depend on the degree to which those perturbations alter the vegetation structure and associated hydrologic processes (Kormos et al. 2017b; Ludwig et al. 2005; Niemeyer et al. 2016; Ochoa et al. 2018; Pierson et al. 2007; Roundy et al. 2017; Wilcox et al. 2003a; Williams et al. 2016a, 2019).

Drought

The literature is particularly limited regarding drought impacts on water availability in pinyon and juniper woodlands. Recent landscape- to regional-scale die-offs of pinyon and juniper in the southwestern United States have been attributed to periods of drought and associated limited soil water availability, plant water stress, bark beetle infestations, and reduced tree regeneration (see Section 2—Climate Controls on Pinyon and juniper Distribution).

There are few studies to indicate how drought and mortality of pinyon and juniper will affect hydrology and erosion. Allen and Breshears (1998) and Wilcox et al. (1996a) chronicle how prolonged drought at a site in the Jemez Mountains of northern New Mexico in the 1950s facilitated landscape-scale plant community transition from ponderosa pine to twoneedle pinyon-oneseed juniper vegetation and thereby altered runoff and erosion processes. The authors indicated that fire suppression in the years prior to the study, and dating back to the 1880s, allowed pinyon and juniper trees to establish beneath and adjacent to ponderosa pines (Allen and Breshears 1998). Drought and subsequent beetle infestations in the 1950s contributed to ponderosa pine mortality and allowed the more drought tolerant pinyon and juniper trees to dominate site resources. Herbaceous cover was not evaluated at the site in the 1950s, but the authors contend that herbaceous cover at the site was likely low then, declined with conversion to a pinyon and juniper woodland, and approached approximately 2 percent at the time of the Allen and Breshears (1998) study. Overgrazing and vegetation competition for limited soil water facilitated increases in bare ground at the site over the mid- to late-1900s, culminating in extensive and well-connected bare intercanopy area and amplified erosion (Wilcox et al. 1996a).

Wilcox et al. (1996a) estimated from a 2-year study that annual runoff at the site accounted for less than 10 percent of the annual water budget, but that there was little storage of runoff across the site at the watershed scale. The studies by Allen and Breshears (1998) and Wilcox et al. (1996a) do not necessarily indicate that runoff increased following the vegetation type conversion from ponderosa pine to pinyon and juniper woodland, but increased erosion at the site as reported by the authors suggests runoff may have been elevated in association with the plant community transition.

In general, increased runoff and erosion are likely during high water input events where cover is decreased by disturbance (Pierson et al. 2011; Williams et al. 2014a), but research regarding increased plot- to hillslope-scale runoff and erosion associated with drought are scant in the literature. Guardiola-Claramonte et al. (2011) evaluated streamflow for eight basins over a four-State regional area in the southwestern United States with recent (2000s) drought-related die-off of pinyon pine. The study found that streamflow for the study basins declined over 3- to 6-year periods after pinyon die-off and that only a small portion of the decline was attributable to climate variability (Guardiola-Claramonte et al. 2011). The snowline elevation in these watersheds was above the pinyon die-off, thus differences in snow processes were not a plausible explanation (Guardiola-Claramonte et al. 2011). Based on literature, the authors imply that the streamflow reductions were most likely due to increased soil water use associated with increases in understory vegetation following tree die-off and perhaps amplified evaporation from surface soils (Guardiola-Claramonte et al. 2011).

A similar study by Biederman et al. (2015) evaluated streamflow for eight catchments of the Colorado River that underwent substantial tree die-off associated with bark beetle infestations. That study found that streamflow, evaluated over nearly a decade after tree die-off, decreased in three study catchments and exhibited no change in the remaining five study catchments. The authors suggested the results reflect increased water use by residual vegetation and possibly increased water losses to snow sublimation and evaporation following die-off (Biederman et al. 2015). To date, there is little evidence that drought-related changes to vegetation in pinyon and juniper woodlands significantly affect water availability at the annual time scale, particularly in climate regimes where evapotranspiration demands commonly exceed precipitation.

Fire

Consumption of canopy and groundcover by fire reduces interception capacity and surface water retention and increases the quantity and intensity of water arrival at the soil surface and the flow volume and velocity across it (fig. 4-13; DeBano et al. 1998; Shakesby and Doerr 2006). The amount of additional water input made available by burning is dependent on the interception and storage capacity of residual cover and how quickly postfire ground cover returns. General estimates suggest that the quantity of interception by unburned rangeland trees, shrubs, and grasses approximates 0.04–0.08 inches (1–2 mm) of rainfall per storm (Bonan 2002), depending on the cover biomass, rainfall intensity and duration, cover moisture content, and the horizontal and vertical arrangements of cover elements. The conversion of interception loss and stemflow to rainfall arrival at the soil surface is nearly 100 percent where severe burning uniformly removes canopy and groundcover.

Figure 4-13—Consumption of canopy and groundcover by fire in the first year reduces interception capacity and water retention at the soil surface. It increases the quantity and intensity of water arrival at the soil surface and the flow volume and velocity across it. Recovery of this site in years 2 and 3 will depend on prefire herbaceous vegetation or seeding, postfire precipitation, and fire severity. (Photo courtesy of Northwest Watershed Research Center, USDA Agricultural Research Service, Boise, Idaho.)

Estimates are limited regarding fire-induced increases in soil water availability for woodlands. Roundy et al. (2014a) found that prescribed fire and mechanical tree-removal treatments in late succession woodlands of the Great Basin increased available water in the resource growth pool in the growing seasons by 26, 20, 15, and 19 days in the first through fourth year after burning, respectively. The overall additional time that water was available each year posttreatment decreased as plant cover increased.

Seyfried and Wilcox (2006) suggested that woody plant removal by burning can increase deep soil water, but only where soils are deep enough to store excess water below the rooting zone. Postfire reductions in raindrop intensity are as important as changes in the quantity of water (Shakesby and Doerr 2006). Greater raindrop impact after canopy and groundcover removal results in increased soil detachment from rainsplash processes (Pierson et al. 2008b, 2009, 2013, 2014; Williams et al. 2014a,b, 2016a). Reductions in groundcover abate surface retention of overland flow, allowing flow to concentrate and move downslope with greater velocity, erosive energy, and transport capacity (Al-Hamdan et al. 2012a, 2012b, 2013; Pierson et al. 2009, 2011; Williams et al. 2014a,b, 2016a). The potential overall effect is a decrease in the time to runoff generation and an increase in cumulative runoff and sediment yield over the duration of a storm event.

Overall, the degree to which fire affects infiltration and runoff and erosion processes depends on the magnitude of alterations to soil properties, amount, and recovery time of vegetation and litter cover as well as inherent site attributes such as soil type, slope angle, and topography (Pierson et al. 2011; Shakesby and Doerr 2006; Williams et al. 2014b). For snow-dominated environments, burning of vegetation may alter snow accumulation, timing of runoff initiation, cessation, peak flow within the year, and amount of snowmelt runoff. Burning may also result in increased surface temperatures and snowmelt rates due to greater incoming solar radiation postburn. Any reduction in vegetation, therefore, reduces snow accumulation and water availability for biological processes and streamflow generation. Reduced snow retention also potentially alters runoff characteristics from summer thunderstorms on water-limited sites by inhibiting vegetation production and groundcover recruitment. Where snow does accumulate, runoff responses to mid-winter rain-on-snow events may be substantial after burning (see Marks et al. 2001 and Pierson et al. 2001).

A number of rainfall simulation experiments have been conducted in burned and unburned areas of pinyon and juniper woodlands and document fire effects on plot-scale infiltration, runoff, and erosion for these communities (table 4-2; Pierson et al. 2013, 2014, 2015; Roundy et al. 1978; Williams et al. 2014a, 2018). Roundy et al. (1978; table 4-2) measured infiltration and erosion immediately after and 1 year after burning on a singleleaf pinyon-Utah juniper site. Rainfall was applied at 3.31 in h^{-1} for 1 hour across dry and wet antecedent moisture conditions. Infiltration was generally similar across burned (2.85–3.23 in h^{-1}) and unburned (3.08–3.28 in h^{-1}) shrub and tree canopy areas for the dry soil conditions, and was lower in burned shrub plots (2.25–2.68 in h^{-1}) than burned tree plots (2.58–3.07 in h^{-1}) for wet soil conditions. Interspaces generally exhibited the lowest infiltration rates and burned interspaces (0.89–1.52 in h^{-1}) had substantially lower infiltration relative to burned tree and shrub plots (2.63–2.66 in h^{-1}) for wet soil conditions.

In a multi-site study of burned pinyon and juniper woodlands in the Great Basin, Pierson et al. (2013, 2014) and Williams et al. (2014a) applied rainfall at 2.52 in h^{-1} for 45 minutes for the dry soil conditions and 4.02 in h^{-1} for 45 minutes for the wet soil conditions, 1 and 2 years after fire. In those studies (table 4-2), fire had varying impacts on infiltration and runoff generation, with the main impact being reduced infiltration and increased runoff from burned relative to unburned tree plots at two of three sites. Soils were water repellent on burned and unburned tree plots. Litter on unburned plots buffered

repellency effects on runoff generation. Burning had no significant impact on sediment yield from the singleleaf pinyon-Utah juniper site, but increased sediment yield by three- to sevenfold for tree and shrub plots at the Utah juniper site (Pierson et al. 2014). For the western juniper woodland, burning increased sediment yield for wet soil conditions in the first year following fire by 34-fold for tree plots, 24-fold for shrub plots, and 4-fold for interspace plots (Pierson et al. 2013; Williams et al. 2014a).

Overall, burning created more uniform conditions at the fine spatial scale, resulting in greater amounts of runoff and/or sediment for transport to coarser scales in the first year postfire (Pierson et at. 2013, 2014; Williams et al. 2014a). Pierson et al. (2015, table 4-2) applied the same rainfall rates to 140 ft^2 plots on the same sites as in the Pierson et al. (2014) study 1 year postfire. High runoff and sediment yield at the coarser scale persisted in intercanopy areas after burning at the singleleaf pinyon–Utah juniper site, but runoff and sediment yield were largely unaffected by burning in tree canopy areas at that site. In contrast, runoff and sediment yield increased dramatically (approximately 8-fold and approximately 30-fold, respectively) after burning tree plots and remained high in intercanopy areas at the Utah juniper site.

In the Pierson et al. (2013) and Williams et al. (2014a) studies, high levels of runoff and sediment yield persisted one year after burning in the intercanopy and burning increased runoff and sediment yield by approximately 6-fold and more than 20- fold, respectively. Rainfall simulation methodologies and plot size in that study were consistent with those in the Pierson et al. (2015) study. The studies by Pierson et al. (2013, 2015) and Williams et al. (2014a) all attribute increases in postfire runoff and erosion following burning to accumulation of runoff and sediment sources from fine scales into high velocity concentrated overland flow over coarser scales. Burning in the studies created more homogeneous bare ground conditions at all three sites, and the variation in hydrologic and erosion responses across sites postfire reflects differences in initial vegetation and surface conditions and soil type across the three sites.

Studies by Williams et al. (2018a,b) and Nouwakpo et al. (in review) repeated the experiments of Pierson et al. (2015) in the summer of 2015, 9 years after burning. Williams et al. (2018a,b; fig. 4-6) measured increases in herbaceous cover within the intercanopy 9 years postfire, resulting in improved infiltration and reduced runoff and sediment yield from interspaces by more than twofold. On larger scale plots, reductions were 3- to 7-fold for intercanopy runoff and 3– to more than 75–fold for intercanopy sediment yield for the highest intensity storm at both sites (Nouwakpo et al. in review). Those studies support the importance of postfire herbaceous groundcover recovery to enhance infiltration and limit runoff and sediment transport to the hillslope scale. Collectively, the rainfall simulations discussed here demonstrate that fire can impart an initial increase in runoff and sediment yield on woodlands sites depending on initial vegetation and soil attributes, and that, where burning enhances herbaceous cover, improved infiltration and reduced runoff and erosion at the plot to hillslope scales are likely over time (Nouwakpo et al. in review; Williams et al. 2018a). At the watershed scale, peak discharge rather than cumulative runoff tends to be greater after burning, and is most pronounced after short-duration, high intensity, convective thunderstorms over large expanses of severely burned landscapes (Shakesby and Doerr 2006). Estimates are limited regarding fire impacts on watershed scale runoff and sediment yield from woodlands. Studies from mountainous forested settings indicate hillslope erosion can approach 24–40 t ac^{-1} yr^{-1} the first few years following burning, and recovery to prefire erosion rates may take 4 to 7 years (Robichaud 2009; Robichaud et al. 2008). Debris flows are uncommon for woodlands following burning, but have been documented in the literature (Cannon et al. 1998, 2001).

Mechanical Tree Removal

In contrast to fire, mechanical treatments can retain much of the existing understory vegetation and therefore typically pose few negative impacts on hydrology and erosion (Miller et al. 2013, 2014; Roundy et al. 2014a, 2014b; Williams et al. 2019). Bates et al. (2000) found that tree cutting at a western juniper woodland increased soil water availability at 0–8 inch and 8–16-inch soil depths in each year of a 2-year study, and that the greater soil water availability in the cut versus an uncut areas resulted in greater total understory plant biomass. As noted above for fire, Roundy et al. (2014a) found tree removal increased the number of wet days up to 26 days when applied to sites with high tree cover and that soil water was available 8.6 days and 18 days longer in treated versus untreated areas the fourth year posttreatment where tree removal was applied at moderate to high tree cover. Also, the authors noted that the additional time that water was available each year posttreatment decreased as plant cover increased.

Increases in the number of wet days were similar for mechanical and prescribed fire treatments in that study of 13 woodland-encroached sagebrush rangelands in the Great Basin (Roundy et al. 2014a). Mollnau et al. (2014) assessed soil water depletion over a 2-year period for a western juniper site on plots approximately 65 x 65 feet and spanning a variety of cover conditions resulting from vegetation manipulations including juniper removal. The authors found that spring season soil water content in the top 24 inches of soil was 4.2 inches on juniper-dominated plots with a shrub and herbaceous understory as compared to 5.3 inches on shrub- and herbaceous-dominated plots where trees had been removed and 6.1 inches on plots primarily with bare ground. Spring season soil water content over the same depths was 4.5 inches for both juniper-dominated plots without a shrub and herbaceous understory in comparison with the shrub and herbaceous-dominated plots in which trees were removed. Over depths of 24 inches to 35 inches, spring season soil water content was lower for plots with juniper cover (2.6 inches) relative to plots without juniper cover (3.5 inches). Soil water content over 0–24 inches depth by the fall season was similar across juniper plots (2.6 inches) and shrub and herbaceous covered plots (3.2 inches) and was highest for the primarily bare plots (5.4 inches).

Mollnau et al. (2014) attributed the seasonal differences in soil water contents across cover types to differential use of soil water to meet plant needs on vegetated plots, potentially greater interception loss on juniper plots, and minor evaporative losses from bare plots. The authors further suggested evaporation accounted for soil water depletion only to about 6-inch soil depth. Based on regression analysis of seasonal soil water content at different depths, Mollnau et al. (2014) concluded that juniper, shrubs, and herbaceous plants all shared soil water resources in the upper 24 inches of the soil profile, that juniper was the primary user of soil water below this depth, and that juniper water use limited deep soil recharge and seasonal soil water carryover. However, lack of differences in fall season soil water across the juniper-dominated versus shrub and herbaceous-dominated plots suggests that available soil water is readily used by whatever vegetation occupies a respective site. This is similar to the Roundy et al. (2014a) study in which increases in the number of wet days declined with time after tree removal as the understory responded to available soil water. In the winter months in central Oregon western juniper depleted winter soil moisture at 20 inches, if subsoil temperatures were above freezing (Jeppesen 1978). Where soils remained frozen or woodlands had been thinned, soil moisture was significantly higher.

A substantial number of studies have evaluated the effects of mechanical treatments on hillslope runoff and erosion processes (table 4-2; Cline et al. 2010; Hastings et al. 2003; Jacobs 2015; Pierson et al. 2007, 2013, 2015; Roundy et al. 2017; Williams et al. 2019). At the fine spatial scale, Cline et al. (2010) and Pierson et al. (2014) found that

placing shredded tree mulch and debris on 5.4 ft^2 interspace plots enhanced infiltration and reduced runoff and erosion from high intensity rainfall (2.52 in h^{-1} and 102 in h^{-1}, 45-minute durations) in a Utah juniper woodland. Sediment yield during application of the highest intensity on mulch-covered interspace plots was more than fivefold less than from mulch-free interspaces (Pierson et al. 2014). Cline et al. (2010) reported that mulch residue reduced sediment yield from the same simulated storm by nearly twofold for grass interspaces and by eightfold for bare interspaces. The study further found that bare interspaces with tire tracks from the shredding equipment generated the highest amount of sediment during rainfall simulations, but these areas were reasonably surrounded by areas with masticated debris (Cline et al. 2010).

Pierson et al. (2013, 2015) found that cutting and placing downed pinyon and juniper trees in the intercanopy had no immediate beneficial impact on runoff and erosion rates at multiple woodland sites in the Great Basin. In those studies, runoff tended to route through downed trees where there were voids in contact of the tree debris with the ground surface (fig. 4-14). In a followup study of the sites, Williams et al. (2019) found that runoff and erosion from overland flow were greatly reduced by downed trees 9 years after cutting. The downed trees 9 years after cutting had settled into place and were in good contact with the soil surface. Downed trees and debris detained concentrated flow releases during overland flow experiments and allowed more time for water to infiltrate and for sediment deposition (Williams et al. 2019). This supports the common postfire rehab practice of placing downed logs (usually trees felled as part of suppression/mop-up and hazard tree-removal operations) across the slope—sometimes staking them in place to prevent rolling downhill—to decrease erosion. Pierson et al. (2007) found that

Figure 4-14—Placing downed pinyon and juniper trees on the intercanopy had no immediate beneficial impact on runoff and erosion rates at multiple woodland sites in the Great Basin. In those studies, runoff tended to route through downed trees. However, 9 years following treatment, runoff and erosion from overland flow were greatly reduced. (Photo courtesy of Northwest Watershed Research Center, USDA Agricultural Research Service, Boise, Idaho.)

intercanopy runoff and erosion from rainfall simulations (2.17 in h^{-1} intensity, 60-minute duration,) in a cut western juniper woodland were substantially less than in an adjacent uncut control woodland 10 years after cutting. The authors concluded that increased herbaceous cover 10 years following cutting improved infiltration and limited formation of concentrated overland flow within the intercanopy.

Roundy et al. (2017) compared runoff and sediment from natural rainfall events on intercanopy plots (approximately 110 ft^2) in chained-and-seeded and untreated areas of a pinyon and juniper woodland in Utah over a 5-year period. The site frequently received rainfall from high intensity summer monsoonal thunderstorms. Chaining and seeding increased vegetation and reduced bare ground by threefold relative to the untreated area. The authors estimated that chaining and seeding reduced runoff and sediment by 5– to 10–fold as averaged over the 5-year study.

Hastings et al. (2003) found that cutting pinyon and juniper and evenly distributing tree debris (lop-and-scatter) within the intercanopy reduced erosion from high intensity rain events on a degraded and rapidly eroding twoneedle pinyon-oneseed juniper woodland in New Mexico. Erosion from natural rainfall events over two rainy seasons was one to three orders of magnitude more for untreated than treated micro-watersheds (3,230–11,840 ft^2 area). Hastings et al. (2003) attributed the reduced erosion following tree cutting to enhanced infiltration and soil water retention afforded by slash, herbaceous cover recruitment, and reduced interconnectivity of runoff and sediment source areas. Jacobs (2015) assessed cover, runoff, and erosion responses to the treatments at the Hastings et al. (2003) study sites over a 16-year period posttreatment. During that time, the sites underwent a multi-year drought, wildfire, and beetle outbreak (Jacobs 2015). Treated areas more rapidly revegetated and improved in hydrologic function relative to untreated areas following the disturbances, and therefore the authors concluded that the treated areas exhibited greater resilience to perturbations than the untreated areas.

Collectively, these field studies demonstrate that mechanical tree-removal treatments can effectively improve infiltration and reduce hillslope runoff and sediment yield where the treatments enhance vegetation and groundcover and reduce bare ground.

Studies of the effectiveness of mechanical tree-removal treatments to increase watershed streamflow have produced mixed results. In a paired watershed study, Deboodt (2008) determined that tree cutting in a snow-dominated western juniper woodland reduced overall water use for transpiration during the cool season, and thereby allowed soil water to increase over the year, yielding higher end-of-year deep soil water content in cut versus uncut juniper woodlands. He also found that tree cutting had a favorable effect on groundwater levels and springflow, but treatment effects on streamflow were difficult to discern from variability in precipitation input. Ochoa et al. (2018) studied the cut and uncut watersheds from the Deboodt (2008) study 13-years postcutting. The authors found that the timing and amount of precipitation strongly influenced soil water recharge and that increased soil water recharge during snowmelt led to a rapid water table rise and streamflow. That study detected 1.5- and 1.7-fold greater peak streamflow and springflow rates in the cut versus uncut watershed and annual streamflow was 3.6 times greater for the cut watershed over the 4-year study period.

However, Ochoa et al. (2018) noted that annual streamflow and spring flow prior to the treatments were on average 1.8 and 3 times greater for the watershed subsequently cut in comparison with the control watershed. The authors did not explicitly state that interception loss affected soil water recharge but reported that up to 46 percent of annual precipitation was intercepted. Niemeyer et al. (2016) and Kormos et al. (2017a) suggested through modeling that juniper dominance on mid elevation snow-dominated sagebrush rangelands can alter the distribution and amount of snow accumulation across a watershed, and thereby affect the spatial distribution of and the timing and delivery of water availability for soil

water recharge and streamflow (e.g. Flerchinger and Cooley 2000; Seyfried et al. 2009; Williams et al. 2009). The Kormos et al. (2017a) study applied 6 years of measured and modeled data to determine that woodland-dominance of 4 small sagebrush watersheds in southwest Idaho limited accumulation of snow in deep drifts (fig. 4-15). The authors found that the more evenly distributed snow cover under the juniper-dominated condition resulted in earlier spring snowmelt and summer streamflow cessation relative to conditions dominated by sagebrush cover. The Deboodt (2008), Ochoa et al. (2018), and Kormos et al. (2017a) studies were on sites in which precipitation occurs primarily as snow and streamflow occurs mostly as cool season runoff.

Wilcox (1994) summarized results from a number of mechanical tree-removal studies aimed at increasing streamflow from southwestern U.S. woodlands and found that results varied substantially with runoff regime, summer thunderstorm driven versus dominated by cool season frontal rains, snowmelt, or rain-on-snow water input. The Cibeque Ridge paired watershed study (table 4-1) in Arizona found that chaining pinyon and juniper combined with slash burning and seeding increased streamflow the first 2 years posttreatment on a 100-acre watershed relative to an adjacent control watershed of the same size, but streamflow in the subsequent year declined to that below the untreated watershed, presumably due to increased transpiration losses associated with seeded grasses (see Ffolliott and Gottfried 2012). At two watersheds in the Beaver Creek Experimental watershed in Arizona (table 4-1), 100 percent removal of pinyon and juniper trees had no effect on water yield. However, a third watershed targeted only 83 percent of juniper with herbicide—and this increased annual streamflow 65 percent for the first 4 years and 157 percent after 8 years (Clary et al. 1974). The increase in streamflow translated on area basis to 0.49 inches y^{-1}. However, after 8 years, dead trees were removed and post treatment streamflow returned to pretreatment levels (Zou et al. 2010). This suggests that the actual disturbance from the tree-removal treatments reduced infiltration in the other two watersheds and/or removal of canopy cover increased evaporation. And, it seems plausible that the remaining pinyon trees and dead standing juniper trees continued to modify the near-surface energy balance and reduce evaporation in the third watershed. Results from the study are also consistent with the idea that juniper extracts deeper sources of water that recharge streamflow (Deboodt 2008; Mollnau et al. 2014).

As indicated above in the discussion of drought, the limited research on streamflow patterns following large-scale tree die-off on woodlands and pine forests in the
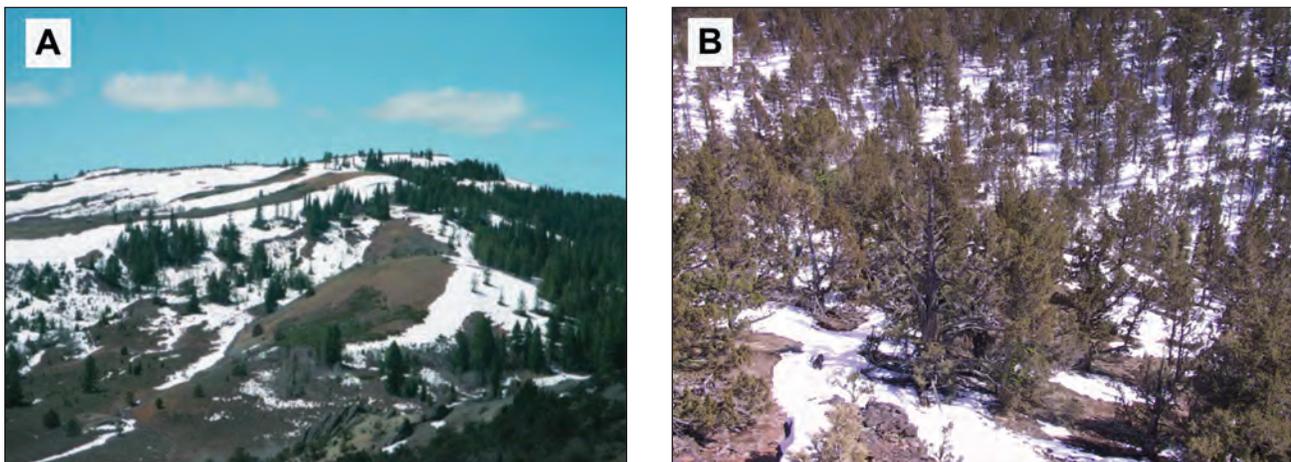
Figure 4-15—Photographs showing snow accumulation in (A) drifts in sagebrush dominated areas of the USDA Agricultural Research Service, Reynolds Creek Experimental Watershed, Idaho and (B) more uniform snow distribution at a western juniper dominated sagebrush site in the South Mountain study area (Kormos et al. 2017; photographs provided by the USDA Agricultural Research Service, Northwest Watershed Research Center, Boise, Idaho).

USDA Forest Service RMRS-GTR-403. 2019.

southwestern United States indicate that runoff is unchanged by tree reductions in mixed-phase precipitation climates (Biederman et al. 2015; Guardiola-Claramonte et al. 2011). Overall, the literature suggests increases in soil water and streamflow associated with mechanical tree removal vary with climate regime in addition to site-specific attributes of the area treated. More long-term studies are needed to definitively state whether mechanical tree removal, as well as tree removal in general, is effective at increasing streamflow across various ecoregions containing pinyon and juniper woodlands. For more detailed discussion on tree-reduction practices on vegetation and water, see Section 5, Restoration and Management.

SECTION 5: RESTORATION AND MANAGEMENT

Some assume that managing rangelands for presettlement conditions can successfully maintain sustainability and biodiversity into the future. But this paradigm is challenged by several factors—a continually changing climate, new species introductions, and changing disturbance regimes.

Summary

Vegetation management of pinyon and juniper woodlands began primarily after World War II, when fuel costs were low, and a large surplus of heavy mechanical equipment became available. The primary goals were to increase livestock forage production, improve watersheds to prevent downstream flooding of towns (which occurred along the Wasatch Range in Utah), and improve declining big-game winter habitat (Aro 1975; Dwyer 1975; Terrel and Spillett 1975). In the 1950s and 1960s, the primary method for tree removal was chaining, which often included broadcast seeding between chainings to introduce perennial grass species (Aro 1975). At the time there was little distinction made between pre- and postsettlement woodlands or the resilience and resistance to invasive species of the areas to be treated, which led to mixed results.

Two-way chaining with broadcasting seeding between chainings provided the best results for increasing perennial grass cover. By the 1970s, the acres of rangelands chained annually declined significantly, but chaining is still being used at smaller scales. When applied properly, chaining and other treatments can provide good results. Since 2000, shredding (mastication) and cutting have become the most common mechanical methods of tree removal. Both treatments often ultimately result in increases in available soil water, length of growing season, and shrub, perennial grass, and forb cover (Bates et al. 2000; 2017b; Miller et al. 2014b; Roundy et al. 2014a; Young et al. 2013a). Variation in cutting treatments include cut-and-leave; cut-limb-and-scatter; cut-and-broadcast burn; and cut-pile-and-burn—all with advantages and disadvantages. Tree removal in Phase III woodlands by mastication or cutting can increase the growing season by 2 or more weeks. Response of invasive annual grasses is always a concern and is closely linked to pretreatment tree and perennial herbaceous cover, and soil moisture and temperature regimes. Locations with warmer fall temperatures are especially susceptible to cheatgrass dominance, while those with cooler fall seasons and relatively wet winters and springs are more resistant.

Seeding is an important consideration where native perennials are depleted and soil moisture temperature regimes are favorable for invasive annuals (Bybee et al. 2016). Successful seeding can significantly reduce the abundance of annual grasses following mechanical treatment. A major disadvantage of mechanical treatments compared to

prescribed fire is the survival of small trees (less than 6 feet tall) and resprouting from remaining basal limbs, which can significantly reduce the longevity of the treatment, often requiring followup treatments. Abundance of slash must also be dealt with in late succession phases.

The broad-scale application of herbicides to mature woodlands has been limited due to mixed results and concerns related to impacts on native perennials and water. But the use of Picloram ™ has proven to be an effective tool following mechanical treatment when applied selectively on small trees or resprouts at the base of the trees. It has also been used to treat invasive annuals following tree removal, a consideration on sites with low resistance to invasive plants.

Although early studies evaluating the effects of prescribed fire were conducted in the early 1950s, the use of prescribed fire was very limited until the 1970s (Blackburn and Bruner 1975). Federal and State agencies were hesitant to use prescribed fire for fear of escaped fires, and limited surface fuels in many pinyon and juniper woodlands required extreme weather conditions to carry fire. In the early 1970s, a survey of burned woodlands reported increases in postfire perennial grasses. Between 2002 and 2016, 5.5 million acres were prescribed burned in the West. Prescribed fire has proven to be a useful tool under the right conditions. However, under the wrong conditions (closely linked to characteristics of the site), it can significantly increase invasive annuals.

When treating woodlands, the key components to success are an adequate level of residual perennial herbs present on the site and ecological site characteristics including soil texture, depth, and moisture and temperature regimes (Miller et al. 2013). The relative dominance of trees to perennial understory vegetation at the time of treatment is closely linked to plant succession following treatment. The vast majority of studies comparing mechanical versus prescribed burning reported larger increases (at least in the first few years following treatment) of invasive annuals in the burn treatments, especially on warm and dry compared to cool and moist sites. The advantages of fire are the removal of small trees, little to no resprouting, and costs that are typically less than most mechanical treatments. However, disadvantages are the removal of important nonsprouting shrub species (although these are often lacking in Phase III) and a greater threat of invasive species.

The lack of surface fuels in late Phase II and III areas also usually requires a pretreatment such as cutting a portion of the trees to conduct a prescribed burn under moderate weather conditions. Recovery of the perennial herb layer to preburn or greater levels typically takes 2–3 years versus in mechanical treatments—for which increases can occur in the first posttreatment year. However, burning Phase III woodlands under more severe conditions will significantly increase perennial grass mortality (more than 80 percent). Prescribed fire also greatly reduces residual cover of sagebrush and bitterbrush if they are still present in the understory. Recovery of sagebrush canopies to 20–30 percent cover on cool and moist sites usually takes 20–35 years. On warm and dry sites, sagebrush recovery takes considerably longer. Over the past 70 years our goals for restoration have broadened to maintaining or restoring ecosystem function. We have also learned that short- and long-term vegetation responses following treatment are closely related to the woodland successional phase, residual understory vegetation at the time of treatment, and soil moisture and temperature regimes in addition to management. However, invasive plants have proven to be an ever-increasing challenge, especially on warmer and drier sites with warm springs and falls. Sites with drier or cooler falls, and wet winters and springs favor perennial herb cover.

Introduction

Evolution of Rangeland Management

Initial observations and concerns about woodland expansion were noted in the early and mid 1900s (Cottam 1929; Cottam and Stewart 1940; Leopold 1924; Nicol 1937; Woodbury 1947). Until World War II, tree removal was done with handsaws and axes—primarily to obtain material for fenceposts, for fuel, and (in Nevada, Utah, and eastern California) for conversion of pinyon to charcoal for mining during the Comstock period. The use of heavy machinery, power saws, prescribed fire, and chemicals for tree removal did not begin in the West until after World War II (fig. 5-1). A surplus of heavy equipment and low fuel costs following the war led to the mechanization of management for semiarid landscapes. Aircraft were used for fire suppression and to spray herbicides across large swaths of land. Crawler tractors were used to plow roots, push trees over, and to pull plows, pipe harrows, anchor chains, roller cutters, and seed drills. Between the late 1940s and the 1970s the most common tree-removal methods on BLM lands in Utah, Colorado, Arizona, and Nevada were chaining, bulldozing, and cabling (Aro 1975; Redmond et al. 2014). Beginning in the 1970s, hand-cutting with chainsaws became the primary method used in central Oregon (Winegar and Elmore 1978) and later in southern Idaho and eastern California. Shredding became the mechanical method of choice in Utah starting around 2003. Beginning in the late 1970s and 1980s, prescribed burning became more common for treating rangelands, alongside mechanical treatment.

The primary goals of tree removal in the '50s, '60s, and early '70s were to increase forage production, improve watershed conditions, and enhance deer winter range (Johnson 1967; Terrel and Spillet 1975). During this period, treatment rarely distinguished between persistent versus newly expanded woodlands. By the early 1970s, concern about possible woodland mismanagement (Dwyer 1975) increased when observations of understory responses to tree removal were not always positive (Arnold et al. 1964; Blackburn and

Figure 5-1—Mechanical treatments including shredding (2006, foreground), thinning, and chaining have created a more diverse landscape mosaic, increasing both herbaceous vegetation and shrubs in 2018. We have learned much from our past successes and failures. Eastern Nevada. (Photo by Rick Miller, Oregon State University.)

Bruner 1975; Clary 1971, 1974; O'Rourke and Ogden 1969). Since then, objectives for tree removal on public lands have increasingly focused on watershed improvement, fuel reduction, wildlife habitat, and restoration of sagebrush ecosystem function. Funding for tree removal since 2000 has come largely from fuel reduction and sage-grouse habitat restoration budgets. The threat of sage grouse listing as a threatened or endangered species in 2002 resulted in an influx of financial resources directed toward improving and protecting sage grouse habitat (Miller et al. 2017). But it is increasingly recognized that in order for management actions to have long-term benefits in these semiarid ecosystems, goals and objectives need to focus on long-term maintenance and restoration of ecosystem structure and function, rather than on one particular factor in that system (Benson 2012; Boyd et al. 2014; Miller et al. 2017).

Management Today

One of the most frequent questions that managers grapple with is how resistant a particular site is to invasive annual plants—and how various treatment methods might influence the magnitude of their increase or eventual decline. The complex answer is—it depends. Long-term effectiveness of vegetation management on ecosystem function depends on a complex set of interacting factors that drive plant succession across spatially and temporarily dynamic Great Basin and Colorado Plateau ecosystems. The benefits and drawbacks of vegetation management on ecosystem function are closely linked to the type and severity of treatment as well as ecological site characteristics, pretreatment vegetation composition, and posttreatment management. Vegetation management to limit conifer expansion into shrub-steppe and shrubland communities must be developed with clearly defined objectives and all of the above factors in mind.

The ability to predict plant succession following various vegetation treatments is one of the most important skills a land manager can employ. Several key questions help to identify the best treatment strategy for a given ecological site and to predict potential outcomes (fig. 5-2; Miller et al. 2014a, 2015). These questions address the potential for resilience and resistance to invasive plants for the area to be treated. They include:

1) What are the ecological site characteristics of the area to be treated that will influence resistance to invasive annuals?

2) What is the current vegetation on the site?

3) How will different treatment methods influence posttreatment succession and invasive annuals?

4) What are the nonsprouting shrubs on the site?

5) What is the erosion potential if plant cover is temporarily reduced?

6) Will the density of small trees (less than 4 feet tall) require followup treatments?

7) Are surface fuels (less than 3.3 feet tall) adequate to carry fire under moderate weather conditions? Or is pretreatment required?

8) Will posttreatment fuel loads be acceptable?

From chaining to prescribed fire, managers over the years have adopted various methods to manage pinyon/juniper landscapes in attempt to reach specific ecological goals. The discussion below addresses the history, strengths, and drawbacks of various treatments and what we know about how these complex semiarid ecological systems respond to the impacts of treatment.
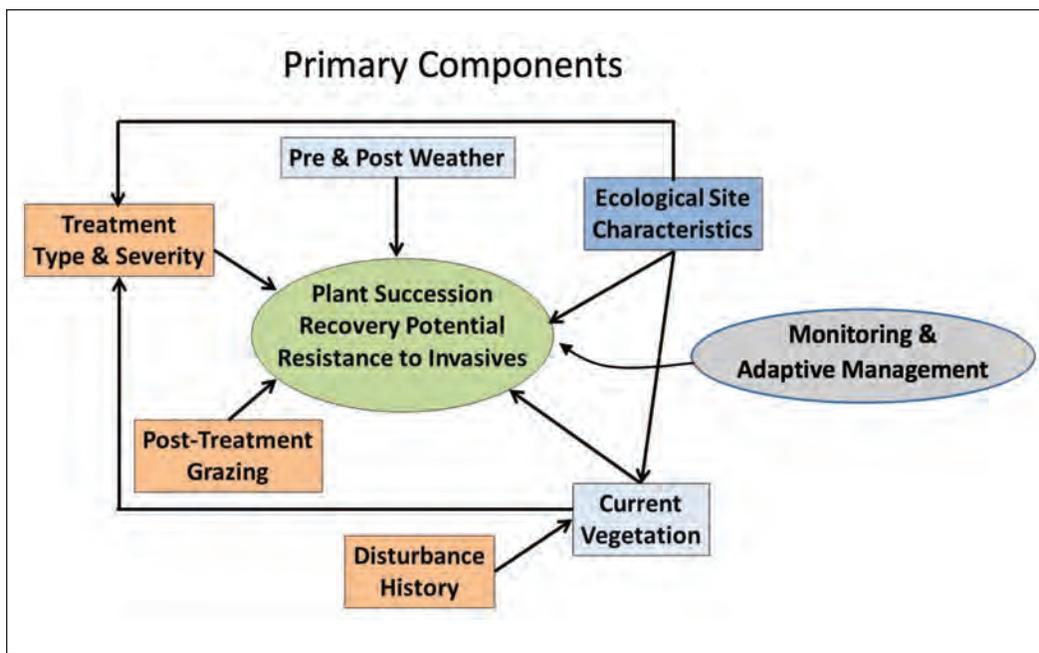
Figure 5-2—A conceptual model of the primary components that drive successional trajectories following vegetation management treatment. Consideration of key characteristics of these primary components substantially increases the ability to predict outcomes following disturbance and vegetation management (Miller et al. 2013, 2014a).

Chaining

In the Intermountain West, pulling anchor chains across the landscape was the most widespread method for pinyon and juniper removal from the late 1940s through the 1960s (fig. 5-3; Arnold et al. 1964; Aro 1975; Cotner 1963). Use of the method peaked in 1967, when a total of 80,000 acres were chained, and then declined to less than 5,000 acres in 1972. By the early 1970s, an estimated 514,000 acres of BLM land had been chained, with the largest proportion occurring in Utah (table 5-1; Aro 1975). Chaining fell out of favor in the 1970s as a result of high fuel prices, the indiscriminate selection of sites treated, high levels of soil disturbance, and the rise of the environmental movement—resulting in the increase of public concern. However, with the proper application of modern methods and careful site selection, chaining can be a useful tool for rehabilitation (fig. 5-4).

Chaining involves dragging a ship anchor chain, 200–500 feet long, between two bulldozers, pulling out large woody plants. Early work in Utah found 60– to 90–pound links were most effective, and that the addition of several swivels in the chain decreased the accumulation of debris caught up in the chain, reducing surface disturbance (Vallentine 1971; Steve Monsen, Retired Revegetation Specialist, USDA Forest Service, Rocky Mountain Research Station, Provo, Utah; and Richard Stevens, Biologist, Utah Division of Wildlife Resources, Provo, Utah, personal communications, 2017). Various modifications could also be made to the chain including welding 18-inch pieces of railroad rails or bars to the individual links. Bars were welded perpendicular to the chain links on the Ely Chain and parallel to the links on the Sagar Chain. Chaining was done (frequently, but not always) parallel to the slope contour, which reduced power requirements and diminished potential erosion by leaving both soil furrows and brush windrows parallel to the contour. A length-to-swath ratio for the chain of 2:1 to 3:1 was recommended. It was pulled in a J-shape for increased effectiveness of uprooting trees.

Figure 5-3—In the Intermountain West, pulling anchor chains across the landscape was the most widespread method for pinyon and juniper removal from the late 1940s through the 1960s. Under the right conditions it can be a useful tool to retain a portion of the shrubs and increase seed establishment by seeding between chainings. This image shows the first growing season after chaining. Schell Creek Range, eastern Nevada. (Photo by Rick Miller, Oregon State University.)

Table 5-1—Acres of pinyon and juniper woodlands chained by the BLM between 1960 and 1972 (Aro 1975).

State	Acres (x 1,000)	Percent
Utah	257	50
Colorado	93	18
Arizona	61	12
New Mexico	46	9
Nevada	43	8
Oregon, Idaho, and California	14	3
Total	**514**	

Increasing the distance between dozers in the second pass (resulting in a gentler curve in the chain) could increase the survival of sagebrush in the understory (Cody Coombs, Hazardous Fuels Program Manager, BLM, Ely, Nevada, personal communication, 2018).

In the late 1960s and early 1970s, the cost of chaining ranged from $15–$18 per acre (without posttreatment seeding) and jumped to as much as $33 per acre with followup hand-cutting of small trees (Winegar and Elmore 1978). In 2016, chaining cost estimates were around $90 to $150 per acre (Brad Washa, State Fuels Specialist, BLM, Utah; and Cody Coombs Hazardous Fuels Program Manager, BLM, Ely, Nevada, personal communications, 2017, 2018). Chaining is generally less expensive than cutting or shredding, and is still used in vegetation management.

Approximately 11,000 acres were chained between 2013 and 2017 in eastern Nevada. Two tractors, 100–150 feet apart, pulling a 200-foot chain, could treat 5–15 acres per

Figure 5-4—Broadcasting seed between chainings in 2013 resulted in more than 30 percent cover of perennial grasses and increasing sagebrush cover on this productive mountain big sagebrush-bluebunch wheatgrass site on moderately deep Mollisol soils. Photo taken in 2018. Schell Creek Range, eastern Nevada. (Photo by Rick Miller, Oregon State University.)

hour and cause 28–95 percent tree mortality—depending on tree size and density as well as terrain (Arnold et al. 1964; Aro 1975; Plummer et al. 1968; Vallentine 1971). Tree reductions in both cover and density were usually greater for pinyon than juniper (Tausch and Tueller 1977). Chaining resulted in greater tree mortality in even-aged stands where trees were 15–25 feet tall (Cotner 1963) than when trees were less than 6 feet tall. Across Nevada, chained sites where tree cover was reduced by 84 percent (Tausch and Tueller 1977) were again dominated by trees within 40 years of chaining (Bristow et al. 2014). In some cases, tree densities increased following chaining, resulting from the release of small trees (Aro 1975) and establishment of new trees (Winegar and Elmore 1978). In Utah, 75 percent of the trees present 25 years after chaining had established prior to treatment (Van Pelt et al. 1990). In central Nevada 40–50 years after chaining, tree cover was 23.5 percent—compared to 0.3 percent in an adjacent burned area (Bristow et al. 2014).

Two-way chaining (repeating the treatment in the opposite direction from the first chaining) significantly increases tree mortality compared to single chaining, but costs more. Two-way chaining doubled the cost but reduced Utah juniper cover from 35.5 percent to 4.1 percent (Skousen et al. 1986). This method is desirable when seeding is necessary because it allows seeds to be broadcast between chainings, which covers a portion of the seed and improves seedling establishment compared to just broadcast seeding (Aro 1971, 1975; Ott et al. 2003). Near Ely, Nevada, broadcasting a seed mix of natives and introduced species between chainings resulted in successful establishment (fig. 5-4; Cody Coombs, Hazardous Fuels Program Manager, BLM, Ely, Nevada, personal communication, 2018). Chaining burned woodlands may also help bury seed for more successful revegetation by breaking up soil water repellent layers (Madsen et al. 2012a). To increase the longevity of chaining treatments, followup vegetation management of small trees and basal sprouting is typically required.

Vegetation Response

In northern Arizona, chaining increased herbaceous vegetation production from 200 to 700 pounds per acre and increased the abundance of winterfat, cliffrose, and bitterbrush (Aro 1971), which suggests that in this case, native vegetation dominated the understory prior to treatment. Native shrubs also increased across five sites in Utah following chaining (Skousen et al. 1986, 1989). Chaining can result in a high percentage of old shrubs being killed, but most young shrubs survive (Skousen et al. 1986, 1989; Tausch and Tueller 1977). In Utah, however, on sites where shrubs and native perennial grasses were severely depleted, chaining plus seeding in the 1950s and 1960s often resulted in a conifer dominated overstory with an understory of introduced perennial grasses (fig. 5-5a,b; Steve Monsen, Retired Revegetation Specialist, USDA Forest Service, Rocky Mountain Research Station, Provo, Utah, personal communication, 2018). Shrubs did not recover 50–60 years following treatment. When native perennial grass cover is severely depleted (less than or equal to 5 percent cover), cheatgrass will often dominate following chaining in the absence of seeding (Davis and Harper 1990; Tausch and Tueller 1977; Ott et al. 2003). In central Nevada, where perennial grass cover was low prior to treatment, cover remained low 50 years following chaining (Bristow et al. 2014).

Chaining and Seeding

The vast majority of studies show significant increases in perennial grasses following chaining of depleted understories that were seeded to introduced perennial grasses. Seed was typically broadcast before or between chainings (Juran et al. 2008; MacDonald 1999; Ott et al. 2003). Establishment of seedling plants following a broadcast seeding (without covering the seeds) frequently resulted in limited or no success on warm and dry (mesic/aridic) ecological sites. Increased success of greater perennial grass cover (largely crested wheatgrass and intermediate wheatgrass) on the chained areas resulted in lower cheatgrass cover (Ott et al. 2003). But there are limitations to this approach—introduced species can decrease establishment of desirable native perennial species resulting in an alternative seeded state (Davies et al. 2013; Knutson et al. 2014; Lesica and Deluca 1996). To maintain biodiversity and a predominance of native plants, several things are needed—increased development of native plant materials (Ott et al. 2003), improved methods of planting natives, and increased availability of local native seed for planting (Plant Conservation Alliance 2015; Steve Monsen, Retired Revegetation Specialist, USDA Forest Service, Rocky Mountain Research Station, Provo, Utah, personal communication, 2017).
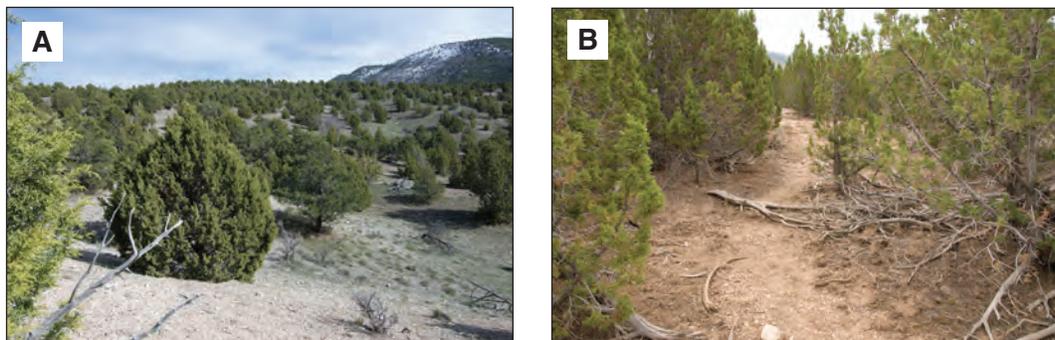
Figure 5-5—(A) This late Phase II woodland was chained and seeded with introduced grasses in the early 1960s. Few shrubs, native grasses, or forbs are present. A thinning stand of introduced grasses still persists as the tree canopy continues to increase. Many of these trees were likely small saplings at the time of the chaining. Understory was also likely severely depleted. (B) Early 1960s chaining with no seeding. Currently a closed Phase III woodland with Sandberg bluegrass in the understory and no shrubs or deep-rooted perennial grasses. West slope of the Wasatch Range near Ephraim, Utah. (Photos by Rick Miller, Oregon State University.)

Soil Disturbances Related to Shredding

The impact of chaining on water runoff and sediment yields depends largely on the slope angle and changes in the abundance and density of ground surface cover, surface roughness, compaction, soil characteristics, and soil disturbance (Gifford 1973a, 1973b, 1975; Gifford and Tew 1969; Gifford et al. 1970; Myrick 1971). Maintaining the rotation of the chain to limit the accumulation of debris greatly minimizes soil surface disturbance (fig. 5-6). Herbaceous vegetation cover on a site with high erosion potential following chaining and seeding in Utah was 23.5 percent on treated and 4.5 percent on untreated plots (Roundy et al. 2017). Runoff and sediment yields were 4.5 and 10 times lower, respectively, on treated than untreated sites.

Windrowing

Windrowing was a method used in the '60s and '70s that involved chaining followed by dozers piling or "windrowing" downed trees (Aro 1975). The method has proven to be highly successful, especially where seeds were drilled following windrowing. It is, however, expensive, as it requires one to two passes with the chain, followed by a third pass with heavy equipment to pile the debris (in addition to seeding), and it can potentially result in considerable soil disturbance. Thus, its use has been limited (Steve Monsen, Retired Revegetation Specialist, USDA Forest Service, Rocky Mountain Research Station, Provo, Utah, personal communication, 2016).

Dozing

In the past, bulldozing was used to reduce tree densities and open woodland canopies by pushing over and uprooting trees (Vallentine 1971). The method was best adapted for

Figure 5-6—First growing season following a two-way chaining and seeding with a mix of native species. Maintaining the rotation of the chain to limit the accumulation of debris greatly minimizes soil surface disturbance and retains a higher percentage of shrubs. Center of the photo shows an area heavily disturbed by debris accumulation on the chain. Chaining typically requires close supervision by the land manager in charge. Schell Creek Range, eastern Nevada. (Photo by Rick Miller, Oregon State University.)

removing scattered stands of low-density trees. It had less impact on shrubs than chaining (Skousen et al. 1989). But dozing was ineffective for removing small trees and left large pits and upturned soil. During the 1940s, Hula dozing was one of the most widespread methods for tree removal in the Southwest (Steve Monsen, Retired Revegetation Specialist, USDA Forest Service, Rocky Mountain Research Station, Provo, Utah, personal communication, 2016). The Hula dozer is a power-controlled tilt dozer blade with four teeth, which allowed operators to control blade angle and tilt (Vallentine 1971).

Shredding

Since 2003, shredding—also referred to as mastication, chipping, or mulching—has become a common method for tree removal in Utah (fig. 5-7). It is a process in which trees are shredded into small pieces using machinery carrying a rotating cylinder of cutting teeth (e.g., Bull Hog). The amount of mulch deposited from the treatment depends on the size and density of trees under treatment, but typically ranges from 0.4–7.9 inches (10–200 mm) deep on the ground, distributed within a few feet of the masticated tree. Mulch induces the increase in nitrogen mineralization at 5.9–6.7 inches (150–170 mm), which can result in increases in both cheatgrass and bluebunch wheatgrass (Aanderud et al. 2017). Over time, the positive effects increased for bluebunch wheatgrass and declined for cheatgrass.

Between 2012 and 2015 in Utah, over 61,569 acres of pinyon and juniper woodland were shredded, at an approximate cost of $280–$300 per acre (Brad Jessop, Natural Resource Specialist, BLM, Salt Lake City, Utah; and Brad Washa, State Fuels Specialist, BLM, Salt Lake City, Utah, personal communication, 2017).

Figure 5-7—First growing season following shredding of a Phase II woodland, with an understory of native grasses and shrubs. The resilience and resistance to invasives of this site is high. (Photo courtesy of Northwest Watershed Research Center, USDA Agricultural Research Service, Boise, Idaho.)

Response to Shredding

Reducing tree competition by shredding results in significant increases in soil water availability and surface litter biomass. Masticated surface litter can affect vegetation recovery, surface fuel loads, and soil moisture and temperature. Thick debris from shredding can suppress plant establishment (fig. 5-8; Kane et al. 2010; Young et al. 2013a). Masticated debris layers concentrated around the tree and existing tree litter limit seed-soil contact and seedling access to light, reducing germination and establishment. However, debris from shredding can enhance seedling establishment by lengthening the period of soil water availability, which can increase the growth and biomass of both invasive plants and perennial grass seedlings (Young et al. 2013a,b). Little work has been done to evaluate the life or rate of tree recruitment following shredding treatments. However, longevity of the treatment relates directly to the density of small trees remaining on the site, potential seed sources, and resprouting from basal limbs remaining after shredding (fig. 5-9).

Response to Shredding: Perennial Grasses and Forbs

Across multiple sites in Utah with a range of pretreatment tree cover, shredding that reduced tree cover by at least 20 percent resulted in a twofold to threefold increase in tall perennial grass cover (fig. 5-10; Bybee et al. 2016). On Shay Mesa in southeastern Utah, masticated sites had 24 percent perennial grass and 17 percent forb cover while perennial grass and forb cover were less than 1 percent on untreated sites (Ross et al. 2012). In several studies, perennial grass cover on masticated plots was also greater compared to sites treated with lop-and-scatter and cut-pile-and-burn (Bybee et al. 2016; Ross et al. 2012; Shakespear 2014;). In these studies, shrub cover was relatively unchanged by shredding in the first few posttreatment years.

Figure 5-8—Thick mulch layer following a recent shredding of a late Phase II woodland. Schell Creek Range, eastern Nevada. (Photo by Rick Miller, Oregon State University.)

Figure 5-9—Shredding and cutting can result in resprouting at the base of both Utah and western junipers, usually in the axis of the branch and trunk. A followup treatment is often required to increase the life of the treatment. (Photo by Rick Miller, Oregon State University.)

Figure 5-10—Nine years following a shredding and no seeding of a late Phase II and III woodland. The treatment resulted in a significant increase in native deep-rooted perennial grasses, primarily bluebunch wheatgrass, along the east slopes of the Egan Range in eastern Nevada. Note the limited amount of mulch remaining on the site. High elevation, cold winters, and precipitation of more than 12 inches make this black sagebrush and mountain big sagebrush bluebunch wheatgrass site resilient and resistant to invasive grasses. Egan Range, Nevada. (Photo by Rick Miller, Oregon State University.)

Shredding and Invasive Annuals

The way invasive annual plants respond following shredding is closely linked to pretreatment tree and perennial herbaceous cover. On tree-dominated, masticated sites (Phase III), where few pretreatment shrubs and perennial grasses remain, resistance to invasive weeds was very low (Bybee et al. 2016). In southeastern Utah, increases in invasive annual cover were greater on masticated plots compared to untreated plots but less than on plots that were cut, piled, and burned (Ross et al. 2012) or broadcast burned (Redmond et al. 2014). However, Redmond et al. (2014) reported the increase in invasive plants was greater on masticated plots than on plots that were cut, piled, and burned regardless if they were seeded or not seeded following treatment. The mixed response of invasive plants among studies is likely related to differences in the pretreatment vegetation composition, posttreatment disturbances (e.g. seeding), and ecological site characteristics (fig. 5-2; Bybee et al. 2016; Ross et al. 2012; Young et al. 2013a).

Shredding and Seeding

Seeding is an important consideration where native perennials are depleted, and invasive annual grasses pose a threat (Bybee et al. 2016). Cheatgrass cover increased after shredding—or seeding and then shredding—but was considerably less (less than 10 percent cover) where perennial herbaceous cover was 43 percent (fig. 5-11). In northwestern Colorado, Stephens et al. (2016) compared no seeding and seeding a native mix of grasses, forbs, and shrubs on untreated and masticated plots in western Colorado. Biomass of perennial grasses was 160 kg/ha in masticated-and-seeded and masticated-and-unseeded treatments compared to 10 kg/ha in untreated plots. There was no difference between seeded and unseeded masticated treatments. The lack of difference between seeding and no seeding may have resulted from the presence of an adequate understory prior to treatment or poor reseeding success.

Figure 5-11—Twelve years following shredding and seeding on a site with a residual cover of sagebrush—resulting in a strong response of shrubs and grasses. Limited mulch remains. Eastern Nevada. (Photo by Rick Miller, Oregon State University.)

But in another study, the perennial herbaceous cover on seeded plots increased 234 percent and 160 percent in masticated and broadcast burn treatments, respectively— compared to only 32 percent on masticated unseeded plots (Redmond et al. 2014). When 22 shredded sites across Utah were compared to adjacent untreated sites, those seeded before shredding (largely to introduced species) had greater cover and density of perennial forbs, tall grasses, and sagebrush than untreated sites (Bybee et al. 2016). Cover and density trended higher on seeded than nonseeded shredded plots, but differences were not statistically significant. Seeding did suppress increases in cheatgrass. For cheatgrass, cover remained around 5 percent when shredding was followed by seeding on plots where pretreatment tree cover was more than or equal to 35 percent. In contrast, cheatgrass cover was more than 16 percent on masticated plots with no seeding when pretreatment tree cover was more than or equal to 25 percent (Bybee et al. 2016). Many of the seeds broadcasted before shredding end up covered or buried by tractors during the shredding process. Proper seed burial can greatly increase revegetation success in semiarid environments (Ott et al. 2003; 2016; Roundy and Call 1988; Stevens et al. 2004). Managers often prescribe seeding in conjunction with shredding if either cheatgrass cover or lack of perennial plant cover is a concern.

### Soil Disturbances Related to Shredding

Soil compaction from the tires of shredding equipment impacted 15 percent of a treatment area in Utah, but compaction effects can be reduced if shredding operations are conducted when soils are dry. Dodson et al. (2006) found soil compaction significantly increased in the upper 4 inches of soil where rubber-tired skidders were used to remove cut western juniper in central Oregon. Compaction of soil decreased infiltration rates, increased runoff and sediment yield, and reportedly increased penetration resistance threefold. However, effects of compaction may be offset by the reduction in bare ground from shredded debris. On masticated plots, infiltration rates were reduced in the tire tracks of the grass interspace (Cline et al. 2010). However, nontire impacted interspace and tree mounds were unaffected, and infiltration rates higher and cumulative runoff and sediment yields lower on interspace plots with shredded residue. Ross et al. (2010) reported that soil aggregate stability was higher on masticated than untreated plots on Shay Mesa, Utah, where bare ground was 26 percent on treated and 54 percent on untreated plots. Shredding also increased percent total nitrogen and soil organic carbon in the top inch of the soil by the second year following treatment (Aanderud et al. 2017; Young et al. 2014).

Fuel Loads and Shredding

Shredding significantly increases surface fuel loads, and the increases in surface fuel biomass are closely related to pretreatment tree cover (figs. 5-7, 5-8, 5-12, 5-13; Young et al. 2015). Changes in surface fuel abundance, structure, and compaction in masticated plots may result in lower fire intensities (Shakespear 2014), but the potential for longer soil heating may result in higher fire severities (see Glossary for definition). Shredding transforms trees into surface fuels of compact 1- and 10-hour size classes, reducing fire intensity and rate of spread (Young et al. 2015). Winter burning on frozen soils has been found to reduce fire effects on woodlands that have been cut (Bates et al. 2006; Bates and Svejcar 2009), but this has not been tested on masticated woodlands.

Cutting

Historically, cutting pinyon and juniper trees for firewood, fenceposts, and conversion to charcoal for silver mines was common in the late 1800s and early 1900s. With the closure of mines and conversion to the use of chemicals for mineral extraction (rather than wood heat), woodcutting for mining significantly declined. Just before World War
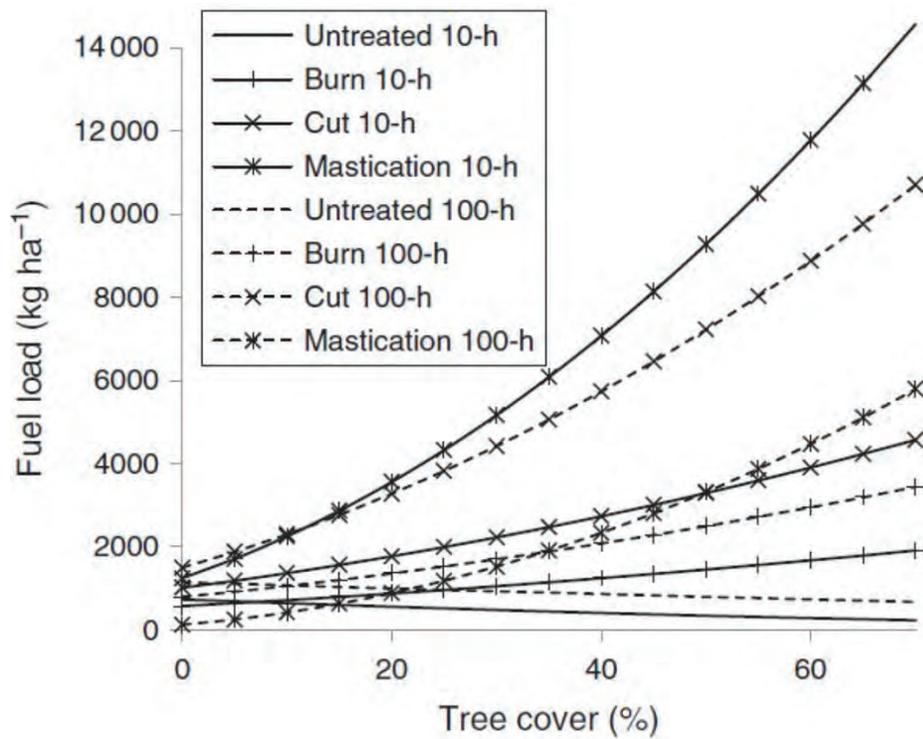
Figure 5-12—Comparison of biomass of surface fuel loads for untreated and three different tree reduction treatments 1-year posttreatment separated by time-lag fuel moisture (TLFM) class (from Young et al. 2015).
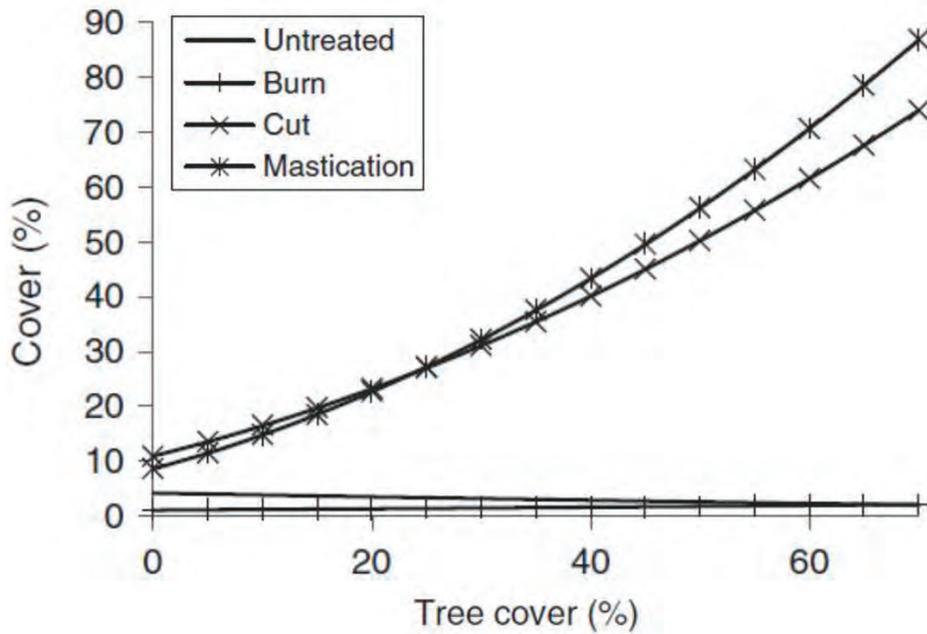
Figure 5-13—Percent cover of surface fuels (dead organic material) 1-year posttreatment in relation to pretreatment tree canopy cover among untreated and three different tree reduction treatments (from Young et al. 2015).

II, woodcutting occurred primarily for firewood and fenceposts. Following the war, pinyon and juniper woodlands cut with chainsaws in the name of restoration and/or forage production between the 1940s through the early 1970s was limited. In Vallentine's (1971) *Range Improvements* textbook, cutting was described as highly selective but expensive because of labor costs. In central Oregon, beginning in the early 1970s, hand-cutting with chainsaws became the primary tool for juniper removal in open stands where understory vegetation was present and could respond to competitive release (Winegar and Elmore 1978). In the late 1970s and early 1980s, cutting expanded into eastern Oregon, northeastern California, and southwest Idaho. By 1976, 12,500 acres of western juniper were thinned or cleared using chainsaws in Oregon at a cost of $5–$28 per acre (Winegar and Elmore 1977). Cutting significantly increased in the mid-'90s especially in the Columbia Basin in central Oregon, in an attempt to increase ground cover for watershed restoration and streamflow to address concerns over the listing of salmon as an endangered species (Tim Deboodt, Retired County Extension Specialist, Oregon State University, Crook County, Oregon, personal communication, 2017). Cutting remains a primary method for conifer removal today, especially for sage-grouse habitat restoration in Phase I and early Phase II woodlands (Miller et al. 2017).

Cutting is a highly selective management method of tree removal. Like shredding, it becomes especially useful when retention of shrubs—especially sagebrush—is a primary management goal. There is also a broad seasonal window for cutting, which is limited primarily by access (e.g. winter snows), fire danger, and species of concern (e.g., sensitive nesting periods). Cutting generally results in minimal soil disturbance and a more gradual release of soil nutrients (e.g., nitrogen) compared to fire. There are many variations of cutting and followup treatments, including cut-and-leave; cut-limb-and-scatter (also called lop-and-scatter); cut-and-broadcast burn; and cut-pile-and-burn—all with advantages and disadvantages (table 5-2).

Table 5-2—Advantages and disadvantages of different variations of hand cutting (Miller et al. 2005, 2013)

| Method	Advantages	Disadvantages
Cut-and-leave	Decreases bare ground. Least expensive cutting method. Highly selective.	Leaves too much material on the ground in mid-phase II and III. Can have smothering effect causing mortality of perennial herbs. Misses small trees requiring follow-up treatment.
Cut-lop-and-scatter	Increases cover of slash. Highly selective. Minimal smothering effect.	Leaves too much material on the ground in mid-phase II and III. Increases cost. Misses small trees requiring follow-up treatment.
Cut-and-broadcast burn	Removes high amounts of slash in Phase II and III. Increases treatment longevity by killing small trees.	Not selective resulting in loss of nonsprouting shrubs. Increases cost. Increases availability of soil nutrients (e.g., N) which can reduce resistance to invasive species
Cut-pile-burn	Removes high amounts of slash in Phase II and III. Selective, minimizes area burned. Broad seasonal window of when piles can be burned. Minimal burn impact on frozen soils.	Increases cost. Some impact to soils. Misses small trees requiring follow-up treatment. Increases availably of soil nutrients but on a smaller proportion of the treated area as compared to broadcast burning

Vegetation Responses to Cutting

Vegetation response following cutting is influenced by ecological site characteristics and pretreatment tree dominance (TDI or Phase I, II, or III; see Glossary for definitions) and understory plant composition at the time of cutting (figs. 5-2 and 5-14; Bates et al. 2000; Everett and Sharrow 1986b; Miller et al. 2014a,b; Roundy et al. 2014a; Williams et al. 2017). The above components are closely linked to the site's resilience and resistance to invasive annuals (Bybee et al. 2016; Miller et al. 2000; Roundy et al. 2014a; Williams et al. 2017). The majority of studies evaluating cut-and-leave treatments reported increases in perennial understory vegetation (Bates et al. 2000, 2007a, 2017b; Everett and Sharrow 1985a,b; Miller et al. 2014b; O'Connor et al. 2013; Rose and Eddleman 1994; Ross et al. 2012; Roundy et al. 2014a; Vaitkus and Eddleman 1987; Williams et al. 2017). Cutting increases the availability of soil nutrients and water and lengthens the growing season by increasing soil water availability by 2 or more weeks (Bates et al. 2000; Roundy et al. 2014b). Current work following 12–13 years of posttreatment by both fire and mechanical tree reduction shows additional wet days in spring more closely related to October-June precipitation than number of years since treatment (Roundy et al. in review).

One disadvantage of cutting (similar to shredding) is that small trees are easily missed during treatment, and sprouting can occur from branch buds that exist below the cut line. With an adequate seed source and/or density of small trees, woodlands can approach Phase II within 25 to 40 years after treatment, depending on soil water and temperature regime (fig. 5-15; Johnson and Miller 2006; Miller et al. 2000; Tausch and Tueller 1977). In central and southeast Oregon, 300 to 500 small trees per acre were present 3 years after cutting, due to posttreatment seedling establishment and because many small trees were left uncut (Bates et al. 2017b; Winegar and Elmore 1978). On 10 sites across the Great Basin, tree saplings were

Figure 5-14—Ten years following cutting of a Phase II woodland with an intact understory. The treatment resulted in a significant increase in shrubs and perennial grasses and forbs. This mountain big sagebrush/Idaho fescue site has high resilience and resistance to invasive annual grasses resulting from relatively cool falls and wet winters and springs. In comparison, locations with warmer fall temperatures are especially susceptible to cheatgrass dominance (Cline et al. 2018b). Modoc Plateau, northeastern California. (Photo by Rick Miller, Oregon State University.)
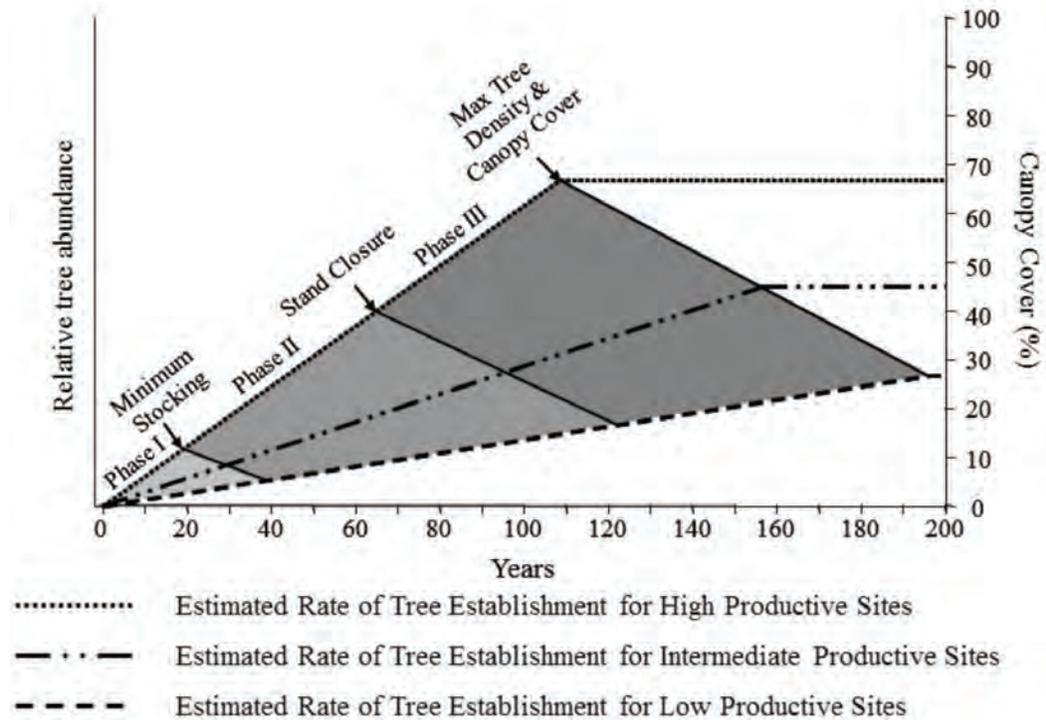
Figure 5-15—The hypothesized time periods from initial tree establishment (early Phase I) to minimum stocking adequate for Phase III, and estimated maximum potential for tree density and cover for stands developing on sites with varying productivity due to differences in elevation and/or exposure (Johnson and Miller 2006).

recorded on 80 percent of cut plots, 53 percent of burned plots, and 32 percent of untreated plots 6 years following treatments (Williams et al. 2017). In southeast Oregon, posttreatment tree cover and density 1 and 25 years following treatment increased from less than 1 percent and 32 trees per acre (all less than 12.6 inches tall) to 4 percent and 132 trees per acre respectively, a level sufficient to fully occupy the site (Bates et al. 2017b).

Shrub Response to Cutting

Shrub retention, especially on warm and dry ecological sites where reestablishment of nonsprouting shrubs can be very limited, is one of the primary advantages of using cutting or cut-pile-burn treatments (fig. 5-16). Cut-pile-burn is a viable option for reducing large amounts of tree slash in Phase II and III woodlands. In Phase I or early Phase II, cut-and-leave is a viable option for maintaining and/or restoring the shrub layer. However, tree removal in Phase III woodlands may not result in quick shrub restoration (Bates et al. 2017b; Roundy et al. 2014a; Williams et al. 2017). Posttreatment increases in total shrub and big sagebrush cover were 25 percent less when the tree dominance index (TDI) exceeded 0.5 (Williams et al. 2017), which is approaching late Phase II. Twenty-five years after cutting in a Phase III woodland, total shrub cover had only increased from 0 to 5.5 percent and sagebrush cover from 0 to 2.8 percent (Bates et al. 2017b).

However, sagebrush density increased from just a trace to 1,700 plants per acre. Sagebrush establishment can also be suppressed by the release of tall grasses when cutting in high tree dominance stands (Phase III) (Roundy et al. 2014a). Across 10 study sites in the Great Basin ranging from cool and moist mountain big sagebrush to Wyoming big sagebrush and mountain big sagebrush communities on cool and moist bordering on warm and dry, sagebrush density 3 years after cutting was $0.41/10$ ft^2 on cut and $0.09/10$ ft^2 on uncut sites (Miller et al. 2014b). On a cool and moist mountain big sagebrush-mountain

Figure 5-16—Two primary advantages of cutting over burning are retention of sagebrush cover if present, and a smaller response of invasive annuals compared to burning. Disadvantages are the large amount of slash left in late Phase II and III woodlands; followup is usually necessary to remove the small trees and resprouting around lower limbs. East-central Nevada. (Photo by Rick Miller, Oregon State University.)

mahogany community in eastern Oregon, mountain mahogany cover was 12.5 percent greater and density twofold greater 4 years after cutting (O'Connor et al. 2013).

Cutting and Perennial Grasses

Cutting often increases perennial grasses within the first year of treatment, whereas prescribed fire often results in a short-term reduction in perennial grasses followed by an increase (Miller et al. 2013, 2014b). The initial increase in perennial grasses is usually attributed to increased plant size rather than density, emphasizing the importance of pretreatment vegetation composition (Bates et al. 2000; Everett and Sharrow 1986b; Miller et al. 2014b). Increases in perennial grasses following cutting treatment range from 1.4 to 20-fold (Bates et al. 2005; Everett and Sharrow 1985a,b; Miller et al. 2014b; O'Connor et al. 2013; Rose and Eddleman 1994; Ross et al. 2012; Roundy et al. 2014a).

Across a broad range of sites in Oregon, northeastern California, southwestern Idaho, Nevada, and Utah, perennial grass cover was 1.8 times greater than adjacent uncut areas 3 years following treatment (Miller et al. 2014b). In central Nevada, perennial grass cover increased from 5 to 15 percent and 2 to 13 percent following cutting on cool north and west aspects, respectively (Everett and Sharrow 1985b). On warm and dry southern aspects, cover only increased 1 to 3 percent. In eastern Oregon, perennial grass cover was less than 3 percent before cutting but was 10 percent and 13 percent in the 6th and 14th posttreatment years, respectively (Bates et al. 2005). In Phase III woodlands, perennial grass biomass was less than 50 pounds per acre on uncut and 660–1,000 pounds per acre on adjacent cut in the 6th and 14th posttreatment years, respectively. However, 25 years after treatment, perennial grass yields declined from peak posttreatment levels by 30–40 percent (Bates et al. 2017b). The reason for the decline was unknown, but it may be related to an increased presence of woody vegetation or climatic factors. In

western Colorado, where a Phase III tree canopy was 30 percent, perennial grass cover was less than or equal to 1 percent in uncut compared to 11 percent in cut stands 2 years posttreatment (Ross et al. 2012). The majority of studies reported little or no changes in Sandberg bluegrass cover with cutting (Bates et al. 2005, 2017b; Miller et al. 2014b; Vaitkus and Eddleman 1987; Williams et al. 2017), although Bates et al. (2017b) reported a significant increase in Sandberg bluegrass density, which persisted 25 years.

Across multiple sites in the northern and central Great Basin, tall perennial grass cover on plots with high TDI (Phase III) increased 5 percent to nearly 15 percent in the third and over 22 percent in the sixth posttreatment years, resulting in a fourfold increase (fig. 5-17; Roundy et al. 2014a; Williams et al. 2017). In more open conifer stands where TDI was low, perennial grass cover was 17 percent prior to cutting and 22–24 percent following cutting. In comparing cutting and shredding, the response of tall perennial grass cover was similar between the two treatments (Bybee et al. 2016). However, cutting in high TDI or Phase III, can result in greater increases in invasive annual grasses (Roundy et al. 2014a), especially where perennial grass cover is less than 5 percent.

Cutting and Perennial Forbs

Perennial forb cover is typically a relatively small portion of total perennial herbaceous cover and biomass in sagebrush communities and conifer expansion woodlands. In sagebrush communities dominated by Wyoming big sagebrush, basin big sagebrush, and black sagebrush, perennial forb cover is often less than 6 percent (Bates et al. 2017a; Davies et al. 2006; Miller et al. 2014b; Ross et al. 2012; Roundy et al. 2014b; Stringham 2017). In late Phase II and III woodlands, forb cover is usually less than 2 percent (fig. 5-17; Ross et al. 2012; Roundy et al. 2014a; Williams et al. 2017). In several studies, perennial forb cover nearly doubled in cut plots but still remained less than 8.5 percent (Bates et al. 2017a; Miller et al. 2014b; Ross et al. 2012). Increases in perennial forbs following cutting is highly variable. Within the first 3 years of cutting, studies reported increases ranging from 1.5–10 times that of uncut controls (Bates et al. 2017a,b; Miller et al. 2014b; Ross et al. 2012; Roundy et al. 2014a). The wide variation in perennial forb response can be attributed to one or more of the primary components in figure 5-2. The greatest potential increase in perennial forbs following disturbance is on cool and moist compared to warm and dry soils (Miller et al. 2013). In addition, the amount of increase following cutting closely correlated to TDI (Roundy et al. 2014a), but a dry year can modify this response (Williams et al. 2017).

The longevity of increased perennial forb abundance following cutting is considerably shorter than increased levels of perennial grasses (Bates et al. 2017a,b). While undoubtedly the longevity of the perennial forb response is linked to ecological site conditions, posttreatment weather, and foraging pressure, the persistence of increased perennial forb abundance is typically shorter than increased levels of perennial grass abundance (Bates et al. 2017a,b; Williams et al. 2017). In eastern Oregon, perennial forb biomass remained higher than that of adjacent uncut Phase III woodlands for 16 years following cutting. At posttreatment year 24, perennial forb biomass was similar to uncut plots, whereas abundance of perennial tall grasses was still threefold greater in cut plots 24 years postcut (Bates et al. 2017a,b). In Utah, perennial forb cover was initially greater in cut plots but declined to similar levels in untreated plots by 6 years following treatment, possibly a result of increased tall grass cover or below average precipitation (Williams et al. 2017). Type of treatment can also influence the magnitude of increases in perennial forbs. Although cutting can result in increased perennial forb abundance, increases are usually greater in prescribed burn treatments (Miller et al. 2013; 2014b). On 11 study sites across the Great Basin, perennial forb cover was 1.8-fold greater on cut treatments and 2.6-fold greater on prescribed burned compared to adjacent untreated plots (Miller et al. 2014b).
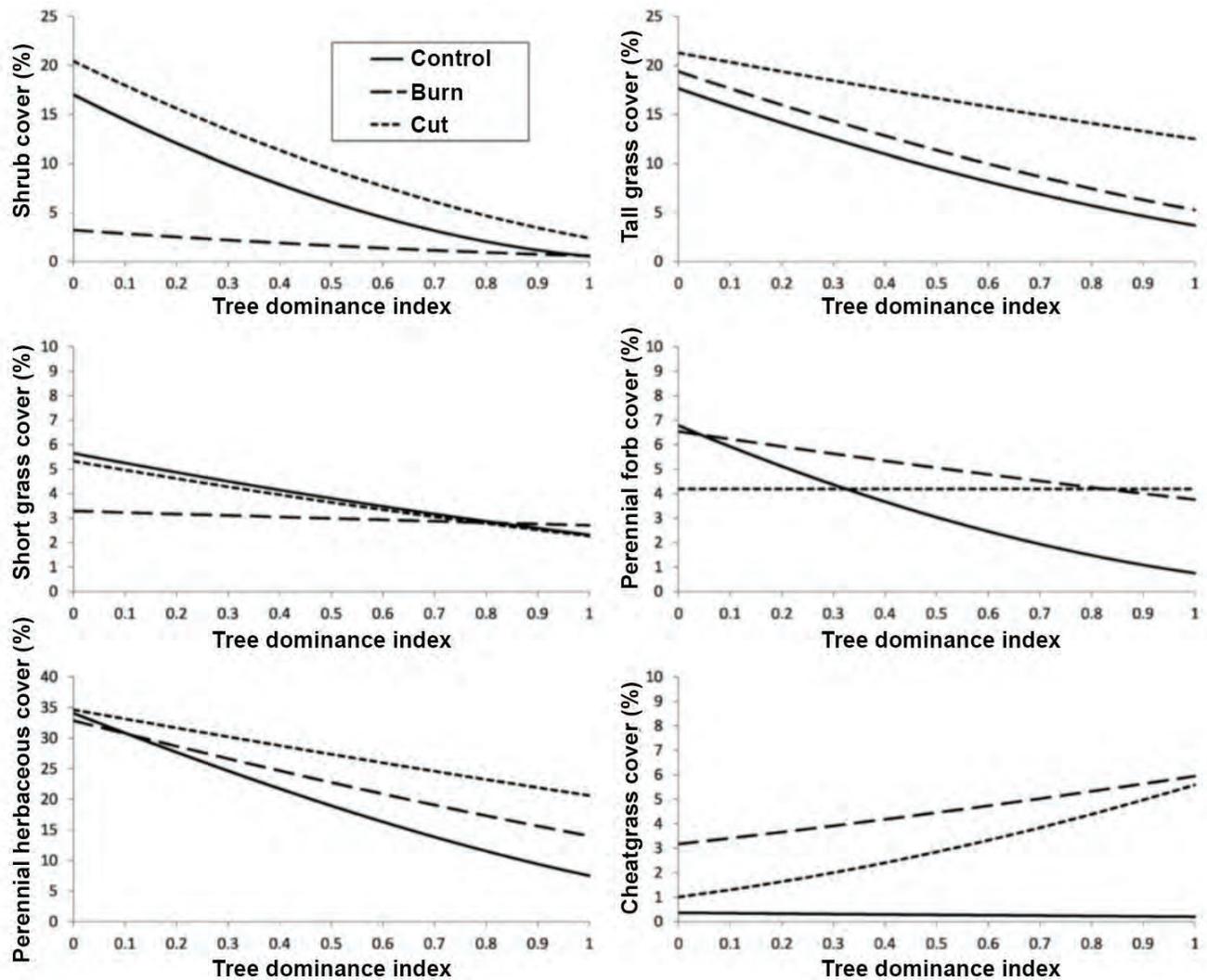
Figure 5-17—Vegetation cover 3 years after treatment in relation to pretreatment tree dominance index (TDI) for multiple sites across the Great Basin (from Roundy et al. 2014a).

Invasive Annual Plant Response to Cutting

The presence of cheatgrass and other nonnative annuals often increases following cutting (Bates et al. 2000, 2005, 2017b; Miller et al. 2014b; Roundy et al. 2014a). However, the magnitude and timing of these increases depends on pretreatment plant composition, soil moisture and temperature regimes, and how the severity of the treatment influences the seed pool, disturbance of the soil surface, and mortality of understory perennial vegetation (Miller et al. 2013). In western Colorado, invasive annual grasses increased just slightly and accounted for less than 2 percent of the total understory cover in an area where annual grass cover was only a trace in control plots (Ross et al. 2012). Across 24 sites in the Great Basin, cheatgrass cover generally decreased with increasing perennial cover, but the ratio of cheatgrass to perennial grass cover increased exponentially with pretreatment tree dominance (TDI values above 0.7, see Glossary) (fig. 5-18; Roundy et al. 2014a). Baughman et al. (2010) reported a similar relationship was reported in east-central Nevada following thinning trees to 25 per acre in Phase II and III sites. Increasing levels of invasive annuals were significantly greater on treated Phase III sites where perennial understory vegetation was less abundant.
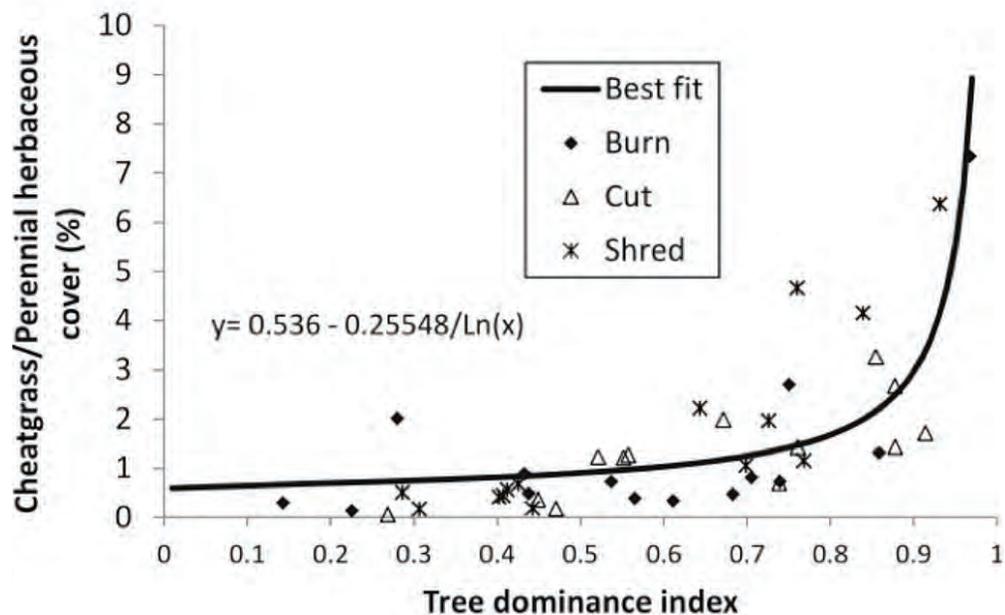
Figure 5-18—Ratio of cheatgrass to perennial herbaceous cover in relation to pretreatment tree dominance index (TDI) for burn, cut, and shred tree control treatments at Scipio, Utah ($r^2 = 0.64$) (Roundy et al. 2014a).

Soil moisture and temperature regimes are also linked to the magnitude of increase in invasive annual plants. Invasive annual cover 3 years after cutting was greater than adjacent untreated plots across multiple sites in the Great Basin (Miller et al. 2014b). However, there were distinct differences between warm and dry and cool and moist sites. Invasive annual cover was less than 5 percent on cut cool and moist ecological sites compared to 18 percent on the warm and dry sites (Miller et al. 2013). Roundy et al. (2014) reported cheatgrass cover remained low (below 6 percent) after cutting on multiple cool sites across the Great Basin. However, cheatgrass cover on warmer sites increased significantly from 19–27 percent after cutting treatments. Across 17 sites in the Great Basin, locations with warmer fall temperatures were especially susceptible to cheatgrass dominance, while those with cooler falls and relatively wet winters and springs were more resistant because they more frequently supported higher perennial herbaceous dominance (Roundy et al. 2018).

Within treatment sites, increases in cheatgrass cover can be greater beneath dropped trees and in tree-litter mats that result in mortality of perennial grasses from heavy shading (Bates et al. 2007a). Few long-term studies have followed the persistence of invasive annuals following treatment. In a 13-year study, cheatgrass cover generally remained less than 5 percent following cutting, but cover varied with precipitation and reached 15 percent cover in a year when precipitation was 200 percent of average (Bates et al. 2005).

Although cutting often results in increased invasive annual abundance, the increase is usually less than that following prescribed burns (Miller et al. 2014b; O'Connor et al. 2013; Roundy et al. 2014a; Williams et al. 2017). Cut-pile-burn can also result in large increases in invasive plants, resulting from high mortality of perennial grasses directly beneath the pile and large increases in nitrogen. In northern Arizona, treatment by burning slash piles resulted in a 50-fold increase in soil ammonium concentrations and a 20-fold increase in nitrate (Covington et al. 1991). The increases in inorganic nitrogen disappeared in 5 years. On 11 conifer-encroached sagebrush sites across the Great Basin, cheatgrass and exotic forbs increased after cutting and fire, but the increase was slightly more than 1.5 times greater in burned than cut sites (Miller et al. 2014b). On cool and moist ecological sites, invasive annual cover remained below 5 percent—but was more

than 30 percent on burned and 18 percent on cut warm and dry ecological sites (Miller et al. 2013). In this same regional study, cheatgrass and nonnative annual forb cover were similar on untreated and cut plots by the sixth posttreatment year but remained greater on burned pretreatment sites where tree dominance was high (Williams et al. 2017). Low levels of cheatgrass and nonnative forb cover were associated with higher tall grass cover on cut plots or on burn plots at lower pretreatment tree dominance.

Chemical Treatment

Widespread herbicide uses on rangelands did not occur until after World War II. Even then, use of herbicides to kill pinyon and juniper was primarily experimental, and no herbicides were registered for use on grazing lands by the Environmental Protection Agency until 1975 (Evans et al. 1975). The majority of herbicide research related to pinyon and juniper was conducted in the Colorado Plateau and northern Arizona, with few herbicide studies completed in the Great Basin. Possibly the earliest herbicide studies on pinyon and juniper were investigations on foliar applications of arsenite in the early 1940s (Parker 1945), which was never registered for use on grazing lands (Evans et al. 1975). Herbicides tested in the 1950s and '60s had mixed results with respect to tree-kill and understory response—and most never became management tools (Evans et al. 1975; Johnson 1967). By the 1970s, Picloram™ and Tebuthiuron™ showed some promise for juniper control. Picloram™ is a systemic herbicide used to control woody plants and broad-leaved forbs, which can be sprayed, injected, applied to cut surfaces, or applied to the soil near plant roots. Tebuthiuron™ is a broad-spectrum systemic herbicide used to control herbaceous and woody plants. It is usually applied to the soil near plant roots. The morphology of juniper, including dense canopies, thick leaf cuticles, and stomates on the underside of leaf scales (fig. 2-16b), make them relatively resistant to foliar chemical application (Johnson 1967; Miller and Shultz 1987). Soil herbicide applications had greater control potential because of the extensive, shallow root systems of pinyon and juniper (Young et al. 1984). In the early stages of the transition of western juniper into sagebrush communities, spot treatment or basal bark application of Picloram™ resulted in 90–98 percent mortality of small trees (Sbatella and Twelker 2013). This would also support the effectiveness of spot treatment following mechanical control. Spring application of a mixture of Tordon™ (Picloram), Milestone™ (aminopyralid), and the surfactant Alligare 7™ was also effective in killing trees of less than 4 inches with 100 percent mortality, and trees 4–12 inches with 98 percent mortality (Monument Soil and Water District 2017).

Responses to Chemical Treatment

In the early 1960s, Picloram™, an herbicide with both foliar and soil applications, was tested in both spray and pellet form in the eastern United States (Watson and Wiltse 1963; Wiltse 1964). It showed promise for control of juniper in the West (Johnson 1967). Picloram™ is nontoxic to mammals but persists in the environment. It is not strongly absorbed by soil colloids, which allows it to move freely in water (Evans et al. 1975), raising concerns about potential contamination in runoff from treated watersheds (Johnson 1967; Vallentine 1971). Typically, only limited amounts (less than 1 pound per acre) is applied for woody plant control. In the late 1970s and early 1980s, Tebuthiuron™ was another herbicide evaluated for juniper and pinyon control (Johnsen and Dalen 1990; Van Pelt and West 1993). However, cool season perennial grasses are sensitive to relatively low rates, resulting in greater mortality compared warm season grasses. There are also concerns about Picloram's™ longevity and the potential for contamination in groundwater.

Since the 1970s, a combination of Picloram™ and Tebuthiuron™ have been commonly used, especially with mechanical treatments for pinyon and juniper control. Both often

have mixed results when used alone. They can be effective following mechanical tree removal when applied selectively on small trees (Young and Evans 1976) or on stumps with live limbs. In western Nevada, Picloram™ was used following chaining to kill small sapling trees (Young and Evans 1976). Although the followup treatment resulted in greater herbaceous production, the increase in biomass was largely from cheatgrass—even though crested wheatgrass was present on the site.

Several studies report large increases in annual invasive grasses following control of pinyon and juniper (Evans and Young 1985; Roundy et al. 2014a; Young and Evans 1976; Young et al. 1985). In all cases, resistance to invasive annuals was low due to severely depleted native perennial grass and forb communities and in sites where soil moisture temperature regimes were warm and dry. On these sites, the increase in soil moisture and nitrogen following tree removal were quickly exploited by invasive annuals already on the site and in the soil seed bank. The most common use of herbicides associated with tree removal is for posttreatment control of invasive annuals on low-resistance ecological sites. The most commonly used herbicides for invasive annual control following pinyon and juniper treatment is Imazapic (Plateau)™.

Prescribed Fire

Prescribed fire was little used in the first few decades following World War II (fig. 5-19). In the early part of the 20th century, many ecologists supported the active suppression of fire, a European philosophy that spilled over to the management of plant communities in North America (Wright and Bailey 1982). But early in the 1900s, some people, including Leopold (1924), recognized the importance of fire for ecosystem health. The Leopold Report of 1963 (Leopold 1963) reported that fire prevention could lead to an accumulation of fuels as well as changes in plant community composition and structure. Some of the earliest research on prescribed fire in pinyon and juniper woodland was conducted on the Hualapai Indian Reservation in northwestern Arizona in 1953, when an experimental prescribed fire escaped and burned 16,000 acres (Arnold et al. 1964; McCulloch 1969; Schroeder 1966). After additional prescribed burning and monitoring over the next 11 years, Schroeder (1966) concluded that burning of mature pinyon and juniper woodland was only successful during peak fire weather conditions, which occurred only a few days a year—and in some years never occurred. He found that woodlands with higher proportions of pinyon carried fire better than stands with higher proportions of juniper.

In possibly the first synthesis on burning pinyon and juniper woodlands, Blackburn and Bruner (1975) concluded results of these early studies were mixed and generally focused on the postfire response of perennial grasses. They suggested that widespread use of prescribed fire was limited by "the extremely hazardous burning conditions necessary to treat such large areas and the fact that the Grand Canyon is not available to everyone for a firebreak," (as exists on the Hualapai Indian Reservation). The required extreme weather conditions discouraged land managers from using prescribed fire. Burning mature pinyon and juniper stands with little understory was found to be extremely difficult, requiring hazardous weather conditions and/or pretreatment to increase fuel continuity (Arnold et al. 1964; Aro 1971; Wright et al. 1979). Federal and State agencies were hesitant to use prescribed fire for fear of escaped fires. By the early 1970s, prescribed fire treatments on Federal lands were uncommon (Aro 1971). But following an evaluation of 50 sites across the Colorado Plateau, Aro (1971) recommended the use of burning rather than mechanical treatments where fuels were adequate to carry fire. The presence of adequate fuels and significant increases in postfire perennial grasses reported by Aro (1971) strongly suggests these sites had an adequate abundance of preburn perennial

Figure 5-19—In the early part of the 20th century, many ecologists supported the active suppression of fire, a European philosophy that spilled over to the management of plant communities in North America. But early in the 1900s, Leopold recognized the importance of fire for ecosystem health. Northcentral Oregon. (Photo by Rick Miller, Oregon State University.)

grasses to recover after the fire. In the 1970s and '80s, acres treated with prescribed fire increased throughout the Intermountain West. Between 2002–2016, 5.5 million acres were prescribed burned across public lands in Oregon, Idaho, California, Nevada, Utah, Colorado, and Arizona (National Interagency Fire Center 2018), which accounted for 12 percent of the total area burned across these states over the same period.

Responses to Fire

Today the increasingly overwhelming problem of invasive annual plants across the Intermountain Region requires cautious consideration when deciding to use prescribed burning treatments. To predict postfire vegetation response, important considerations are prefire vegetation, ecological site characteristics, and potential fire severity, all of which affect resilience and resistance to invasive annuals (table 5-3; figs. 5-2, 5-19) (Chambers et al. 2016a; Dhaemers 2006; Miller et al. 2013, 2014a, 2015; Roundy et al. 2018; Urza et al. 2017). Tree dominance at the time of treatment can also have a significant impact on understory plant composition and fire severity (Bates et al. 2011; Roundy et al. 2014a; Williams et al. 2017).

Tree Mortality and Fire Severity

Tree mortality in burned pinyon and juniper stands is a function of fire weather and fuels, which includes wind speed (and/or slope), relative humidity, air temperature, tree size, and moisture content—as well as fuel structure, abundance, and continuity (Martin 1978; Wright et al. 1979). One of the primary advantages of burning is the high mortality of small trees, which are often missed during mechanical treatments and require followup (Arnold et al. 1964; Dwyer and Pieper 1967; Jameson 1962; Martin 1978). Prescribed fire easily kills trees less than 4 feet tall even under relatively mild to moderate weather

conditions. However, mortality of trees over 6 feet tall is often less than 70 percent, except under extreme fire weather conditions (fig. 5-20; Dwyer and Pieper 1967; Martin 1978; Wright et al. 1979) or when sufficient surface and ladder fuels are present, like those occurring in Phase I and early Phase II woodlands where understories are intact (Miller et al. 2013).

There is a geometric decrease in fire intensity with the decline in percent cover and height of sagebrush, which are important ladder fuels (fuels that connect the surface fuels to the tree canopy) in pinyon and juniper woodlands (fig. 5-21; Brown 1982). The

Table 5-3—Advantages and disadvantages of different vegetation treatment methods commonly used on pinyon and juniper woodlands.

| Method	Advantages	Disadvantages
Chemical	Effective as a spot treatment on small trees or on stumps with remaining basal branches following mechanical treatments.	Mixed results when used alone.
Chaining	Improved establishment of broadcast seeding between double chaining. Majority of studies reported increases in perennial grasses when seeded. Chain + seeding decreases runoff and sediment loss.	Misses the small trees requiring a follow-up treatment. Some studies have reported increases or no change in runoff or sediment production, which is closely linked to posttreatment herbaceous cover. Limited by steep topography. Expensive.
Windrowing	Increases mortality and life of treatment compared to chaining. Invasive annual increase usually less than following prescribed fire.	Expensive and increases soil surface disturbance. Usually requires seeding.
Shredding	Minimal impact on shrubs and biological crust. Decreases bare ground. Compaction of fine fuels reduces the hazard for wildfire.	Tire compaction increases runoff and sediment yield on about 15 percent of the treatment area however compensated by decrease in bare ground. Potential for a small to moderate increase in invasive annuals. Mulch depths of > 5 m can inhibit emergence of perennial grasses. Misses small trees.
Hand Cutting	Minimal impact on shrubs and biological crust layer. Minimal surface disturbance. Invasive annual increase usually less than following prescribed fire. Wide seasonal window. Usually a first-year increase in perennial grasses and forbs. Decreases bare ground especially with lop-and-scatter. Costs are low in phase I.	Leaves large amounts of slash when treating mid and late phase II and phase III. Significantly increases 1- and 10-hr surface fuels in the first 3 years. Heavy slash can have a smothering effect resulting in mortality of perennial herbage. Cost are high in phase III and require follow-up treatment of slash. Potential for a small to moderate increase in invasive annuals. Misses small trees.
Prescribed Fire	Treat large areas usually at lower cost than other methods. Effective in killing small trees. Removes slash, especially important in late phase II and III. Usually results in an increase in residual perennial herbs in the second or third postfire year.	High impact on nonsprouting shrubs. Usually results in the highest increase in invasive annuals although the magnitude of response is closely linked to ecological site characteristics and preburn plant composition. Narrow seasonal window of application and weather conditions can be unpredictable on proposed dates of ignition. Pretreatment deferment from grazing to allow for accumulation of surface fuels. Pretreatment of fuels often required in late phase II and III. High fire severity can result in > 50 percent mortality of perennial grasses. Temporary increase in wind and water erosion dependent on recovery of vegetation cover. Can have a high impact on biological crusts, with moderate to high severity fires.
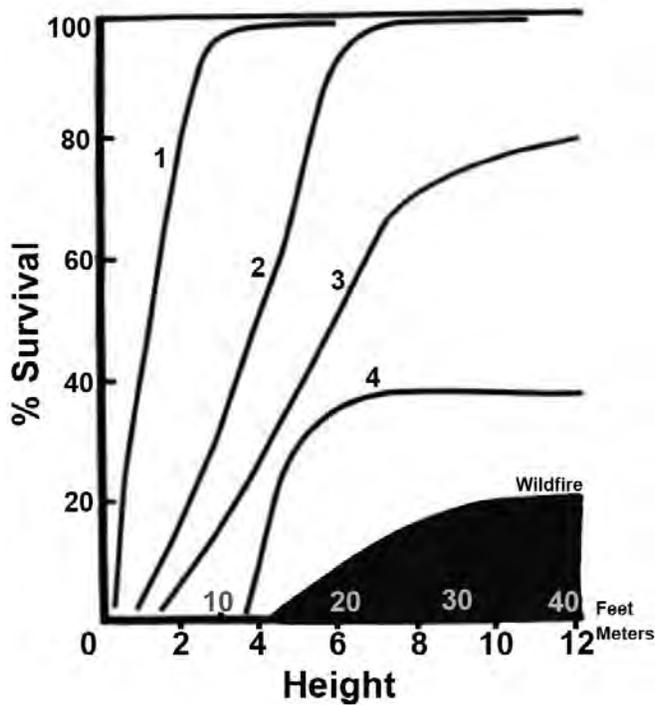
Figure 5-20—First year survival of western juniper in juniper-sagebrush-bunchgrass communities following four prescribed burning conditions; winds 5 to 12 mph, and conditions: (1) 70 ˚F, RH 25 to 30 percent; (2) 70 to 85 ˚F, RH 25 to 30 percent: (3) 75 to 85 ˚F, RH 18–20 percent; and (4) 80 ˚F, RH 10 percent. Under conditions 1 and 2, survival of larger juniper was nearly 100 percent, whereas 73 and 37 percent of the larger trees survived the more severe conditions of 3 and 4. Wildfire conditions resulted in 20 percent or less tree survival (Martin 1978).
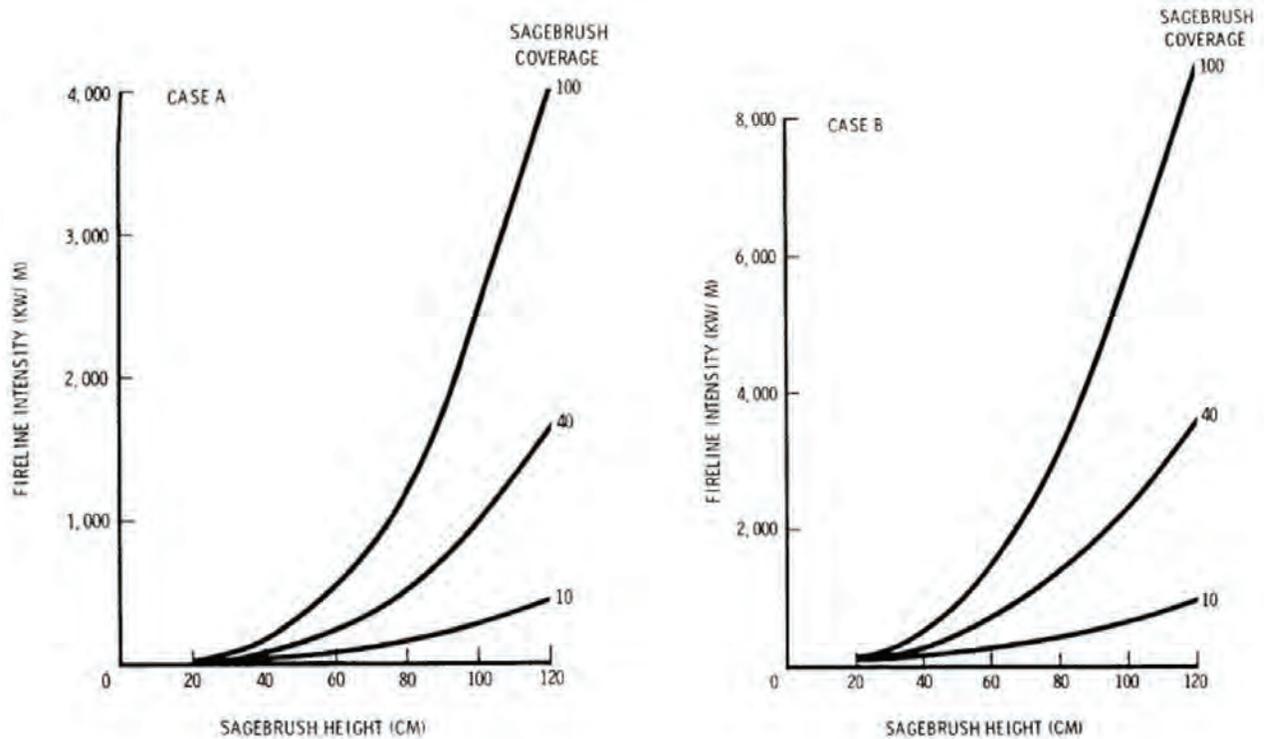
Figure 5-21—Fireline intensity at 8-mi/h midflame-height windspeed and 900 pounds per acre of grass and forbs. (A) Grass and forbs are 57 percent alive and sagebrush entirely alive. (B) Grass and forbs are entirely dead and sagebrush foliage is one third dead (Brown 1982). Fireline intensity is the rate of energy released per unit length of fire front (units kilowatts/meter).

proportion of dead surface fuels also significantly influences intensity. As woodlands mature, height and cover of sagebrush and other shrubs, necessary to support crown fires under moderate weather conditions, decrease as does fine surface fuel abundance and continuity necessary for the spread of surface fires. As understory fuels decline, increased wind speeds, higher temperatures, and decreased relative humidity are required for successful burning, but these conditions can result in higher intensity and potentially greater severity fires. Grazing may also be deferred to allow for the accumulation of fine fuels for successful application of fire (Arnold et al. 1964; Humphrey 1962).

Wright et al. (1979) reported that successful prescribed burning of late Phase II and III woodlands without prior treatments requires air temperatures of 95 to 100 ˚F, low relative humidity, and wind speeds of between 8–20 mph, although minimum wind speeds of 3 mph have been reported to successfully carry fire (Dicus et al. 2009; Martin 1978). Unstable atmospheric conditions also increase the spread of fire by influencing convection, which is the primary means of heat transfer in wildfires. Wind speeds required for successful burning depend on topography and fuel sizes, abundance, structure, continuity, and moisture content. To burn mature woodlands with low amounts of understory fuels (e.g. Phase III) during moderate weather conditions, pretreatments such as tree cutting a portion of the tree overstory to add surface fuels can be done the fall or winter prior to burning (fig. 5-22; Bates and Svejcar 2009; Bates et al. 2011). In woodlands with 40–60 percent tree canopy cover, cutting 25 percent of mature trees in the fall or winter prior to burning was adequate to kill the majority of remaining trees under moderate fire weather conditions (Bates et al. 2011).

Shrub Response to Fire

One of the biggest concerns managers have with burning Phase I and Phase II pinyon and juniper woodlands is the temporary loss of nonsprouting, fire intolerant shrubs such as sagebrush (Davies et al. 2014; Miller et al. 2000; Roundy et al. 2014a). Recovery of the shrub canopy to 20–30 percent cover on cool and moist mountain big sagebrush communities usually takes 20–35 years following a fire (Barney and Frischknecht 1974; Bunting et al.

Figure 5-22—When burning a Phase II or III woodland it is often necessary to cut a portion of the trees in the fall prior to burning to increase surface fuel loads. This also allows the burn to be conducted under less severe weather conditions. Steens Mountain, eastern Oregon. (Photo by Rick Miller, Oregon State University.)

1987; Lesica et al. 2007; Miller and Heyerdahl 2008; Moffet et al. 2015; Nelson et al. 2014; Ziegenhagen 2003; Ziegenhagen and Miller 2009).

In complex, patchy burns where a big sagebrush seed source remains after a fire, recovery can occur within 9–15 years (Pyle and Crawford 1996; Ziegenhagen 2003). However, in high-severity burns in late Phase II and Phase III woodlands, big sagebrush seed is often unavailable because of the scarcity of mature plants in the prefire understory and/or combustion and mortality of seed in the seed bank. Following high-severity fires in Phase II and III woodlands, big sagebrush recovery comes from an outside seed source or artificial seeding (Davies et al. 2014). Persistence of the postfire shrub layer once reestablished depends on the rate of conifer reestablishment and growth (fig. 5-15; Johnson and Miller 2006). In west-central Utah and northeastern California, sagebrush was the dominant woody layer 30–60 years following fire but declined to less than 1 percent cover on burns 80–100 years old as a result of increasing juniper dominance (fig. 5-23; Barney and Frischknecht 1974; Miller and Heyerdahl 2008). Aspect, as it relates to temperature and moisture, also influences big sagebrush establishment, which is generally more rapid on the
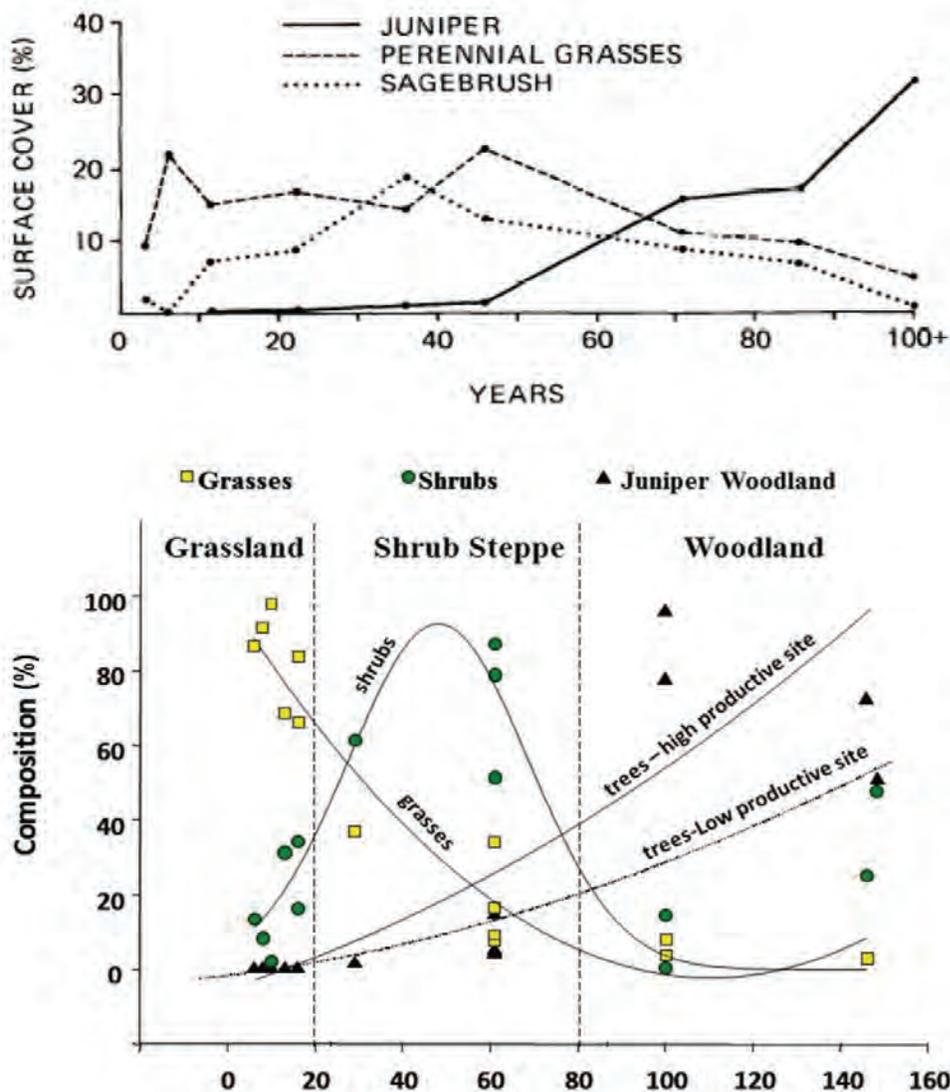
Figure 5-23—Plant succession models following fire for (A) surface cover of juniper, perennial grasses and sagebrush by age of fire in west central Utah (Barney and Frishknecht 1974); and (B) percent composition based on plant cover in cool/moist mountain big sagebrush in northeastern California (Miller and Heyerdahl 2008).

cool and moist versus warm and dry sites (Chambers et al. 2017; Davies and Bates 2017). Postfire recolonization of Wyoming big sagebrush is typically very slow to nonexistent (Miller et al. 2013). Wyoming big sagebrush cover 18–25 years following fire was less than 5 percent (Lesica et al. 2007; West and Yorks 2002).

Bitterbrush is a weak resprouter (Nord 1965), and its postfire sprouting and persistence are highly variable due to differences in genetics, soil site characteristics, soil moisture conditions, fire severity, and the age and phenology of plants at the time of burning (Blaisdell and Mueggler 1956; Clark et al. 1982; Cook et al. 1994; Nord 1959, 1965; Riegel et al. 2006). Residual seed in the soil depends on prefire seed input and fire severity and is likely scarce in Phase III woodlands. An important mechanism of bitterbrush seedling establishment is dispersal and caching by rodents (Nord 1965; Sherman and Chilcote 1972; Vander Wall 1994; West 1968), which typically occurs within 80 feet of the mother plant (Vander Wall 1994). Mountain mahogany is also fire sensitive and a weak postfire resprouter. Four years following a cut-and-broadcast burn, mountain mahogany cover and density declined from 8 percent to less than 1 percent and 71 to 30 plants per acre, respectively (O'Connor et al. 2013). In adjacent stands that were only cut, mountain mahogany cover was unchanged from pretreatment levels but density increased from 71 to 162 plants per acre in the fourth posttreatment year.

Common resprouting shrubs, green and grey rabbitbrush, and snakeweed are consistently found in recently burned areas (Barney and Frischknecht 1974; Wright et al. 1979). However, their abundance depends on prefire vegetation composition, fire severity, and posttreatment management (Miller et al. 2013). They can establish rapidly in open areas where competition from other perennials is low (Barney and Frischknecht 1974). Green rabbitbrush and snakeweed presence often increases following fire and can remain abundant for 10–36 years, at which point they begin to decline in the absence of frequent disturbance (Barney and Frischknecht 1974; Harniss and Murray 1973; Young and Evans 1974). Green rabbitbrush is a tap-rooted species and is short-lived and less competitive than big sagebrush (which has a lateral root system). Seedling density, flower production, and shoot growth of green rabbitbrush decline as competition from other species increases (McKell and Chilcote 1957; Young and Evans 1974). Snakeweed is also a relatively short-lived (20 years) tap-rooted species (Mozingo 1987; USU Ext 2016). Grey rabbitbrush is more sensitive to fire severity, sprouting from stems rather than the base as green rabbitbrush and snakeweed do (Wright et al. 1979).

Fire and Perennial Grasses

More than 80 percent of studies evaluating plant response following prescribed fire in sagebrush ecosystems across the Great Basin reported that the cover of native deep-rooted perennial grasses declined the first year following a fire—but recovered to or above preburn levels by the second or third year (Miller et al. 2013). In no study did perennial grass cover increase in the first year postfire. In contrast, cut-and-leave treatments without burning resulted in an immediate increase in perennial grass cover (Miller et al. 2014b; O'Connor et al. 2013; Roundy et al. 2014a; Williams et al. 2017). This pattern of first-year decline followed by recovery of deep-rooted perennial grasses is common in sagebrush ecosystems throughout the Great Basin (Bates and Svejcar 2009; Bates et al. 2011, 2014; Dhaemers 2006; Miller et al. 2014b; O'Connor et al. 2013; Roundy et al. 2014a; Williams et al. 2017). In 11 conifer expansion sites across the Great Basin, tall perennial grass cover declined 35 percent in the first postfire year but was 146 percent greater than unburned sites in the third postfire year (Miller et al. 2014b). By the third and fourth years, herbaceous cover and density were similar to cut treatments (Miller et al. 2014b; O'Connor et al. 2013). However, sagebrush encroached communities with high tree dominance (Phase III) typically had lower understory vegetation response compared to communities with lower tree dominance

(Dhaemers 2006; Williams et al. 2017). Perennial grass abundance typically peaks within the first 10 postfire years and can remain an important component of the plant community for 20–35 years following fire, after which gradual declines are likely (fig. 5-23; Barney and Frischknecht 1974; Miller and Heyerdahl 2008).

Increased cover of perennial grasses in the early postfire years is generally due to increased plant size rather than increased plant density (fig. 5-24a; Bates et al. 2009, 2011; Everett and Sharrow 1985a,b; Miller et al. 2014b). Increased growth during the first 3 postfire years is associated with the longer period of soil water availability and increased nutrients resulting from the reduction of trees (Bates et al. 2000; Roundy et al. 2014b). Increases in the density of perennial grasses do not typically occur until 3 or more years following fire (Bates et al. 2011; O'Connor et al. 2013; Williams et al. 2017). Therefore, the occurrence and relative abundance of prefire perennial vegetation is more important to early and mid-succession than the soil seed bank (fig. 5-24b; Allen et al. 2008b; Everett and Sharrow 1985a; Everett and Ward 1984; Koniak and Everett 1982; Pekas 2010).

In addition to prefire understory plant composition and tree dominance, fire intensity can have a significant impact on perennial grass recovery (fig. 5-25). Low- to moderate-intensity fires with limited duration generally produce minimal perennial grass mortality (Miller et al. 2014b). However, high intensity and/or longer duration fires in late Phase II and III woodlands resulted in mortality rates above 50 percent (O'Connor et al. 2013) and as much as 85 percent mortality (Bates et al. 2011). A high-severity fire in Phase III woodlands reduced perennial bunchgrass cover from 5 percent to less than 1 percent (Bates et al. 2014). Nine years after the fire, cover had increased to near preburn levels. High mortality can mean slow postfire recovery and low resistance to invasive annuals. In adjacent burned Phase II woodlands, pretreatment perennial bunchgrass cover was 10 percent and declined to nearly 2 percent in the first year following fire, but 9 years later increased to nearly 20 percent, compared to only 5 percent in Phase III. Although mortality was considerable in both phases, the density of perennial bunchgrasses in Phase II the first year following fire was nearly 4 plants per 10 square feet compared to less than 1 plant per 10 square feet in the Phase III woodland. Unfortunately, limited work has investigated the number of deep-rooted perennial grasses required for recovery, which varies with ecological site factors, especially soil water and temperature regimes.
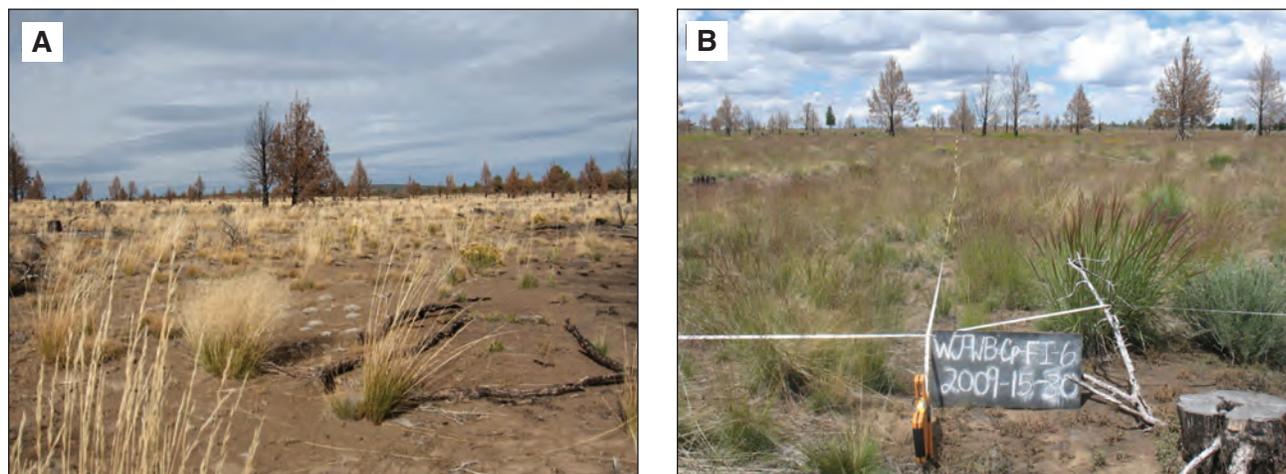
Figure 5-24—Prescribed burn in cool/moist mountain big sagebrush/Idaho fescue community in Phase I and early Phase II in postburn years (A) 1 and (B) 3. Prior to the burn there was a high mortality of sagebrush from an Aroga moth infestation. Central Oregon. (Photo by Rick Miller, Oregon State University).

Figure 5-25—This high severity wildfire is made evident by 90 percent consumption of surface organic matter, few remaining shrub skeletons, consumption of tree needles and bark, and high mortality of perennial grasses, if they were present prior to the burn. High fire severity in this Phase II mountain big sagebrush woodland (a 12- to 14-inch precipitation zone on frigid soils bordering on mesic) was the result of high temperatures, winds, and low humidity. The advantage and disadvantage of high severity fires is the consumption of both native and invasive seed pools, which provides a 1-year window of low competition from invasive annuals, increasing the potential for successful seeding. Central Nevada. (Photo by Jeanne Chambers, USDA Forest Service.)

Fire and Perennial Forbs

The response of perennial forbs to prescribed fire in sagebrush and pinyon and juniper ecosystems is highly variable (Bates et al. 2017a) and closely linked to soil moisture and temperature regimes and pretreatment composition (Miller et al. 2013). Most studies reporting no postfire response for perennial forbs occurred on warm and dry (mesic/aridic) ecological sites. In contrast, increases were reported for 70 percent of prescribed fire studies on cool and moist (frigid/xeric) ecological sites. A rapid increase of perennial forb abundance typically occurs in the first few postfire years (Bates et al. 2011; Bates and Svejcar 2009; Dhaemers 2006; McCulloch 1969; Miller et al. 2014b; Roundy et al. 2014a; Stager and Klebenow 1987). Where fires were low- to moderate-severity, difference in perennial forbs between burned and control plots 3 years following treatment ranged from none in early Phase I (TDI = 0), 2.5-fold in late Phase II (TDI =.6), and more than 5-fold in Phase III (TDI = more than 0.9) (fig. 5-17; Roundy et al. 2014a). However, abundance often decreases to preburn levels 4-10 years following fire (Bates and Svejcar 2009; Bates et al. 2014). Increases in the postfire abundance of perennial forbs were often greater on sites where pretreatment perennial grass abundance was low (Everett and Ward 1984; Roundy et al. 2014a) or where high-severity fire resulted in high perennial grass mortality (Bates et al. 2011). Perennial forb abundance following fire also increased along an elevation gradient of decreasing temperatures and increasing moisture (Dhaemers 2006; Urza et al. 2017). The season of available green forb abundance was also longer in burned

compared to unburned plots (Wrobleski and Kauffman 2003), as a result of a longer season of increased soil water availability (Bates et al. 2000; Roundy et al. 2014a).

Invasive Annual Plant Response to Fire

Increases in invasive annual plant abundance is frequently greater following fire as compared to mechanical treatments (figs. 5-17, 5-26), especially on warm and dry sites (Barney and Frischknecht 1974; Dhaemers 2006; Koniak 1985; Miller et al. 2013, 2014a; Roundy et al. 2014a). The increased response of invasive plants after fire is associated with an immediate initial decrease in perennial grasses and increased available soil nutrients, especially nitrogen (Miller et al. 2013). Fire also increases potential germination of cheatgrass by as much as threefold more than cutting or shredding by increasing seedbed water and temperature conditions, which enhances cheatgrass (Cline et al. 2018; Roundy et al. 2017, 2018). Cover of invasive annual grasses and forbs often increase in the early postfire years and then can later decline to preburn levels (Barney and Frischknecht 1974; Bates et al. 2011, 2014; Dhaemers 2006; Everett and Ward 1984; Koniak 1985; Miller et al. 2014b; Roundy et al. 2014a). However, significant increases that persist for 25 years or more have also been reported (Bates et al. 2014; Chambers et al. 2017; Urza et al. 2017). Persistence and dominance of invasive annuals is often a function of several factors, including depleted perennial plant cover at the time of the fire, fire severity (which when high often results in high perennial plant mortality), postfire management, ecological site characteristics, and pretreatment tree dominance (Miller et al. 2013; Roundy et al. 2014a). Roundy et al. (2018) found that the negative relationship between perennial herb (mainly perennial grass) and cheatgrass cover varied from weak to strong among 17 sagebrush sites on untreated, burned, and mechanically

Figure 5-26—Fire is a useful tool, but the threat of invasives following a burn is of major concern. Increases in invasive annuals following a fire is often greater than following cutting and shredding treatments. Site characteristics—including moisture and temperature regimes—and surface vegetation present on the location prior to treatment are key components that influence resistance to invasive plants in addition to fire severity. Egan Range, Nevada. (Photo by Rick Miller, Oregon State University.)

treated plots. Differences in seasonal soil temperature and available water explained 71 percent of the variation in perennial grass cover and 69 percent of the differences in cheatgrass cover among sites 6 years after prescribed fire. Drier falls and wetter winters and springs favored perennial herb cover while wetter and warmer falls and warmer late springs favored cheatgrass cover.

Pretreatment cover of conifers and perennial bunchgrasses is also closely related to the postfire response and persistence of cheatgrass, especially on sites with low resistance (Bates et al. 2014; Roundy et al. 2014a; Williams et al. 2017). Six years after fire, cheatgrass cover was still higher on burned than unburned and cut plots at mid- to high-pretreatment TDI (Williams et al. 2017). But cheatgrass cover on burned plots varied greatly among sites. The combination of low preburn perennial grass abundance and high fire severity in Phase III woodlands resulted in significantly less posttreatment perennial grasses and greater cheatgrass abundance in Phase III compared to Phase II woodlands (Bates et al. 2014). In Phase III woodlands, cheatgrass cover increased from a trace to nearly 40 percent 4 years after fire and remained higher than 30 percent 9 years after fire. The high-severity fire in the Phase III woodland resulted in a decline in perennial grass cover from less than 5 percent to less than 1 percent and pretreatment density from 4 to less than 1 plant per 10 square feet in the first year following fire.

In Phase II, woodlands where perennial grass density and cover were greater after fire because of lower fire mortality, cheatgrass cover remained below 10 percent and declined to less than 5 percent 9 years after fire. The large amount of variation across ecological sites in both time and space makes it difficult to pin down the abundance of perennial grasses necessary for recovery and to limit persistent increases in invasive annuals following disturbance. However, limited studies indicate that a minimum of two to three (or possibly three to four) deep-rooted perennial grasses per 10 square feet are needed for recovery of the native perennial grass layer (Bates et al. 2007a, 2014; O'Connor et al. 2013). Type of burn treatment can also influence the magnitude of cheatgrass response. In a relatively cool and moist sagebrush-mountain mahogany community, a broadcast burn in cut western juniper stands resulted in greater cheatgrass increases than in pile-and-burn treatments (O'Connor et al. 2013). However, cover of cheatgrass was less than 6 percent across all treatments.

Postfire Tree Regeneration

The rate of postfire tree recruitment depends on tree survival, residual and outside seed sources, ecological site characteristics, and postfire management. Immediately following fire, there can be a lag in tree establishment due to the loss of shrubs, which act as nurse plants (Burkhardt and Tisdale 1976; Chambers 2001; Erdman 1970; Wangler and Minnich 1996). Shrubs provide desirable microsites for tree establishment (Burkhardt and Tisdale 1976; Everett and Ward 1984; Meagher 1943; Miller and Rose 1995; Phillips 1909), especially for drought susceptible pinyon seedlings (Bristow et al. 2014). Shrubs also provide perching sites for avian seed dispersers (Chavez-Ramirez and Slack 1994). Recruitment of singleleaf pinyon pine and Sierra and California junipers along the east slopes of the Sierra Nevada in southern California was very slow in the first 25 years postfire but accelerated in the following 15 years (Wangler and Minnich 1996). Others have reported that during the first 50 years, postfire recruitment was greater for Utah juniper than singleleaf pinyon in Nevada (Tausch and West 1988).

But after 60 years, singleleaf pinyon recruitment exceeded that of Utah juniper. In southeast Oregon and southwest Idaho, development of western juniper woodlands from initial tree establishment to tree dominance (Phase I to Phase III) varied from 60 to 80 years on cool and moist ecological sites to more than 125 years on warm and dry ecological sites (fig. 5-15; Johnson and Miller 2006). In Utah and northeast California,

pinyon and juniper woodlands were well-developed 80–90 years after fire (Barney and Frischknecht 1974; Miller and Heyerdahl 1974). Others reported slower rates of postfire tree establishment. In central Nevada, tree cover was less than 1 percent in burned areas compared to 23.5 percent in adjacent chained areas 40 years after treatment (Bristow et al. 2014). In some cases, woodlands remained relatively open (11 percent cover) 115 years following fire (Stager and Klebenow 1987), possibly resulting from soil conditions or competition from understory vegetation. Once established, tree canopy growth of individual western and Utah juniper trees is relatively slow for the first 45–50 years, and then it increases during the next 50–70 years unless impacted by neighboring trees (Barney and Frischknecht 1974; Miller and Tausch 2001; Miller et al. 2005; Tausch and West 1988). Required retreatment cycles will vary with both site productivity and initial tree establishment. Once the area has an adequate density of tree seedlings to stock the area, woodland succession transitioning from Phase I to Phase II will take between 20–40 years and 60–120 years to transition from Phase II to Phase III (see fig. 2-2).

Response to Fire: Bare Ground and Litter

The proportion of bare ground increases immediately following fire but often decreases to near preburn levels 2–4 years after fire (fig. 5-24; Barney and Frischknecht 1974; Bates et al. 2009, 2011; Miller et al. 2014b). But with increased fire severity it can take longer for the percent cover of bare ground to return to preburn levels (Bates et al. 2014; O'Connor et al. 2013). Increased bare ground in the first postfire year occurs with the reduction of cover in vegetation, biological soil crusts, and litter. In Wyoming sagebrush and mountain sagebrush communities that have pinyon and juniper, low- to moderate-severity fires doubled the amount of bare ground from 30 percent prefire to 60 percent postfire in the first year (Miller et al. 2014b). Bare ground returned to near preburn levels by the second year after fire. In mountain big sagebrush communities, cover of bare ground in the first year increased from 20 percent to 40 percent in Phase II and 30 percent to 80 percent in Phase III woodlands following high-severity fires (Bates et al. 2011, 2014; O'Connor et al. 2013). Bare ground remained higher on burned than unburned plots in the third postfire year but declined to preburn levels within 6 years of the fire (Bates et al. 2014).

Litter cover is significantly reduced immediately following fire but typically returns to preburn levels within 2–3 years (Bates et al. 2011; Miller et al. 2014b). With more severe fires, however, litter recovery takes longer, often remaining below preburn levels for 4–9 years after fire (Bates et al. 2014; O'Connor et al. 2013). Little research has addressed the rate of recovery of litter biomass and depths following fire (Miller et al. 2013). Both litter biomass, depth, and cover influence ecosystem functions including surface soil temperatures and moisture (Young et al. 2013b). When litter abundance is sufficient to both increase and reduce soil temperatures and increase soil moisture, it can enhance seed germination and seedling establishment of desirable and nondesirable plants (Chambers 2000; Evans and Young 1970; Young et al. 2013a).

Fire and Seeding

During the early years of prescribed burning in pinyon and juniper, the goals of seeding were primarily about increased forage production and soil stabilization, and therefore introduced species were used. Schroeder (1961) reported that seeding grasses (crested and western wheatgrass, and weeping lovegrass) on burned sites for which grazing had been deferred for 3 years resulted in substantial increases in forage production reaching 1,660 pounds per acre. On the adjacent unburned area, forage production was only 60 pounds per acre. Between 1955 and 1963 on the Hualapai Indian Reservation in northwestern Arizona, forage production was increased by an average of 500 pounds per acre with burning and seeding treatments (McCulloch 1969). As

management goals began to consider wildlife (such as mule deer), rangeland revegetation trials began to seed native plants (Monsen et al. 2004; Plummer et al. 1968; Roundy 1996). Over time, emergency burned area rehabilitation plans evolved to emphasize the restoration of diverse, native plant communities. But success for seeding with native species has often been low, which allows cheatgrass and other introduced species to become established and dominant a site quickly. This at least partially due to the use of seed from inappropriate sources—and the rate of success is closely tied to soil moisture and temperature regimes.

Postfire seeding can be effective when treating pinyon and juniper stands where advanced infilling has left few residual perennial plants (Bybee et al. 2016). Successful postfire revegetation in these types of stands limits dominance by invasive annuals (Floyd et al. 2006; Redmond et al. 2014; Sheley and Bates 2008; Thompson et al. 2006). Seeding success after wildfire or prescribed fire in pinyon and juniper areas is highly dependent on species selection and seed source, seed placement, and posttreatment precipitation (Hardegree et al. 2016; Knutson et al. 2014; Monsen et al. 2004; Pyke et al. 2017; Roundy 1996; Roundy and Call 1988). Because pinyon and juniper expansion into sagebrush generally occurs in relatively higher precipitation zones compared to low elevation warm and dry big sagebrush sites, their potential for revegetation success is greater.

However, seeding success in woodlands bordering on occupying warm and dry sites can be expected to be lower. Postfire broadcast seeding is generally more successful when seeds are covered— drilling seed is usually more successful than broadcast seeding because it can place seed at the correct depth. However, postfire chaining is one method used to cover seeds following broadcasting (Juran et al. 2008; Monsen et al. 2004; Ott et al. 2003). Chaining may also help seedling establishment by breaking up water repellent soil layers, which can occur around the base of the tree (Madsen et al. 2015). Subsurface water repellency can prevent seedling roots from reaching deeper soil moisture and lead to seedling mortality (Madsen et al. 2012a,b). Although postfire chaining may initially reduce residual plant cover, it can improve establishment of broadcast seed sufficiently to prevent dominance by invasive annuals (Ott et al. 2003). One disadvantage of chaining is the reduction of biotic soil crusts, which can limit invasive plant dominance on some soils, especially in the Colorado Plateau (Evangelista et al. 2004). Broadcasting seed on snow has some benefits and has been observed to successfully establish seedlings (Meyer 1994). However, winter broadcasting (on snow, for two out of three sites) was less successful than fall broadcasting for Wyoming big sagebrush (Ott et al. 2017).

In the past, postfire seed mixes included primarily nonnative species, but recently mixes have included more or even exclusively native species. Some managers and researchers are concerned that introduced species in seed mixes often outcompete native species (Gunnell et al. 2010; Knutson et al. 2014; Vernon et al. 2001). But another, and perhaps more pressing concern, is taking advantage of the one opportunity to quickly establish a perennial herbaceous layer and prevent the establishment and dominance of invasive annuals. Planting sagebrush seed postfire has been considerably less successful than seeding perennial herbaceous species (Board et al. 2011; Davies et al. 2013; 2014; Knutson et al. 2014). Again, the appropriate seed source, seed application, and soil moisture conditions following planting are important factors determining success.

Knutson et al. (2014) suggested that, based on the success reported by McAdoo et al. (2013), transplanting sagebrush plants should be considered. While large-scale transplanting is not practical, it could be useful for the establishment of seed dispersal zones. Establishment of native species can be increased when plant materials are adapted to site conditions, posttreatment precipitation is adequate, seeds are buried at proper depths, and moderate to high seeding rates are used (Davies et al. 2014; Sheley and Bates 2008; Thompson et al. 2006). Accurate weather forecasting may also improve

revegetation success by timing treatments to avoid low soil water availability (Hardegree et al. 2016, 2018). Experimentation with seed coating also shows promise in improving seeding success by protecting seeds and seedlings from frost mortality, drought, and predation (Boyd and Lemos 2013; Madsen et al. 2014; Roundy and Madsen 2016).

Biological Crusts and Fire

Fire generally results in a significant and immediate decline in the cover of biological soil crusts (Miller et al. 2013, 2014b; Warren et al. 2015). It also has a significantly greater impact on soil crusts than tree cutting (Miller et al. 2013) but has not been compared to treatments with greater soil surface disturbance such as chaining. In central Utah, cover of biological crusts in juniper woodlands were often more abundant in tree and shrub interspaces than beneath the canopies (Warren et al. 2015). Following a prescribed fire, bryophyte cover in the interspace declined from 13 percent to 3 percent and lichens from 6 percent to 4 percent. Beneath tree and sagebrush canopies, bryophytes and lichens were uncommon and thus little affected by fire. However, the loss of tortula moss beneath the tree canopies was nearly complete after the fire. Tortula moss commonly occurs beneath shrub and tree canopies where fire severity is typically high (Warren et al. 2015). On two cool and moist ecological sites, prescribed fire reduced biological soil crust cover from 6 percent to 2 percent and from 3 percent to just a trace (Miller et al. 2013; O'Connor et al. 2013). In several warm and dry sites, crust cover was reduced from 28 percent to 8 percent. In the woodlands on warm and dry sites, biological soil crusts were reduced from 3 percent to trace in Phase II and 1.5 percent to trace in Phase III (Bates et al. 2014). There was little recovery of biological soil crusts 9 years after a fire on a Wyoming big sagebrush site bordering warm/cool and dry/moist conditions (Bates et al. 2014). Little work has been done comparing the impacts of mechanical treatments on biological crusts. However, we would expect the impact to vary with the amount of surface area affected and the severity of the mechanical footprint.

CONCLUSION: WHAT WE HAVE LEARNED

The pinyon and juniper symposium held in Logan, Utah, in 1975 was an attempt to summarize 30 years of information regarding the management of these semiarid woodlands since World War II. The primary goals of vegetation management were to increase forage production and improve big-game winter habitat. Restoration of watersheds also became an important goal as a result of extensive flooding around the 1900s along the lower slopes of the Wasatch Range that caused severe cutting of stream bottoms and damage to towns. This was a result of severely overgrazed uplands. The primary method of tree removal post-World War II to the early 1970s was chaining and seeding often with introduced grasses. Conclusions in this symposium were mixed, with concerns about possible woodland mismanagement (Dwyer 1975) as observations and studies of tree removal found that understory responses to tree removal were not always positive.

Forty-eight years later, our knowledge of pinyon and juniper woodlands has significantly increased, with over 2,000 papers on the subject. Most of the soils and ecological site types across the Great Basin and Colorado Plateau have been described and mapped (or are in the process of being mapped), and we have many powerful tools including various types of imagery, computer analyses, and high-tech equipment for measuring plant physiological processes, soil moisture, microclimate, and other ecological factors. We also have become aware of the differences between persistent or presettlement woodlands and those newly expanded into shrub-grassland, savanna, riparian, and aspen communities. Common methods of tree removal today include cutting, shredding, burning, or various combinations—and in some areas, chaining. Although we have a variety of tools and information available for making informed decisions, challenges in woodland management

have significantly increased over the past decades. Temperatures have been warming over the past 100 and more years, big fires have become larger and more frequent in the past 20 years, and invasive plants are an ever-increasing problem—especially the annual grasses. Our success in seeding native plants is still inconsistent and expensive. And, funding for restoration always seems to be limiting.

What is the best form of management of these woodlands—both persistent and expanding into adjacent nonwoodland communities (fig. 5-27)? Our overall goal has changed, broadening to the management and restoration of functional ecosystems that are resilient and resistant to invasive species and resource-conserving. And we do this with careful consideration of the area's ecological characteristics and history (fig. 5-2), types of vegetation management, and methods selected—including to seed or not to seed (table 5-3)—and posttreatment management. We know that as sites become warmer and drier they become more difficult to restore. We also know that plant composition prior to treatment is very important. We recognize that with growing concerns over invasive species, the use of prescribed fire must be carefully considered and generally used in ecological sites with high resilience and resistance to invasive species. In addition to ecological and management concerns, funding is always a limiting factor in what realistically can be done. Looking for creative ways to develop resources for dollars from reduction of fuels, habitation improvement, and wood products are important components to restoring functional ecosystems.

Figure 5-27—Today we have a variety of methods for tree removal including mastication (foreground), chaining and seeding (mid-picture), cutting, and burning. All methods need to be carefully considered and should fit with ecological site characteristics, including the option of doing nothing. Eastern Nevada showing a recent mastication in the foreground and a 5-year-old chained and seeded in the background. (Photo by Rick Miller, Oregon State University.)

GLOSSARY: TERMS AND DEFINITIONS

Organized by topic: Community Ecology and Soils, Ecohydrology, Fire, and Time Periods.

COMMUNITY ECOLOGY AND SOILS

Alfisols—mineral soils that have umbric or ochric epipedons, argillic horizons, and that hold water at more than -1.5 MPa tension during at least 90 days when the soil is warm enough for plants to grow outdoors. Alfisols have a mean annual soil temperature of 8 °C or a base saturation in the lower part of the argillic horizon of 35 percent or more when measured at pH 8.2.

Alluvium—material, such as sand, silt, or clay, deposited on land by water.

Alpine—treeless communities growing above timberline; tundra—treeless communities with permanently frozen subsoil and supporting low-growing vegetation such as lichens, mosses, and shrubs.

Argillic—alluvial horizon of clay accumulation.

Aridic—see soil moisture regime.

Aridisols—mineral soils that have an aridic moisture regime, and ochric epipedon, and other pedogenic horizons but no oxic horizon.

At-risk phase—a community phase that is most vulnerable to transition to an alternative state (i.e., least resilient). See definition of phase below.

Atlantic Multidecadal Oscillation (AMO)—is a climate cycle that affects the sea surface temperature (SST) of the North Atlantic Ocean based upon the average anomalies of sea surface temperatures (SST) in the North Atlantic basin.

Attribute—a characteristic or property inherent in or ascribed to something (e.g., a fire event, current vegetation, site).

Available water holding capacity—the maximum amount of water available (percent of volume) for plants that a soil can store, which varies with such soil characteristics including texture and organic matter content.

Biological soil crusts—also known as cryptogamic, cryptobiotic, microbiotic, or microphytic soil crusts; composition of soil biotic crusts is a complex of cyanobacteria, green algae, lichens, mosses, microfungi, and other bacteria. They grow in the open interspaces of shrubs and herbs and typically increase along an environmental gradient of increasing aridity.

Broadcast seeding—scattering seed on the soil surface across an area by aerial or ground application. Success often increased if followed up by ground disturbance to help cover the seed.

Cache—seed cache is storage of seed by a hoarding animal such as rodents or birds.

Climate—general prevailing weather conditions for an area or region over a long period of time. As compared to weather, which refers to current atmospheric conditions (or over specific time period of interest; example is a 10-year period following fire).

Climax—self-perpetuating community controlled by climate and site (land) that culminates succession (Clements 1936); when vegetation is in equilibrium with all environmental factors.

Crossdating—a method used to date stumps, logs, etc., by matching patterns of wide and narrow tree rings between live and dead trees.

Cabling—a vegetation management method of pulling a long piece of steel cable across the landscape to pull up woody plants.

Comstock period—the discovery of silver in the Virginia Range in the late 1850s, which started an intense mining period peaking between 1884 and 1891 throughout Nevada, Utah, and the east slopes of the Sierras in California.

Concentrated flow—runoff that accumulates or converges into well-defined channels or rills often with sufficient energy to carry or move material and erode.

Cryic—see soil temperature regime.

Current Potential—similar to the reference state in which ecological function has not changed, the site has maintained the presence of all structural and functional groups, nutrient cycling etc., but resiliency is reduced due to the introduction of nonnative annual species (even trace). Also referred to as the "desired potential community."

Degraded seral-stage—characteristics include relatively low productivity compared to their historic state; depleted perennial herbaceous layer, increased bare ground, and invasive species are a co-dominant or dominant component of the vegetation. These communities are either an at-risk phase or have shifted to an alternate state.

Disclimax—human (Eurasian) caused disturbance(s) that degrade communities (e.g., grazing, alterations to natural fire regimes, land development, recreation, and logging)

Disturbance—any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment (White and Pickett 1985, p 7).

Dioecious–male and female reproductive parts are on separate plants; contrast with monecious where male and female parts are on the same plant.

Diplochory—a seed dispersal mechanism in which a plant's seed is moved sequentially by more than one dispersal mechanism or vector.

Duripan—a subsurface horizon that is cemented by alluvial (water transported) silica to the degree that fragments from the air-dry horizon do not slake (take in water or crumble) during prolonged soaking.

Ecological Province—a subdivision of a region having a distinctive combination of geographical features and ecological sites.

Ecological Site—a kind of land with specific physical characteristics that differs from other kinds of land in its ability to produce distinctive kinds and amounts of vegetation and its response to management. Synonymous with ecological type used by USDA Forest Service; and Rangeland Ecological Site. Key components of an ecological site are climate, soils, and topography.

Ecoregion—denote areas of general similarity in ecosystems and in the type, quality, and quantity of environmental resources; they are designed to serve as a spatial framework of the research, assessment, management, and monitoring of ecosystems and ecosystem components. Defined at various levels; level III is similar but not identical.

Endozoochorous dispersal—dispersal of seeds via ingestion by animals.

El Niño-Southern Oscillation (ENSO)—periodic fluctuation in sea surface temperatures and air pressure influencing climate; warming and high air surface pressure phase is El Niño usually reaching full strength toward the end of the year; cooling and low air surface pressure phase La Niña.

Entisols—soils that have no distinct subsurface diagnostic horizons within 1 m of the soil surface.

Expansion—establishment of trees in places that were formerly nonwoodland.

Facultative—having the capacity to regenerate with more than one strategy (e.g., seed and vegetatively) as opposed to a seed obligate (regenerating from seed only).

Frigid—see soil temperature regime.

Fruit abscission—the dropping of fruit or seed before it is developed, usually resulting from some form of stress.

Function—purpose, action, or activity for which a thing is specially fitted or used for (e.g., function of plant cover—protect soil from raindrop impacts, provide nesting or hiding habitat).

Habitat type—see plant association.

Herb—or herbaceous, refers to the nonwoody vascular plant component, which includes both perennial and annual grasses and forbs; aboveground stems usually die back at the end of the growing season.

High pressure systems—an area where atmospheric pressure is greater than that of the surrounding area, usually associated with clear skies and calm weather.

Historical range of variability—variability in disturbances, stressors, and ecosystem attributes in the centuries immediately preceding Euro-American occupation of the region or areas that are used to provide management targets and that can maintain biological diversity and ecosystem services (Jackson 2006).

Illuvial—soil layer or horizon (e.g., argillic) in which material carried from an overlying layer has been precipitated from solution or deposited from suspension. The layer of accumulation.

Inceptisols—soils that have one or more pedogenic horizons in which mineral materials other than carbonates or amorphous silica have been altered or removed but not accumulated to a significant degree. Water is available to plants more than half the year or more than 90 consecutive days during a warm season.

Infill—increasing tree density within persistent or postsettlement woodlands that were previously of lower density.

Intercanopy—the open spaces in a woodland that occur between tree canopies.

Landscape—an expanse of land characterized by such attributes as topography, geology, soils, microclimate, and the spatial arrangement of vegetation patches, which change in both time and space giving each landscape a unique pattern. Landscapes are composed of a multitude of ecosystems at varying scales that are all connected. "A landscape is a mosaic of patches of different spatial arrangement giving each landscape a unique pattern" (Urban et al. 1987). A mosaic of land forms, vegetation, and land uses; varying in extent with the organism(s) of interest, objectives, or purpose.

Larder-hoard—the result of hoarding or caching food (seed) by animals.

Latitudinal gradient—a gradient of change that occurs predominately north and south, such as the weakening of monsoonal moisture moving from south to north.

Lithic—shallow soils over a paralithic (soft bedrock) contact or duripan (subsurface horizon cemented by bedrock).

Low pressure systems—an area where the atmospheric pressure is lower than that of the area surrounding it. Lows are usually associated with high winds, warm air, and atmospheric lifting and often associated with bad weather.

Major Land Resource Areas/MLRAs—geographically associated land resource units, usually encompassing several thousand acres. They are characterized by particular patterns of soils, geology, climate, water resources, and land use. A unit may be one continuous area or several separate nearby areas.

Mesic—see soil moisture temperature regime.

Midden—old dump or waste pile from humans and animals. Archeologists use middens found in old human-inhabited caves or in ancient dwellings to determine the types of plants and animals available during the dated time period. Especially useful are packrat middens, which contain plant macrofossils collected within 150 feet of the cave. Radiocarbon dating have dated macrofossils in some of these dung piles to be well over 10,000 years old.

Mollic Epipedon—thick, dark, soft, surface layer greater than 10 inches thick; high base saturation more than 50 percent and containing at least 1 percent organic matter.

Mollisols—mineral soils that have a mollic epipedon overlying mineral material. They can have argillic, natric, albic, cambic, gypsic, calcic, or petrocalcic horizon, a histic epipedon, or a duripan.

Monoecious—see dioecous.

Mountain shrub—higher elevation sagebrush sites that often include snowberry, serviceberry, currants, bitterbrush, and other shrub species typically growing in precipitation zones more than or equal to 14 inches.

Obligate—essential or necessary.

Old growth—see persistent and presettlement woodlands; encompasses the later stages of woodland development that typically differ from earlier stages in a variety of characteristics, which may include tree size, accumulations of large dead woody material, number of canopy layers, species compositions and ecosystem function (USDA Forest Service 1993). In pinyon and juniper woodland, old-growth characteristics include the majority of the overstory tree canopy being composed of old trees (more than 250 years) and the presence of large dead wood including snags, logs, and weathered stumps (Miller et al. 2014). Characteristics of individual old-growth pinyon and juniper trees compared to younger mature trees (more than 150 years) are differences in tree canopy shape, distinct bark characteristics, and branch structure, which may include large basal branches, and dead branches often covered with lichen (fig. 25a,b). In persistent pinyon and juniper woodlands, disturbance intervals between stand-replacement events are usually measured in centuries, allowing stands to attain old-growth characteristics including trees that are many centuries old.

[18]O—ratio of $^{18}O/^{16}O$ (ppt from the Pacific is isotopically far lighter, less ^{18}O than ppt derived from monsoon ppt; Grayson 2011, pg. 239). $^{18}O/^{16}O$ indication of water temperature. Since ^{16}O evaporates more rapidly from seawater than ^{18}O, ocean surfaces in the tropics and subtropics contain higher ratios of ^{18}O than colder Pacific waters. However, when water vapor condenses, the heavier water molecules ^{18}O tend to condense and precipitate first. Thus $^{18}O/^{16}O$ decreases farther north (greater distance from the source). Changes in climate that alter global patterns of evaporation and ppt therefore change the ratio.

Pacific Decadal Oscillation (PDO)—similar to ENSO in character but varies over 20 to 30 years, while ENSO cycles typically only last 6 to 18 months.

Paralithic—weathered layer of bedrock that becomes soft or crumbly.

Patch—a homogeneous unit of area occurring on a landscape; frequently patches are defined as distinct plant communities.

Pedogenic (fusion)—a surface layer of parent rock or sediment feeds a physicochemical and other imprints downward into a subjacent buried soil formed in a separate parent rock or sediment, leading to pedogenic fusion of the two soils.

Period—a large interval of time that has a specific characteristic and is used to categorize history, cosmology, and climate. However, in geology, period is the basic unit in which a single type of rock system is formed. Two or more periods comprise a geological Era (e.g., Quaternary = period; Quaternary + Neogene + Paleogene = Era).

Persistent pinyon-juniper woodlands—are found where site conditions (soils and climate) and disturbance regimes are inherently favorable for pinyon and/or juniper, and where trees are a major component of the vegetation unless recently disturbed by fire, clearing, or other severe disturbance. Canopy structure varies considerably, from sparse stands of scattered small trees growing on poor substrates to relatively dense stands of large trees on relatively productive sites. Either pinyon or juniper may dominate the canopy, or the two may co-dominate. The understory may be dominated by shrubs or forbs or less commonly by graminoids; a consistent feature of the understory is low total plant cover with frequent patches of bare soil or rock. Notably, these woodlands do *not* represent 20th century conversion of formerly non-woodland vegetation types to woodland but are places where trees have been an important stand component for at least the past several hundred years. Closely synonymous with potential vegetation.

Phase (community)—a plant community within a state that is hypothesized to replace other communities along traditional succession-retrogression pathways; succession from one community to the next is readily reversible over short time periods (years to decades) without management intervention because they are not separated by thresholds. However, an at-risk phase (community) may not progress directly to the most resilient community phase without passing through an intermediate phase; it may be discontinuous from the most resilient phase.

Phase I, II, III—Phase I trees are present but shrubs and herbs are the dominant vegetation influencing ecological processes on the site; Phase II trees are codominant with shrubs and herbs and all three vegetation layers influence ecological processes; Phase III trees are the dominant vegetation on the site and the primary plant layer influencing ecological processes on the site (from Miller et al. 2005). Phases can be quantified using total perennial cover of primary functional groups to calculate a total tree dominance index (TDI) (from Williams et al. 2017).

> *Phase I*—total tree / total tree + shrub + perennial grass = less than 0.34 (tree cover less than one-third)

> *Phase II*—total tree / total tree + shrub + perennial grass = 0.34–0.67 (tree cover one-third to two-thirds)

> *Phase III*—total tree / total tree + shrub + perennial grass = more than 0.66 (tree cover more than two-thirds)

Pinyon and juniper savannas—typically defined as areas of sparse tree cover (less than 10 percent tree cover) and an understory of perennial grasses.

Plant association—or "habitat type"; a reoccurring plant community with a definite floristic composition, specific diagnostic species, uniform habitat conditions, and uniform physiognomy; typically designated by a diagnostic overstory and understory species (e.g., ponderosa pine/Idaho fescue, mountain big sagebrush/bluebunch wheatgrass). *"Relatively stable, fully developed (mature) communities"* (Tansley, A.G. 1920).

Plant community—a relatively homogeneous assemblage of species growing at a particular point in time and space (synonymous with phase); often defined by the dominant species. *"Uniformity, area, boundary, and duration are the essential components of a plant community"* (Gleason H.A. 1939).

Postsettlement—see presettlement.

Presettlement woodlands—woodlands and trees that occupied the landscape prior to settlement and thus in the absence of Eurasian disturbances. The decade of the 1850s is most frequently used to separate presettlement and postsettlement, although livestock were introduced into portions of the Southwest prior to 1850. These woodlands often exhibit old-growth characteristics unless recently disturbed.

PZ—precipitation zone; used by the NRCS when mapping soils and ecological sites.

Rainsplash erosion—when rain falls at sufficient intensity, individual raindrops can detach and move soil particles a short distance.

Reference state—historic or potential plant community including seral (successional) stages; based on conditions believed to be present before widespread alterations by Euro-Americans. (see current potential).

Reference site—a location (stand or watershed) that demonstrates a reference state.

Resilience—the capacity of an ecosystem to regain its fundamental structure, processes, and functioning when subjected to stressors or disturbances like drought, livestock grazing, or wildfire. In this context, resilience is a function of the underlying ecosystem attributes and processes that determine ecosystem recovery, rather than the amount or magnitude of stress or disturbance that an ecosystem can withstand before changes in attributes and processes result in new alternative states. In summary—the ability to recover.

Resistance—the capacity of an ecosystem to retain its fundamental structure, processes and functioning (or remain largely unchanged) despite stressors or disturbances.

Resistance to invasives—the abiotic and biotic attributes and ecological processes of an ecosystem that limit the population growth of an invading species.

Savanna—pinyon and juniper savannas are found where local soils, climate, and fire regime were suitable for both trees and grasses, resulting in a low tree density.

Scatter-hoard—hiding food in many different places within a territory of a scatter-hoarder, typical birds or small mammals.

Sea Surface Temperatures (SST)—surface temperatures in various geographic portions of the oceans that influence climate.

Semiarid climate—regions that receive precipitation below potential evapotranspiration but not extremely; intermediate between arid and humid often supporting shrubs and grasses as compared to arid that supports scrubby vegetation.

Seral (sere)—a plant community that occurs during ecological succession. Seral stages for a site range from early seral following a stand-preplacing disturbance to late seral, which represents a relatively stable community in the absence of a stand-replacing disturbance event.

Shallow soil—see soil depth.

Sheet erosion—detachment of soil particles by raindrop impact and their removal downslope by water flowing overland as a sheet instead of in definite channels or rills.

Shredding—another term used for mastication, a method of using heavy machinery with grinder blades that chips trees or mulches woody plants.

Skeletal soils—contain 35 percent or more (by volume) of rock fragments, cobbles, and gravel having diameters greater than 0.08 inches (2 mm) within shallow depths (less than 20 inches).

Soil depth—very shallow = less than 10 inches; shallow = 10 to 20 inches; moderately deep = 20 to 30 inches; deep = greater than 30 inches.

Soil moisture regime—considered an important soil property in combination with soil temperature that influences plant growth and biological soil processes. The regime is based on the amount of soil moisture available during the growing season in areas with moist-cool winters and hot-dry summers. Although mapped at distinct breaks in precipitation (less than 12 inches or more than 12 inches), soil moisture regimes in reality are continuous gradients changing with location and elevation. It is important to consider where the site fits in along the gradient; for example, a site with an aridic moisture regime that receives 11.5 inches of precipitation will often be more resilient to disturbance than an aridic site receiving 9 inches of precipitation. For a detailed definition and description for each soil regime, see USDA NRCS (1999).

> *Aridic* (dry; mapped at less than 12 inches)—dry in all parts of the profile more than half the time when soils are more than 41 °F at 20 inches. In the Great Basin and Columbia Basin, aridic soils are usually mapped in precipitation zones receiving less than 12 inches. However, soils (e.g., lithic or shallow) with limited storage capacity may be mapped as aridic even though the site may receive more than 12 inches of precipitation. Indicator plants are desert shrub, black sagebrush, and Wyoming big sagebrush.

> *Xeric* (moist; mapped at more than 12 inches)—soils are moist in some part of the profile more than 50 percent of the time that soils are more than 41 °F, or in 6 or more years out of 10 is moist in some part for at least 90 consecutive days when soil temperatures are continuously higher than 47 °F. In the Great Basin and Columbia Basin, aridic soils are usually mapped in precipitation zones receiving more than 12 inches. Indicator plants are mountain big sagebrush, low sagebrush, pinyon pine, and Utah or western juniper. Snowberry and serviceberry also often occur as moisture increases above 12 inches and/or near the boundary of frigid and cryic soils.

> *Ustic* (summer)—moisture regime is intermediate between the aridic and udic regimes (udic soils are of humid climates with well-distributed rainfall that usually is not limiting during the growing season). Its concept is one of moisture that is limited but is present at a time when conditions are suitable for plant growth (summer). The concept of the ustic moisture regime is not applied to soils that have cryic or pergelic soil temperature regimes (defined below). This moisture regime is usually associated with warm season (C4) grasses. Ustic—

moisture regime is limited but present when conditions are suitable for plant growth in the summer. Udic—moisture is not limiting for plant growth.

Soil temperature regime—considered an important property of a soil along with soil moisture that influences plant growth and biological soil processes. Usually measured at the 20 inch depth (0.5 m) (or depth at the lithic or paralithic contact), which is considered deep enough to reflect seasonal temperatures and not daily cycles. Since measurements of seasonal soil temperatures are spatially limited across the Great Basin, soil temperature regimes are estimated based on seasonal air temperatures, which are largely influenced by location, elevation, and aspect. When soils are mapped, temperature regimes are most commonly based on elevation and aspect, which are adjusted for each sub-region (MLRA). For a detailed definition and description for each soil regime, see USDA NRCS (1999).

Mesic (warm)—mean annual soil temperature is more than or equal to 47 °F but less than 59 °F and the difference between mean summer and winter soil temperature is more than 9 °F at 20 inches soil depth. Elevation break between mesic and frigid varies with MLRA; (e.g., Malheur High Plateau 4,000 feet on the flat, ± 500 feet adjustment for north or south aspect); Central Nevada Basin and Range and the Great Salt Lake Basin MLRAs mesic soils are mapped up to 5,500 and 6,500 ft. Wyoming big sagebrush, black sagebrush, and desert shrub most frequently occupy mesic soils. Mesic soils have a low relative resistance to invasive plants compared to frigid and cryic soils. They are also considered to have lower resilience.

Frigid (cool)—mean annual soil temperature is less than 47 °F and the difference between mean summer and winter soil temperature is more than 9 °F at 20 inches soil depth. Usually mapped above 4,000 feet (± 500 feet depending on south or north aspect) in the Malheur High Plateau and between 6,000–8,500 feet Central Nevada Basin and Range and the Great Salt Lake Basin MLRAs. Indicator species are mountain big sagebrush, pinyon pine and low sagebrush (shallow soil), although black sagebrush and occasionally Wyoming big sagebrush may occur on the warmer end of this soil regime or where soil moisture is limiting. Resistance to invasive species and resilience are higher than on mesic soils.

Cryic—mean annual soil temperature is more than 32 °F but less than 47 °F and is cooler in the summer than frigid soils. Usually mapped above 6,000 feet (± 500 feet depending on south or north aspect) in the Malheur High Plateau and above 8,200 feet Central Nevada Basin and Range and the Great Salt Lake Basin MLRAs. Indicator species are curlleaf mountain mahogany, white and grand fir, limber pine, lodgepole pine, and white bark pine, which typically intermingles with mountain brush, mountain big sagebrush, and low sagebrush. Resistance to invasive species is higher than mesic and frigid (although limited data to support) soils. Resilience is also high on the warm end of this regime, but declines as colder temperatures limits the length of the growing season and soil development.

Special habitats—native biological communities or ecosystems that are rare, unique, or highly productive elements of regional landscapes (Salwasser 1990).

Successional trajectories—a sequence of successional stages that can be initiated or changed by disturbance, including drought, fire, herbivory, etc.

Stand—a homogenous group of plants growing together on a contiguous area forming a subset of a plant community. Stands are the unit area measured to estimate current vegetation composition and structure for a designated plant community or ecological site.

State—a suite of plant community successional phases occurring on similar soils that interact with the environment to produce resistant functional and structural attributes with a characteristic range of variability maintained through autogenic repair mechanisms.

Subalpine woodlands—in the paleobotany literature for the Great Basin this is usually referring to woodlands composed of bristlecone, limber or white bark pine, and occasionally Engelmann spruce. (Vegetation growing on mountains just below the limit of tree growth (timberline), and above the foothill or montane zone.)

Subclimax—where natural disturbance (e.g., fire) maintains vegetation different from the climatic climax conditions.

Substrate—often refers to the upper soil horizon where plants and other organisms grow and obtain resources.

Succession—a continuous process of change in vegetation which can be separated into series or phases (Tansley 1935). A predictable, directional, and stepwise progression of plant assemblages that culminates with climax (Clements 1936).

Temporal woodland complexity—fluctuations in woodland expansion, contraction, and infill, in addition to changes in structure and composition over time.

Threshold–boundaries in space and time between alternative stable states. Thresholds are crossed when an ecosystem does not return to the original state via self-organizing processes after stress or disturbance and transitions to a new alternative state that is adjusted to the altered attributes or processes (Stringham et al. 2003; Beisner et al. 2003).

Tree dominance index (TDI)—is a quantitative measure of the relative dominance of conifers based on canopy cover to shrub and perennial grass (or herb) cover and is calculated—tree cover/[tree + shrub + tall perennial grass cover]; e.g. TDI = 0.67 occurs when tree canopy is twice that of perennial grasses and shrubs (Roundy et al. 2014a).

Triggers—biotic or abiotic variables or events, acting independently or in combination, that initiate threshold-related processes by contributing to the immediate loss of ecosystem resilience.

Tundra—treeless communities with permanently frozen subsoil and supporting low-growing vegetation such as lichens, mosses, and shrubs. Finnish word tunturia meaning treeless plain.

> *Arctic tundra* occurs in the northern latitudes; low shrubs, sedges, mosses, liverworts, grasses, sedges, low herbs.

> *Alpine tundra* is found at high elevations at more southerly latitudes—grasses and sedges, dwarf trees, small-leafed shrubs, health, and low herbs.

Ustic—see soil moisture regime

Vapor pressure deficit—the difference between the actual water vapor pressure and the saturation water vapor pressure at a particular temperature.

Varve—a series of layers of sediment deposited in a body of still water in one year.

Vegetation Zones—much of the paleobotany literature refers to the vegetation zones listed below, which include characteristic plant species (Thompson 1990).

> ***Shadscale:*** greasewood, shadscale, budsage.
>
> ***Lower sagebrush or shrubland:*** big sagebrush probably Wyoming big sagebrush and basin big sagebrush and in some areas possibly black sagebrush.
>
> ***Woodland:*** PIMO, JUOC, JUOS, big sagebrush (probably a mix of ARTRt and ARTRw depending on whether in the upper or low elevation), serviceberry, curlleaf and little-leaf mountain mahogany, mormon tea (*Ephedra viridis*).
>
> ***Upper sagebrush steppe or shrubland:*** big sagebrush (probably mountain big sagebrush, possibly mixed with low sagebrush), silvery lupine (*Lupinus argenteus*), snowberry, occasional serviceberry, green rabbitbrush, curlleaf mountain mahogany; at the northern edge of the Great Basin common and horizontal junipers also occurred.
>
> ***Montane Forest:*** White fir, ponderosa pine, Douglas-fir, Rocky Mountain maple, common juniper, Rocky Mountain juniper, quaking aspen, sagebrush; mainly in the south and eastern region of the Great Basin, which is lacking in the central and northwestern parts of the Great Basin, where mountain shrub (upper sagebrush-steppe or shrubland) is more common.
>
> ***Subalpine woodland:*** bristlecone pine, limber pine, whitebark pine, subalpine fir, Engelmann spruce, prostrate or common juniper often associated with sagebrush. Sagebrush species and subspecies associated with subalpine forests are mountain big sagebrush (*Artemisia tridentata* subsp. *vaseyanna*), mountain silver sagebrush (*A. cana* subsp. *viscidula*), low sagebrush (*A. arbuscula*), sticky sagebrush (*A. rothrockii*), and snowfield sagebrush (*A. spiciformis*).

Very shallow soil—see soil depth.

Weather—see climate.

Wooded shrublands—are found where local soils and climate support a shrub or shrub-steppe community, but trees can increase during moist climatic conditions and periods without disturbance and decrease with droughts and following disturbances (Romme et al. 2009). They can also be low sagebrush communities with sparse stands of trees, where densities are limited by soil conditions.

Woodlands—typically refer to land covered by trees to low-density forest with open habitats. Pinyon and juniper are typically referred to as woodlands and in the paleo-literature stands of bristle cone, limber pine, and whitebark pine are often called woodlands.

Xeric—see soil moisture regime.

ECOHYDROLOGY

Hydraulic conductivities—the rate at which water is redistributed through the soil profile and is a function of pore space connectivity and soil wetness. Hydraulic conductivity increases with increasing soil wetness due to greater connectivity of wet pores. Infiltration rates for wet soils approximate saturated hydraulic conductivity and are generally greater for coarse-grained or well-aggregated soils. Hydraulic conductivity is typically measured for both unsaturated and saturated conditions, reflecting wet and dry conditions.

Infiltration rates—the velocity or speed at which water enters into the soil. It is usually measured to the depth of the water layer that can enter the soil in 1 hour.

MPa, h^{-1}, d^{-1}—abbreviation for the standard unit megapascal, used to describe the pressure ranges of hydraulic systems, such as water potentials, which relate to water availability in soils for plant growth.

Rills—concentrated flow paths or microchannels that form when surface roughness elements concentrate sheet-flow into narrow, deeper flow paths, which increase the velocity and erosive energy of runoff.

Rainsplash erosion—the transfer of sediment resulting from raindrop impact.

Sheet-flow—overland flow or downslope movement of water taking the form of a thin, continuous film over relatively smooth soil or rock surfaces and not concentrated into channels larger than rills.

Soil water repellency—is a naturally occurring soil condition that impedes infiltration and typically occurs beneath shrub, pinyon, and juniper canopies. Water repellent soils form by the coating of articles with hydrophobic compounds leached from organic matter accumulations, microbial byproducts, or fungal growth under litter and duff.

Soil water zone—the uppermost portion of the soil profile where soil water is obtained and used by plants or evaporated into the atmosphere.

Splash-sheet erosion—the combined effects rainsplash and sheet-flow erosion.

Vesicular crusts—fragments of physical crusts disperse or "melt" when placed in water. A vesicular crust is a type of physical crust with many small, unconnected air pockets or spaces similar to those in a sponge.

FIRE

Crown fire—fire that burns in the crowns of trees and shrubs. Usually ignited by a surface fire. Crown fires are common in coniferous forests and chaparral-type shrublands.

Duff—partially decomposed organic matter lying beneath the litter layer and above the mineral soil typically found beneath the tree and shrub canopies.

Fire duration—the length of time that combustion occurs at a given point. Fire duration relates closely to downward heating and fire effects below the fuel surface as well as heating of tree boles above the surface.

Fire frequency—the number of times that fires occur within a defined area and time period. Also expressed as the average time interval between successive fires or the number of fires within a specific period of time.

Fire intensity—a general term relating to the heat energy released in a fire; the amount and rate of surface fuel consumption.

Fire interval—or "fire free interval" or "fire return interval"; the number of years between two successive fires documented in a designated area (i.e., the length of time between two successive fire occurrences); the size of the area must be clearly specified (units = years).

Fire occurrence—or "fire incidence"; one fire event taking place within a designated area during a designated time (no units; either yes a fire occurs, or no a fire does not occur).

Fire regime—refers to the nature of fire occurring over long periods and the prominent immediate effects of fire that generally characterize an ecosystem; a generalized description of the pattern of repeated fire on a landscape.

Fire rotation—the length of time necessary for an area, equal to the entire area of interest, to burn; area of interest must be clearly defined (units—years/area).

Fire severity—the effects of fire on ecological processes, soil, flora, and fauna; degree to which a site has been altered or disrupted by fire.

Ground fire—fire that burns in the organic material below the litter layer, mostly by smoldering combustion. Fires in duff, peat, dead moss and lichens, and punky wood are typically ground fires.

High-severity fire regimes—typically infrequent fires (more than 100-year MFRI)—high intensity, stand-replacing fires. Dominant vegetation structure, composition and function are all markedly charged by the incident.

Low-severity fire regimes—typically frequent fires (1–25 year MFRI) with few overstory effects (low mortality of dominant vegetation) due to low intensity.

Mean fire return interval (MFRI)—or "mean fire free interval" or just "mean fire interval"; arithmetic average of all fire intervals determined in a designated area during a designated time period; the size of the area and the time period must be specified (units equal years).

Moderate-severity fire regimes typically infrequent fires (25–100 MFRI); partial stand-replacing fires including burned areas ranging from low to high-severity.

Surface fire—the flaming combustion and spread is mainly fueled from dead and live fuels near the surface of the ground including litter, grasses, forbs, low shrubs, and large woody material laying on the ground (logs and stumps).

Surface fuels—surface organic materials less than 3.3 ft, including surface litter not yet decayed enough to lose their identity, grasses and forbs, shrubs and tree seedlings less than 3 feet in height, and heavier branchwood, downed logs, and stumps interspersed with or partially replacing the litter.

TIME PERIODS

Bølling-Allerød—was a relatively warm and moist interstadial period that occurred during the final stages of the last glacial period. This warm period ran from c. 14,700–12,700 years before the present. It began with the end of the cold period known as the Oldest Dryas, and ended abruptly with the onset of the Younger Dryas.

Dryas—(Oldest) Younger Dryas—followed the Bølling-Allerød period, radio carbon dating about 11–10 ka (ka = 1000 years ago) (10.8–9.6 ka, American Meteorological Society); a period of cold climatic conditions and drought (matches Eddy and Bradley graph); but others have it 12,800–11,500 years before present. Region around Greenland cooled by 9°–12.6 °F.

Eon/ Era/ Period/ Epoch/Age (in geology)

Eon—a major division of geological time, subdivided into eras, usually representing billions of years.

Era—subdivision of Eon and subdivided into Periods.

Period—represents a time of specific geologic events.

Epoch—we are currently living in the Holocene Epoch of the Quaternary Period, which also includes the Pleistocene Epoch. The Quaternary Period is a subdivision of the Cenozoic Era.

Age—subdivision of Epoch.

Glacial Maximum—when the ice sheets were at their greatest extension during the last glacial period (20,000 ±2,000 years ago) and temperatures in the Intermountain Region at their coolest. Synonymous with full glaciation.

Holocene—last 10,000 years AMS.

Early Holocene—11,500–8,000; relatively cool and wet Holocene period.

Mid Holocene—8,000–5,500 (4,500); drought.

Little Ice Age—a period (A.D. 1300 to 1850) of colder climate and increased glaciation compared to the 20th and 21st centuries.

Medieval Warm Anomaly—a period of relatively warm and dry conditions between A.D. 900 and 1300.

Period—large intervals of time that have a specific characteristic and is used to categorize history, cosmology, and climate. However, in geology, period is the basic unit in which a single type of rock system is formed.

1. Two or more periods comprise a geological Era (e.g., Quaternary = period; Quaternary + Neogene + Paleogene = Cenozoic Era).

2. Holocene is an epoch in the Quaternary Period.

3. LIA = age (correct = LIA is not a period but could state during the time period of the LIA).

Wisconsin—was the last major advance of continental glaciers in North America 115,000–11,700 years before present.

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Appendix A. Common and Associated Scientific Plant Names From Text

Common Names	Scientific Names
Trees	
Alligator juniper	*Juniperus deppeana*
Aspen	*Populus tremuloides*
Bigtooth maple	*Acer grandidentatum*
Bristlecone pine	*Pinus longaeva*
Douglas-fir	*Pseudotsuga menziesii*
Eastern redcedar	*Juniperus virginiana*
Engelmann spruce	*Picea engelmannii*
Gambel oak	*Quercus gambeli*
Grand fir	*Abies grandis*
Limber pine	*Pinus flexilis*
Oneseed	*Juniperus monosperma*
Ponderosa pine	*Pinus ponderosa*
Rocky Mountain juniper	*Juniperus scopulorum*
Sierra juniper	*Juniperus grandis*
Singleleaf pinyon	*Pinus monophylla*
Subalpine fir	*Abies lasiocarpa*
Twoneedle pinyon	*Pinus edulis*
Utah juniper	*Juniperus osteosperma*
Western juniper	*Juniperus occidentalis*
White fir	*Abies concolor*
Whitebark pine	*Pinus albicaulis*
Shrubs	
Antelope bitterbrush	*Purshia tridentata*
Apache plume	*Fallugia paradoxa*
Basin big sagebrush	*Artemisia tridentata* spp. *tridentata*
Black sagebrush	*Artemisia nova*
Cliffrose	*Purshia mexicana*
Common juniper	*Juniperus communis*
Creeping juniper	*Juniperus horizontalis*
Curl-leaf mountain mahogany	*Cercocarpus ledifolius*
Desert peach	*Prunus andersonii*
Greasewood	*Sarcobatus vermiculatus*
Green rabbitbrush	*Chrysothamnus viscidiflorus*
Little rabbitbrush	*Chrysothamnus viscidiflorus* spp. *vicidiflorus* var. *stenophyllus*
Low sagebrush	*Artemisia arbuscula*
Mountain big sagebrush	*Artemisia tridentata* spp. *vaseyana*
Nuttall's saltbush	*Atriplex nuttallii*
Rubber rabbitbrush	*Ericameria nauseosa*
Salt rabbitbrush	*Ericameria nauseosa* spp. *consimilis*
Serviceberry	*Amelanchier alnifolia*
Skunkbush sumac	*Rhus trilobata*

Common Names	Scientific Names
Shadscale	*Atriplex confertifolia*
Snakeweed	*Gutierrezia sarothrae*
Snowberry	*Symphoricarpos* spp.
Spiny hopsage	*Grayia spinosa*
Winterfat	*Krascheninnikovia ceratoides*
Wyoming big sagebrush	*Artemisia tridentata* spp. *wyomingensis*
Grasses	
Alkali cordgrass	*Spartina gracilis*
Basin wildrye	*Leymus cinereus*
Blue grama	*Bouteloua gracilis*
Bluebunch wheatgrass	*Pseudoroegneria spicata*
Bottlebrush squirreltail	*Elymus elymoides*
Cheatgrass	*Bromus tectorum*
Columbia needlegrass	*Achnatherum nelsonii*
Idaho fescue	*Festuca idahoensis*
Crested wheatgrass	*Agropyron cristatum*
Indian ricegrass	*Achnatherum hymenoides*
Intermediate wheatgrass	*Thinopyrum intermedium*
James' galleta	*Pleuraphis jamesii*
Junegrass	*Koeleria macrantha*
Lemmon's needlgrass	*Achnatherum lemmonii*
Lettermen's needlegrass	*Achnatherum lettermanii*
Medusahead	*Taeniatherum caput-medusae*
Mountain brome	*Bromus marginatus*
Muttongrass	*Poa fendleriana*
North African grass	*Ventenata dubia*
Pine needlegrass	*Achnatherum pinetorum*
Red brome	*Bromus rubens*
Saltgrass	*Distichlis spicata*
Sandberg bluegrass	*Poa secunda*
Spike fescue	*Leucopoa kingii*
Thurber's needlegrass	*Achnatherum thurberianum*
Western needlegrass	*Achnatherum occidentale*
Birds	
American robin	*Turdus migratorius*
Bohemian waxwing	*Bombycilla garrulus*
Cedar waxwing	*Bombycilla cedrorum*
Clark's nutcracker	*Nucifraga columbiana*
Mountain bluebirds	*Sialia currucoides*
Mountain chickadees	*Poecile gambeli*
Pinyon jay	*Gymnorhinus cyanocephalus*
Red-breasted nuthatch	*Sitta canadensis*
Scrub-jays	*Aphelocoma californica* and *A. woodhouseii*
Townsend's solitaire	*Myadestes townsendi*
Western bluebirds	*Sialia mexicana*

(Continued)

Common Names	Scientific Names
Mammals	
Black bear	*Ursus americanus*
Black-tailed deer	*Odocoileus hemionus* spp. *columbianus*
California ground squirrel	*Otospermophilus beecheyi*
California kangaroo rat	*Dipodomys californicus*
Chipmunks	*Tamias* spp.
Cottontail rabbit	*Sylvilagus audubonii*
Coyote	*Canis latrans*
Deer mouse	*Peromyscus maniculatus*
Great Basin pocket mouse	*Perognathus parvus*
Jackrabbits	*Lepus* spp.
Least chipmunk	*Tamias minimus*
Mule deer	*Odocoileus hemionus*
Nuttall's cottontail	*Sylvilagus nuttallii*
Panamint kangaroo rat	*Dipodomys panamintinus*
Pinyon mouse	*Peromyscus truei*
Raccoon	*Procyonidae lotor*
Red fox	*Vulpes vulpes*
Ringtail	*Bassariscus astutus*
Virginia opossum	*Didelphis virginiana*
White-tailed antelope ground squirrel	*Ammospermophilus leucurus*
White-tailed deer	*Odocoileus virginianus*
Woodrats	*Neotoma cinerea* and *N. fuscipes*
Yellow-pine chipmunk	*Tamias amoenus*

Appendix B. Phase and Tree Dominance Index (TDI)

Phase I, II, III—In Phase I, trees are present but shrubs and herbs are the dominant vegetation influencing ecological processes on the site. In Phase II, trees are codominant with shrubs and herbs and all three vegetation layers influence ecological processes. In Phase III, trees are the dominant vegetation on the site and the primary plant layer influencing ecological processes on the site (from Miller et al. 2005). Phases can be quantified using perennial cover to calculate a total tree dominance index (TDI) (Williams et al. 2017).

- Phase I = total tree / total tree + shrub + perennial grass = less than 0.34 (tree cover less than one-third)

- Phase II = total tree / total tree + shrub + perennial grass = 0.34–0.67 (tree cover one-third to two-thirds)

- Phase III = total tree / total tree + shrub + perennial grass = more than 0.66 (tree cover more than two-thirds)

Tree dominance index (TDI) is a quantitative measure of the relative dominance of conifers based on the proportion of tree canopy cover to the summation of shrub and perennial grass (or herb) cover and is calculated as follows:

- Tree cover / [tree + shrub + tall perennial grass cover]

For example, TDI = 0.67 occurs when tree canopy is twice that of perennial grasses and shrubs (Roundy et al. 2014a). Tree cover can vary widely with both phase and TDI, depending on ecological site characteristics (from Williams et al. 2017).
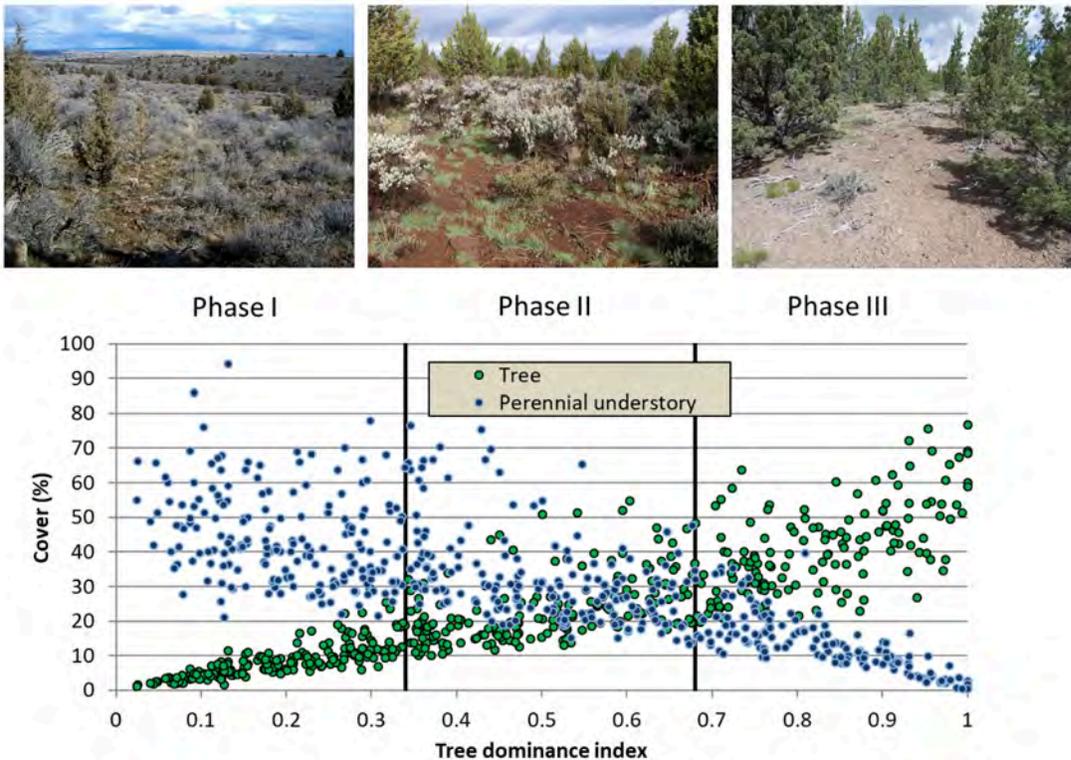
Appendix C. State and Transition Model

Part I: State and Transition model illustrating potential changes in an old-growth twoneedle pinyon/mountain big sagebrush/bluebunch wheatgrass community following different disturbances (Stringham et al. 2015).

PIMO/JUOS/ARTRV/PSSP-ACTH7

Reference State 1.0 Community Pathways

1.1a: High-severity crown fire reduces or eliminates tree cover.

1.1b: Time and lack of disturbance such as fire, disease, or drought allows younger trees to infill.

1.2a: Time and lack of disturbance such as fire or drought. Excessive herbivory may also reduce perennial grass understory.

1.3a: Fire.

1.3b: Time and lack of disturbance such as fire or drought. Excessive herbivory may also reduce perennial grass understory.

1.4a: Low-severity fire, insect infestation, or disease removes individual trees and reduces total tree cover.

1.4b: High-severity crown fire reduces or eliminates tree cover.

Transition T1A: Introduction of nonnative annual species.

Transition T1B: Time and a lack of disturbance allows for trees to dominate site resources; may be coupled with inappropriate grazing management that favors shrub and tree dominance.

Current Potential State 1.0 Community Pathways

2.1a: High-severity crown fire reduces or eliminates tree cover.

2.1b: Time and lack of disturbance such as fire, disease, or drought allows younger trees to infill.

2.2a: Time and lack of disturbance such as fire or drought. Excessive herbivory may also reduce perennial grass understory.

2.3a: Fire.

2.3b: Time and lack of disturbance such as fire or drought. Excessive herbivory may also reduce perennial grass understory.

2.4a: Low-severity fire, insect infestation, or disease removes individual trees and reduces total tree cover.

2.4b: High-severity crown fire reduces or eliminates tree cover.

Transition T2A: Time and a lack of disturbance allows for trees to dominate site resources; may be coupled with inappropriate grazing management that favors shrub and tree dominance.

Transition T2B: Catastrophic fire.

Infilled Tree State 3.0 Community Pathways

3.1a: Time and lack of disturbance such as fire, disease, or drought allows younger trees to infill.

Transition T3A: Catastrophic fire.

Transition T3B: Loss of understory vegetation destabilizes soil surface. Inappropriate

grazing management may further reduce the perennial grass understory.

Restoration Pathway R3A: Thinning of trees coupled with seeding. Success unlikely from State 3.2.

Annual State 4.0 Community Pathways

None

Transition T4A: Catastrophic fire or multiple fires.

Eroded State 5.0 Community

Part II: State and transition model, illustrating possible woodland expansion and successional stages in a Wyoming big sagebrush/bluebunch wheatgrass with different types and severities of disturbance (Stringham et al. 2015).

Gravelly Loam 10–12 inches

Reference State 1.0 Community Phase Pathways

1.1a: Low-severity fire or aroga moth infestation creates sagebrush/grass mosaic; high-severity fire significantly reduces sagebrush cover and leads to early/mid-seral community dominated by grasses and forbs.

1.1b: Time and lack of disturbance such as fire or drought. Excessive herbivory may also decrease perennial understory.

1.2a: Time and lack of disturbance such as fire allows for regeneration of sagebrush.

1.3a: Low-severity fire or aroga moth infestation creates sagebrush/grass mosaic.

1.3b: High-severity fire or severe aroga moth infestation significantly reduces sagebrush cover leading to early/mid-seral community.

Transition T1A: Introduction of nonnative annual species.

Current Potential State 2.0 Community Phase Pathways

2.1a: Low-severity fire or aroga moth infestation creates sagebrush/grass mosaic; high-severity fire significantly reduces sagebrush cover and leads to early/mid-seral community dominated by grasses and forbs; nonnative annual species present.

2.1b: Time and lack of disturbance such as fire or drought. Inappropriate grazing management may also decrease perennial understory.

2.2a: Time and lack of disturbance such as fire allows for regeneration of sagebrush.

2.2b: Tree/shrub removal treatment or prescribed burning in the presence of annual grass species coupled with heavy spring precipitation.

2.3a: Low-severity fire or aroga moth infestation creates sagebrush/grass mosaic.

2.3b: High-severity fire or severe aroga moth infestation significantly reduces sagebrush cover leading to early mid-seral community.

Transition T2A: Time and lack of disturbance and/or inappropriate grazing management (3.1).

Transition T2B: Catastrophic fire or multiple fires (4.1), inappropriate grazing management in the presence of annual nonnative species (4.2).

Transition T2C: Time and lack of disturbance allows for trees to dominate site resources; may be coupled with inappropriate grazing management (5.1).

Shrub State 3.0 Community Phase Pathways

None.

Transition T3A: Catastrophic fire and/or soil disturbing treatments (4.1). Inappropriate

grazing management in the presence of annual nonnative species (4.2)

Transition T3B: Time and a lack of disturbance allows for trees to dominate site resources; may be coupled with inappropriate grazing management that favors shrub and tree dominance (5.1)

Restoration R3A: Shrub removal/management with minimal soil disturbance coupled with seeding of desired species (2.4).

Annual State 4.0 Community Phase Pathways

4.1a: Time and lack of disturbance (unlikely to occur).

4.2a: Fire.

Restoration R4A: Seeding of deep-rooted perennial bunchgrasses, probability of success is medium.

Tree State 5.0 Community Phase Pathways

5.1a: Time and lack of disturbance allows for tree maturation.

5.2a: Stand thinning treatment.

Restoration R5A: Tree removal with minimal soil disturbance and seeding of desired species (2.2).

Restoration R5B: Coming from Tree State 5.1 only: tree-removal treatment or controlled burn. Controlled burn only recommended if nonnative annuals are not present.

Restoration R5C: Tree removal coupled with seeding of desired species (6.2.)

Transition T5A: Catastrophic fire.

Seeded State 6.0 Community Phase Pathways

6.1a: Time and lack of disturbance may be coupled with inappropriate grazing management.

6.2a: Low-severity fire or aroga moth infestation.

6.2b: Time and lack of disturbance coupled with inappropriate grazing management reduces bunchgrasses and increases density of sagebrush; usually a slow transition.

6.3a: Fire, aroga moth infestation, or brush treatment with minimal soil disturbance.

Transition T6A: Time and lack of disturbance allows for trees to dominate site resources; may be coupled with inappropriate grazing management (5.1).

Transition T6B: Severe fire. This transition usually originates from 6.3.

MLRA 28
Group 23
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028BY062NV

Reference State 1.0

1.1
Old growth Utah juniper and singleleaf pinyon dominate; canopy cover 10-20%
Mountain big sagebrush, bluebunch wheatgrass and Thurber's needlegrass dominate
Few tree seedlings and sapling occur in understory

1.2
Bluebunch wheatgrass, Thurber's needlegrass and other perennial bunchgrasses dominate
Forbs increase
Pinyon and juniper seedlings present
Mountain big sagebrush present in unburned patches
Burned tree skeletons present

1.1a →

1.1b ↓   1.4a ↑

1.4b   1.3a

1.2a   1.3b

1.4 (at risk)
Multiple age classes of Utah juniper and singleleaf pinyon dominate
Canopy cover increases
Mountain big sagebrush reduced and/or decadent
Perennial bunchgrasses reduced
Bare ground increases

1.3
Immature Utah juniper and singleleaf pinyon trees dominate
Perennial bunchgrasses and mountain big sagebrush dominate understory

T1A

Current Potential State 2.0

2.1
Old growth Utah juniper and singleleaf pinyon dominate; canopy cover 10-20%
Mountain big sagebrush, bluebunch wheatgrass, Thurber's needlegrass dominate
Few tree seedlings and saplings occur in understory
Annual non-native species present

2.2
Bluebunch wheatgrass, Thurber's needlegrass and other perennial bunchgrasses dominate
Forbs increase
Pinyon and juniper seedlings present
Mountain big sagebrush present in unburned patches
Annual non-native species present

2.1a →

2.1b ↓   2.4a ↑

2.4b   2.3a

2.2a   2.3b

2.4 (at risk)
Multiple age classes of Utah juniper and singleleaf pinyon dominate
Canopy cover increases
Mountain big sagebrush reduced and/or decadent
Perennial bunchgrasses reduced
Bare ground increases
Annual non-native species present

2.3
Immature Utah juniper and singleleaf pinyon trees dominate
Perennial bunchgrasses and mountain big sagebrush dominate understory
Annual non-native species present

T1B

Annual State 4.0

4.1
Annual non-native species dominate
Trace amounts of perennial bunchgrasses may be present

T2A   R3A   T2B

T3A →

Infilled Tree State 3.0

3.1
Pinyon and juniper dominate overstory; canopy cover >30%
Understory vegetation thinning
Mountain big sagebrush skeletons common
Annual non-natives may be present
Bare ground areas increasing

3.1a →

3.2
Pinyon and juniper dominate overstory; canopy cover >50%
Understory vegetation sparse or absent
Annual non-natives may be present
Bare ground areas large and connected
Soil redistribution may be extensive

Index

Seeds: production, caching, dispersal, establishment 9, 20, 21, 22, 23, 24, 25, 26, 30, 31, 32, 33, 35, 36, 37, 38, 39, 40, 41, 42, 47, 169, 170, 171, 176, 198, 199, 202, 214, 216, 219, 222, 226, 230, 236, 241, 257, 258, 264, 270

Shredding 160, 163, 165, 168, 172, 173, 174, 175, 176, 178, 179, 182, 195, 199, 221, 223

Shrub succession 52

Sierra juniper 5, 8, 14, 17, 18, 22, 23, 24, 272

Snake River Basin 113

Snakeweed 55, 61, 192, 263

Snowberry 8, 54, 88, 90, 93, 150, 204, 210

Snowpack 70, 71, 73, 134, 150, 230, 233, 247, 261

Soil moisture 6, 9, 13, 16, 17, 18, 19, 22, 30, 48, 62, 64, 71, 72, 73, 74, 78, 80, 82, 134, 147, 149, 159, 163, 164, 173, 183, 186, 192, 194, 197, 198, 199, 201, 204, 207, 208, 209, 210, 220, 240, 250, 255, 268

Soil temperature regimes 6, 22, 207, 208

Soil water 13, 15, 42, 62, 64, 69, 80, 81, 82, 129, 130, 141, 142, 148, 149, 151, 152, 154, 155, 157, 159, 161, 163, 169, 173, 179, 193, 195, 199, 211, 233, 242, 251, 255, 257, 258

Soil water competition 62

Soil water repellency 141, 142, 149, 242, 255

Spatial heterogeneity 68

Stemflow 149, 156, 221, 258

Stem rust 35, 45

Successional stages 9, 48, 50, 208, 238, 277

Successional trajectories 167, 228

Surface fuel 57, 79, 113, 173, 176, 177, 190, 211

T

Tablelands 57, 58

Taproot 28, 30

Taxonomy 264

Temperature regime 3, 179, 202, 203, 204, 208

Topography 1, 7, 12, 17, 19, 50, 62, 63, 88, 111, 113, 128, 129, 131, 148, 150, 157, 188, 190, 202, 203, 219

Transpiration 26, 27, 42, 149, 151, 152, 153, 161, 162, 215, 231, 240, 251

Tree dominance index 54, 180, 183, 184, 205, 275

Tree removal 4, 69, 129, 154, 159, 163, 164, 165, 166, 172, 178, 180, 186, 199, 200, 252, 253, 268

Tree-ring chronology 120

Tree succession 55

Two-leaf pinyon 25

Twoneedle pinyon var. fallax 26

U

Understory 48, 54, 170, 217, 228, 233, 234, 237, 256, 259

V

Vegetation Zones 7, 8, 12, 85, 94, 97, 210

W

Wasatch Mountains 6, 7, 25, 113

Water budget 131, 132, 134, 144, 145, 146, 153, 155

Watershed 145, 154, 156, 160, 162, 172, 226, 248, 252, 256, 263

Wildfire 10, 68, 69, 76, 81, 82, 127, 161, 188, 194, 198, 206, 213, 224, 226, 234, 237, 238, 241, 242, 243, 246, 250, 252, 253, 259, 266, 268

Wildlife habitat 4, 9, 166

Windrowing 171, 188

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