A Review of Fire Effects on Vegetation and Soils in the Great Basin Region: Response and Ecological Site Characteristics

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Santa Rosa Mountains, northwest Nevada.
Photo courtesy of Nolan Preece.

ABSTRACT

This review synthesizes the state of knowledge on fire effects on vegetation and soils in semi-arid ecosystems in the Great Basin Region, including the central and northern Great Basin and Range, Columbia River Basin, and the Snake River Plain. We summarize available literature related to: (1) the effects of environmental gradients, ecological site, and vegetation characteristics on resilience to disturbance and resistance to invasive species; (2) the effects of fire on individual plant species and communities, biological soil crusts, seed banks, soil nutrients, and hydrology; and (3) the role of fire severity, fire versus fire surrogate treatments, and post-fire grazing in determining ecosystem response. From this, we identify knowledge gaps and present a framework for predicting plant successional trajectories following wild and prescribed fires and fire surrogate treatments. Possibly the three most important ecological site characteristics that influence a site's resilience (ability of the ecological site to recover from disturbance) and resistance to invasive species are soil temperature/moisture regimes and the composition and structure of vegetation on the ecological site just prior to the disturbance event.

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Summary

Our goal was to make this synthesis as comprehensive as possible by summarizing all of the available literature related to fire effects on vegetation and soils within the boundaries of the Great Basin Region (central and northern Great Basin and Range, Columbia River Basin, and Snake River Plain). The review is intended to: (1) serve as a reference for writing environmental assessments and management plans, (2) identify the key ecological site components that influence plant-fire and soil-fire interactions, (3) provide criteria for selecting appropriate ecological sites for treatment, and (4) increase the ability to predict outcomes following fire. Key components that influence plant and soil responses to fire include: fire severity, pre- and post-fire weather, ecological site characteristics, current vegetation (vegetation present on the site prior to the disturbance event), and post-fire disturbance. Possibly the three most important ecological site characteristics that influence a site's resilience (ability of the ecological site to recover from disturbance) and resistance to invasive species are soil temperature/moisture regimes and the composition and structure of vegetation on the ecological site just prior to the disturbance event. However, other soil characteristics, severity of the disturbance(s), post-fire weather, and post-fire disturbance and management modify the response. All of these components need to be considered when predicting or evaluating fire effects in Great Basin communities. The largest knowledge gaps that we identified in this review are the lack of long-term fire studies (>10 years) and studies that evaluate effects of repeated burns.

How to Use This Synthesis

This synthesis is not meant to be read from cover to cover. Divided into sections and subsections, it can be used as tool to find specific information related to fire effects. For example, a reader may wonder what are the impacts of fire on seed banks and the importance of seed banks for recovery? Or, what is the expected rate of recovery for deep-rooted perennial grasses following fire? The Contents act as a road map to locations of information on specific topics and on the effects of different conditions. Also, figures summarize important concepts and key points. For example, fig. 12 identifies the primary ecological site characteristics linked to resilience and resistance, and fig. 15 identifies the key components that influence plant succession following fire. Glossary terms are highlighted in bold text throughout the synthesis. The photographs highlight and reinforce key points throughout the text. The organization of this synthesis is intended to facilitate its use as a reference tool for planning and decision making.

CONTENTS

Introduction

Possibly one of the most important roles of a resource manager is the manipulation of vegetation across broad landscapes. Vegetation structure and composition are important community characteristics that influence ecological processes, including hydrologic, nutrient, and energy cycles; disturbance regimes; and ecological services. Vegetation composition and structure are also important components of resilience to disturbance (the ability of an ecological site to recover) and resistance to invasive species. A key element of successful vegetation management is the ability to evaluate recovery potential following disturbance and predict possible successional trajectories of an ecological site following various management practices (figs. 1A and B). The ability to predict outcomes following disturbances allows managers to develop effective restoration plans, select appropriate ecological sites and methods for treatment, design long-term monitoring and management plans, meet objectives, and justify management action.

The overall purpose of this synthesis is to: (1) provide a summary of the scientific knowledge based on the available literature related to plant and soil responses to fire in the Great Basin Region; (2) identify the ecological site and vegetation characteristics that influence resilience to disturbance and resistance to invasive species; (3) present a framework for predicting plant successional trajectories following wild and prescribed fires; and (4) determine knowledge gaps.

Figure 1. A key to successful management is the ability to evaluate the recovery potential (resilience) of a landscape, which is frequently a heterogeneous combination of different elevations, soils, aspects, and pre-fire communities. Predicting the short- and long-term outcomes requires knowledge of: (1) site characteristics, including soil temperature and moisture regimes and pre-fire plant composition and structure; (2) the potential fire severity that has or will occur as affected by the plant community (fuels) and weather; and (3) post-treatment management. **(A)** Recovery to native vegetation is shown on a site in Five Creeks, Steens Mountain, Oregon (photo by Rick Miller). **(B)** Cheatgrass is the dominant understory herb with native shrub establishment in Underdown Canyon, Nevada (photo by Jeanne Chambers). Although soil temperature/moisture regimes are similar between the two sites, the native herbaceous understory was relatively abundant in (A) and severely depleted by post-treatment livestock grazing in (B). Fire severity may also be a factor in differences between site recovery (see fig. 15).

Key Questions Addressed

(1) How does fire influence post-fire plant composition and structure in the short and long term?

(2) What are the primary ecological site characteristics that influence resilience and plant successional trajectories following fire?

(3) What ecological site characteristics influence resistance to invasive species?

(4) How does fire alter soil nutrients and carbon dynamics, hydrologic processes, and biological soil crusts?

(5) What is known and not known about fire-plant interactions in the Great Basin Region?

To address these questions, this synthesis provides:

(1) specific questions and a conceptual model for predicting plant succession following treatment;

(2) a framework for selecting the most appropriate ecological sites and treatments;

(3) a condensed but comprehensive synthesis of fire effects on plant species and communities and soils in the Great Basin Region that can be used in the development and support of environmental assessments (EA) and management plans; and

(4) definitions related to fire and ecology that are useful in developing EAs and management plans.

We attempted to identify the key ecological site characteristics in each study we reviewed, including soil temperature/moisture regimes, seral stage of vegetation at the time of treatment, and precipitation regime following treatment. If soil temperature/moisture regimes were not provided (which was often the case), we used elevation, average annual precipitation, dominant subspecies of sagebrush (for scientific names see Appendix 1), and site location (Major Land Resource Area, table 1; fig. 2) to assign a regime. In combination, these criteria determined the elevation breaks for soil temperature regimes—mesic, frigid, and cryic (equivalent in this synthesis to warm, cool, and cold, respectively). We discuss the use of fire surrogates, such as mechanical treatments, in several of the sections. We do not address post-fire treatments.

Organization and Content

This synthesis is separated into five primary sections:

Section 1 identifies boundaries and describes the region addressed. It also introduces concepts of resilience to disturbance and resistance to invasive species and illustrates the importance of environmental gradients, particularly moisture and temperature, within the Great Basin Region that influence recovery potential and plant successional trajectories following fire.

Section 2 describes the key components for predicting plant succession following wild and prescribed fires, which include ecological site characteristics, disturbance history, current vegetation, pre- and post-fire weather, and post-fire management. These components form the basic framework of questions managers should ask when predicting plant successional trajectories following management and when selecting appropriate ecological sites and treatments. Here, we use the term, "characteristic," to refer to a distinctive feature or quality of the key components that influences plant successional trajectories, resilience to disturbance, and resistance to invasives. These include characteristics of the fire event (e.g., severity), ecological site, composition and structure of pre-fire vegetation, and weather regimes. It is helpful to consider these characteristics when developing, justifying, and implementing management plans.

Sections 3 and 4 summarize the literature related to fire-plant and fire-soil interactions in the Great Basin Region. The fire-plant section is organized into subsections related to functional groups, species, and plant characteristics. The fire-soil section contains three subsections related to fire soil temperatures, nutrients, hydrology, and erosion. Each of

Table 1. Major Land Resource Area (MLRA) names and identification numbers (see map fig. 2) included in this synthesis. For each MLRA, the size of land area, states (acronyms) listed in descending order of proportion of area covered, most common elevation range (extreme), geology, common soil orders, range of average annual precipitation (PPT) (extremes), and range in temperatures are described. The synthesis only covers the sagebrush, salt-desert, 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.)

MLRA (id #)	Area (mi²)	States	Elevation (ft) (range)	Geology	Soils	PPT (inches) (range)	Temp (°F)
Columbia Plateau (8)	18,505	WA OR ID	1300-3600	basalt	Mollisols	10-16 (6-36)	48-54
Blue Mt Foothills (10)	17,515	OR ID	1300-6600	basalt alluvium sedimentary	Mollisols Aridisols	8-16 (41)	36-53
Snake River Plain (11)	16,475	ID OR	2100-5000	Idaho batholith, basalts	Aridisols	7-12 (20)	41-55
Klamath Valleys (21)	11,495	CA OR	2600-4600 (>7000)	basalt rhyolite andesite	Mollisols	12-30 (9; 30-58)	39-52
Malheur High Plateau (23)	22,896	OR NV CA	3900-6900 (>9000)	basalt & andesite	Aridisols & Mollisols	6-12 (>50)	39-52
Humboldt Area (24)	12,680	NV OR	3950-5900 (>8850)	alluvium (some andesite & basalt)	Entisols Inceptisols Mollisols	6-12 (40)	38-53
Owyhee High Plateau (25)	28,930	NV ID OR UT	3000-7550 (>9800)	andesite basalt rhyolite	Aridisols Mollisols	7-16 (>50)	35-53
Carson Basin & Mts (26)	6520	NV CA	3900-6550 (13,100)	granitic andesite basalt	Aridisols Mollisols	5-36	37-54
Fallon-Lovelock (27)	12,565	NV CA	3300-5900 (<7800)	alluvium andesite basalt	Aridisols Entisols	5-10 (19)	43-54
Central NV Basin & Range (28B)	23,555	NV	3950-6560 basin 6560-11150 mts	playa lakebed deposits, old-sedimentary	Aridisols Entisols Mollisols	5-12 (49)	39-53
Great Salt Lake (28A)	36,775	UT NV ID	4900-6550 basin 6550-11,900 mts	carbonate (north) andesite basalt (south) alluvium & lakebed deposits	Aridisols Entisols Mollisols	4-12 (basins) 8-36 (mountains)	34-52

those includes key questions related to the topic and a synthesis of the literature followed by a brief summary and statement of knowledge and uncertainty based on the strength of the literature.

Section 5 provides a framework of questions and considerations that can be used as a tool for predicting vegetation response following fire, selecting the most appropriate ecological sites to be treated, and deciding if fire is the most appropriate tool. This section is supported by a field guide (Miller and others in preparation a) that identifies a location, ecological site, or management unit's resilience to disturbance and resistance to invasives based on this synthesis.

The Glossary defines terms used in fire and plant ecology that are useful when writing environmental assessments. Glossary terms are highlighted in bold text throughout the document. The appendices list common and scientific plant names used in the synthesis and the Raunkiaer life form classification system used by the U.S. Forest Service Fire Effects Information System.
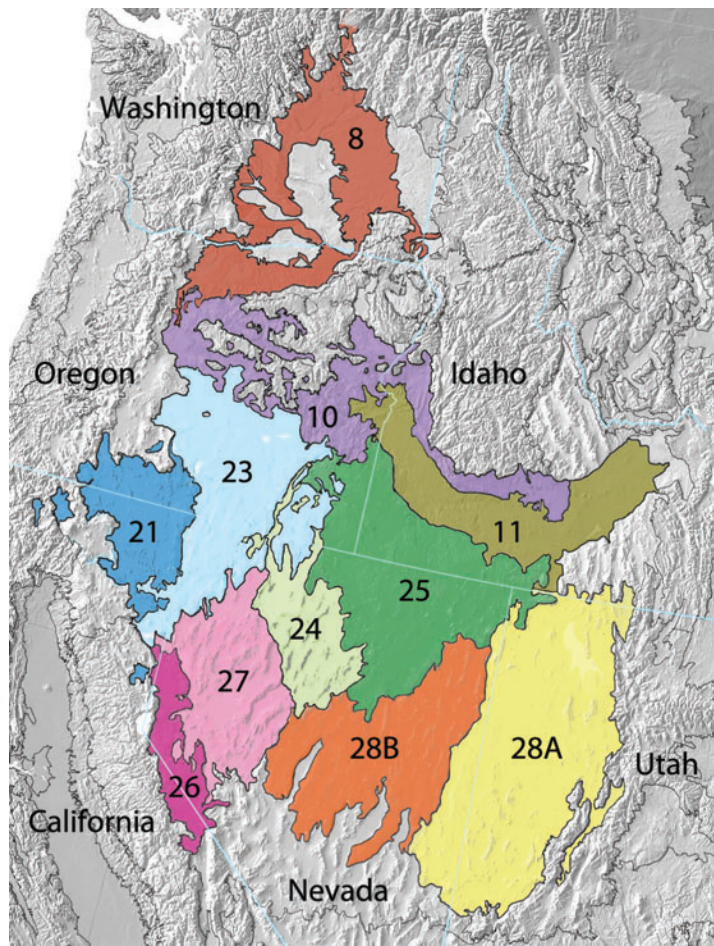
Figure 2. Major Land Resource Areas (MLRAs) located in the Great Basin and Columbia Basin Region; Columbia Plateau (8), Blue Mountain Foothills (10), Snake River Plain (11), Klamath Valleys (21), Malheur High Plateau (23), Humboldt Area (24), Owyhee High Plateau (25), Carson Basin and Mountains (26), Fallon-Lovelock (27), Great Salt Lake (28A), and Central Nevada Basin and Range (28B) (derived from USDA-NRCS 2011 by Eugénie MontBlanc, University of Nevada, Reno).

Section 1: The Great Basin Region

The geographic boundaries of this synthesis extend from the sagebrush and salt-desert regions of the Columbia River Basin and Snake River Plain in the north to the southern boundary of the Central Basin and Range in Nevada and Utah (figs. 2, 3). From west to east, the boundary stretches from the east slopes of the Sierra and Cascade mountains in California, Oregon, and Washington to the west slopes of the Wasatch Mountains in Utah. The region covers nearly 200,000 mi^2 (520,000 km^2) of the semi-arid Intermountain West and encompasses the Central Basin and Range, Northern Basin and Range, Columbia Basin, Snake River Plain, and Blue Mountain ecoregions (fig. 3) (U.S. Environmental Protection Agency 2011). The synthesis includes the floristic sections of the northern and southern Great Basin, Snake River Plain, and Columbia Basin (Küchler 1970; West 1983b), and 11 **Major Land Resource Areas** (MLRAs; see glossary for definition; USDA-NRCS 2011) (table 1; fig. 2) and extends across southern Washington, eastern Oregon, northeastern California, southern Idaho, the northern two-thirds of Nevada, and the western half of Utah. We loosely refer to this area as the Great Basin Region. This synthesis does not include the Colorado Plateau where both the geology and climate (primarily summer precipitation) differ from the Great Basin Region. Although this synthesis covers a large, heterogeneous region, the applicability of research from one location to another within and across MLRAs is largely determined by specific characteristics of the **ecological site** and disturbance event rather than the spatial location (fig. 4).
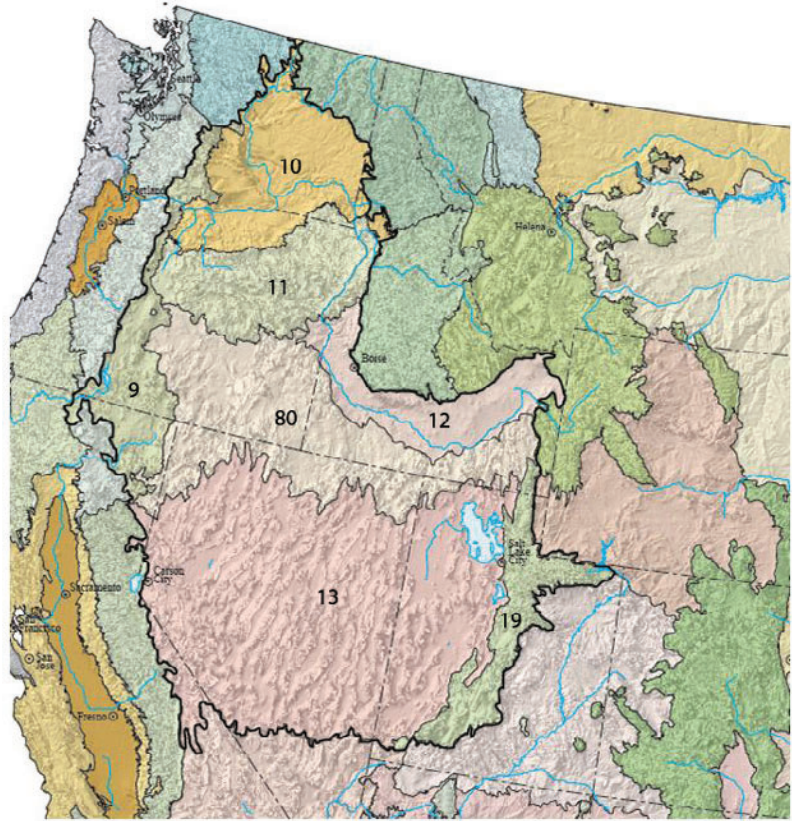
Figure 3. This synthesis covers literature from shrub steppe and shrub desert communities within the ecoregions of the Central Basin and Range (#13), Northern Basin and Range (#80), Columbia Basin (#10), Snake River Plain (#12), Blue Mountain (#11), Eastern Cascades Slopes and Foothills (#9), and Wasatch Uinta Mountains (#19) (from U.S. Environmental Protection Agency 2011). Although based on similar characteristics as MLRAs, ecoregions rely more heavily on general similarities of ecosystems and environmental resources while MLRAs place greater emphasis on soils and agricultural uses.

In the following sub-sections, we describe the topography, climate, soils, geology, and vegetation of the region. We also introduce the topics of **resilience to disturbance** and **resistance to invasives** and key ecological site characteristics that influence these two concepts. Last, we define units of scale used throughout the text and the potential impacts of global change.

Figure 4. The applicability of research from one location to another within and across MLRAs is largely determined by specific characteristics of the ecological site and disturbance event rather than the spatial location. The potential for recovery of different plant functional groups, such as deep-rooted perennial grasses, and a site's resistance to invasive species, throughout the Great Basin Region, depends on ecological site characteristics such as soil temperature and moisture regimes, pre-fire plant species composition and structure, fire severity, and post-treatment disturbance. (Photo by Rick Miller.)

Topography and Environmental Gradients

The Great Basin Region is characterized by basins, mountains, and plateaus that range in elevation from 1300 to >10,000 ft (400 to 3000 m) (the majority lying between 2500 and 7500 ft (760 to 2300 m) (USDA-NRCS 2011). The basins, which are located at the lowest elevations, are typically 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). Differences in elevation, modified by aspect, slope, and soils (primarily depth and texture), create environmental gradients in both moisture and thermal regimes. These gradients produce variation in potential vegetation, productivity, resilience, and resistance to invasive species (Chambers and others 2007) (figs. 5, 6), which in turn influence how ecological sites respond to disturbance (Chambers and others in press).

Climate

The region is characterized by a semi-arid temperate climate with cold, wet winters, wet springs, and warm, dry summers. Amounts of effective summer precipitation are low but typically increase slightly toward the eastern and southern portions of the region. However, the summer monsoonal influence is weak compared to regions lying to the south (Colorado Plateau and the Southwest Hot Deserts). Annual precipitation across much of the region ranges from 6 to 12 inches (150 to 300 mm) at the lower to mid elevations and 12 to 16 inches (300 to 400 mm) at mid to upper elevations. However, extremes range from <6 inches (150 mm) in some of the arid basins to >20 inches (510 mm) at the higher mountain elevations. Precipitation is spatially and temporally highly variable, with variation usually decreasing as precipitation increases (MacMahon 1980).

Soils and Geology

Alluvium and playa lakebed deposits typically fill the basins (USDA-NRCS 2011). Foothills and mountains are commonly composed of volcanic rock (basalt, andesite, and

Figure 5. Conceptual model of (A) resilience to disturbance and (B) resistance to invasives as they relate to three environmental gradients: soil temperature and moisture regimes, elevation, and productivity. Soil moisture availability is also modified by soil characteristics. The solid blue line in the top graph is the resilience as it relates to elevation (dashed dark blue and red lines adjust for aspect), soil temperature/moisture regimes, and productivity. The solid green line in the bottom graph represents potential resistance to invasives in the reference state, and the red line is the decline in resistance as a result of a site that is in an at-risk-phase or global climate change. The potential resilience and resistance of a site is determined by ecological site characteristics. The potential of both can be lowered if certain site components (e.g., perennial grass abundance) are depleted as a result of disturbance history or climate change. The separation between soil temperature/moisture regimes is a gradient (represented by the overlapping arrows) and is not separated by distinct boundaries. Changes in soil temperature and moisture can be gradual (e.g., along a gradual increase in elevation) or abrupt (e.g., shift from a south to an opposing north aspect). An overlap of mountain big sagebrush (ARTRv) into warm (mesic) soils and Wyoming big sagebrush (ARTRw) into cool (frigid) soils can also occur and is often influenced by soil moisture availability. The shift from one sagebrush subspecies to another does not have a definite lower or upper elevation limit, but will vary with other site attributes including aspect, soils, location (MLRA), and microtopography. The mountain big sagebrush-snowberry (ARTRv-SYOR) type would be similar to what is often called the mountain shrub in Nevada and Utah (and often includes mountain big sagebrush, snowberry, serviceberry, bitterbrush, and curl-leaf mountain mahogany). As environmental gradients move to the right, resilience, resistance, and biomass increase, thus increasing fuel loads and the potential for more frequent fires. The relationship between soil temperature/moisture regimes and elevation changes across MLRAs. Adapted from Chambers and others in press, in review.
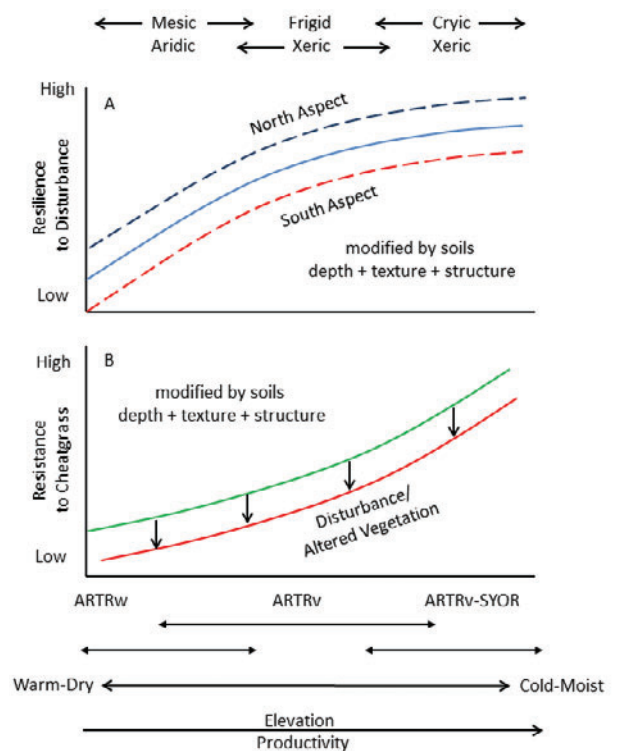
Figure 6. Changes in elevation, modified by aspect, slope, and soils (e.g., depth and texture), create environmental gradients in both moisture and thermal regimes that are associated with predictable changes in potential vegetation, productivity, resilience, and resistance to invasive species. These gradients occur at a range of scales across valley floors to mountain tops, opposing south and north aspects, and concave to convex surfaces (Schell Creek Mountains, Pine County, Nevada; photo by Rick Miller).

rhyolite) with occasional granitics. However, carbonate soils in the Central Nevada Basin and Range and sedimentary soils in the Great Salt Lake Region are also important (table 1). The most common soil orders are Mollisols, Aridisols, and Entisols. Soils are key components in determining potential vegetation, productivity (Passey and Hugie 1962, 1963a; Hironaka 1963; Hugie and others 1964; Hugie and Passey 1964; Jensen 1989; Jensen and others 1989), **resilience** to disturbance, and **resistance to invasives** (Miller and others

2000; Chambers and others 2007, in review). The three most common **soil temperature regimes** in the region are mesic (warm), frigid (cool), and cryic (cold), which are classified by elevation and indicator species. However, the elevation at which these regimes occur varies from one **MLRA** to another.

For example, in the Malheur High Plateau MLRA (fig. 2), the split between warm (mesic) and cool (frigid) soils is typically 4000 ft (+ 500 ft) (1200 + 150 m) depending on aspect (e.g., south aspect 4500 ft [1370 m] and north aspect 3500 ft [1067 m]). On warm soils, Wyoming and basin big sagebrush are the two most common shrubs on moderately deep to deep soils, respectively. Bluebunch wheatgrass typically is the dominant grass on north aspects in these warm (mesic) soils within the 10 to 12 inches (250-300 mm) precipitation zone (**PZ**). On non-slope and south aspects, Thurber's needlegrass, sand dropseed, Indian ricegrass, and needle and thread are often dominant on warm/dry soils. On cool (frigid)/moist (xeric, >12 inches [300 mm]) soil temperature/moisture regimes, the dominant shrub species on moderately deep soils is usually mountain big sagebrush (low sagebrush on shallow or claypan soils) with Idaho fescue (usually dominant on the cooler aspects) and bluebunch wheatgrass (dominant on the warmer aspects) in the understory. Cold (cryic) soils occur at >6000 ft (>1830 m). Mountain big sagebrush (low sagebrush on shallow soils) is usually dominant on these soils with the sub to dominant shrubs snowberry, serviceberry, and bitterbrush.

In contrast, the elevation boundary between warm (mesic) and cool (frigid) soil temperature regimes in the Central Nevada Basin and Range and the Great Salt Lake Basin MLRAs (MLRAs 28A and B in fig. 2) occurs at a higher elevation (between 5500 and 6500 ft [1675 and 1980 m]). Dominant shrubs in the warm soil regime are often black sagebrush and Wyoming big sagebrush. Cool (frigid) soils are split into two groups and mapped as warm-cool or cool-cool between 6000 and 8000 ft (1830 to 2440 m) with black, low, or big sagebrush as the common dominant species. Cool-cool (cool-frigid) soils are mapped between 7500 and 8200 ft (2200 to 2500 m) with mountain big sagebrush, singleleaf pinyon, and Utah juniper as the common overstory species. Cold (cryic) soils, mapped above 8200 ft (2500 m), are often occupied by curl-leaf mountain mahogany, white and grand fir, limber pine, lodgepole, and white bark pine intermingling with mountain big and low sagebrush.

The dominant **soil moisture regimes** mapped within the Great Basin Region are dry (aridic) (<12 inches [300 mm] PZ) and moist (xeric) (>12 inches [300mm] PZ). Soil moisture and temperature regimes are important characteristics that affect ecological site resilience to disturbance and resistance to invasives. For the purpose of this synthesis, we will use the terms warm, cool, and cold for the soil temperature-regime classification terms mesic, frigid, and cryic, and the terms dry and moist for the soil moisture-regime classification terms aridic and xeric. Sites noted in the literature with an aridic-xeric moisture regime (dry-moist) were interpreted to usually fall within the 10 to 12 inches (250-300 mm) PZ. Here, warm/dry and cool/moist are equivalent to mesic/aridic and frigid/xeric, respectively.

Vegetation

This region encompasses a large portion of the sagebrush semi-desert, salt-desert, and intermountain sagebrush steppe, which intermingles with open stands and woodlands of pinyon and juniper described by West (1983a) and sub-alpine forest at the highest elevation. Although species composition changes to some degree across the region and across **MLRA**s, there are many species that occur across most of the region. Species composition

usually changes more dramatically with elevation and landscape position (basins, foothills, and mountains) than from one MLRA to the next. The transitions between general vegetation zones associated with changes in elevation and amounts of available moisture are:

(1) desert basins with <8 inches (200 mm) **PZ**;

(2) sagebrush semi-desert with 8 to 12 inches (200-300 mm) PZ;

(3) upland sagebrush and pinyon-juniper with 12 to 16 inches (300-400 mm) PZ;

(4) mountain brush with >16 inches (410 mm) PZ; and

(5) sub-alpine forest with >20 inches (510 mm) PZ (which is not covered in this review).

Common diagnostic species of these vegetation zones include:

(1a) Desert shrub <6 inches (150 mm) PZ on non-sodic or non-salty soils—spiny hopsage, winterfat, bud sagebrush, shadscale, Nuttall's saltbush, Indian ricegrass, James' galleta, and bottlebrush squirreltail (fig. 7A);

(1b) Desert shrub (salt-desert shrub) <8 inches (200 mm) PZ on salty or sodic soils (typically in the bottoms of ancient pluvial lakes)—greasewood, shadscale, basin wildrye, saltgrass, alkali cordgrass, and alkali sacaton (fig. 7B);

(2) Sagebrush semi-desert 8 to 12 inches (200-300 mm) PZ—Wyoming big sagebrush, basin big sagebrush, low sagebrush (shallow or clayey sub-soils), black sagebrush (shallow to an indurated duripan) (fig. 8), rubber (gray) and green rabbitbrush, bitterbrush, bluebunch wheatgrass (only on north aspects in the southern portion of the region), Sandberg bluegrass, Thurber's needlegrass, needle and thread grass, and Indian ricegrass (fig. 9);

(3) Upland sagebrush and pinyon-juniper 12 to 16 inches (300-410 mm) PZ—mountain big sagebrush, low sagebrush, bitterbrush, western juniper, Utah juniper, singleleaf pinyon, Idaho fescue, bluebunch wheatgrass, Thurber's needlegrass, Columbia needlegrass, Lettermen's needlegrass, western needlegrass, pine needlegrass, June grass, Sandberg bluegrass, and low elevation aspen (north facing snow drift pockets);

(4) Mountain brush >16 inches (400 mm) PZ—mountain big sagebrush, bitterbrush, snowberry, serviceberry, curl-leaf 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 (fig. 10);
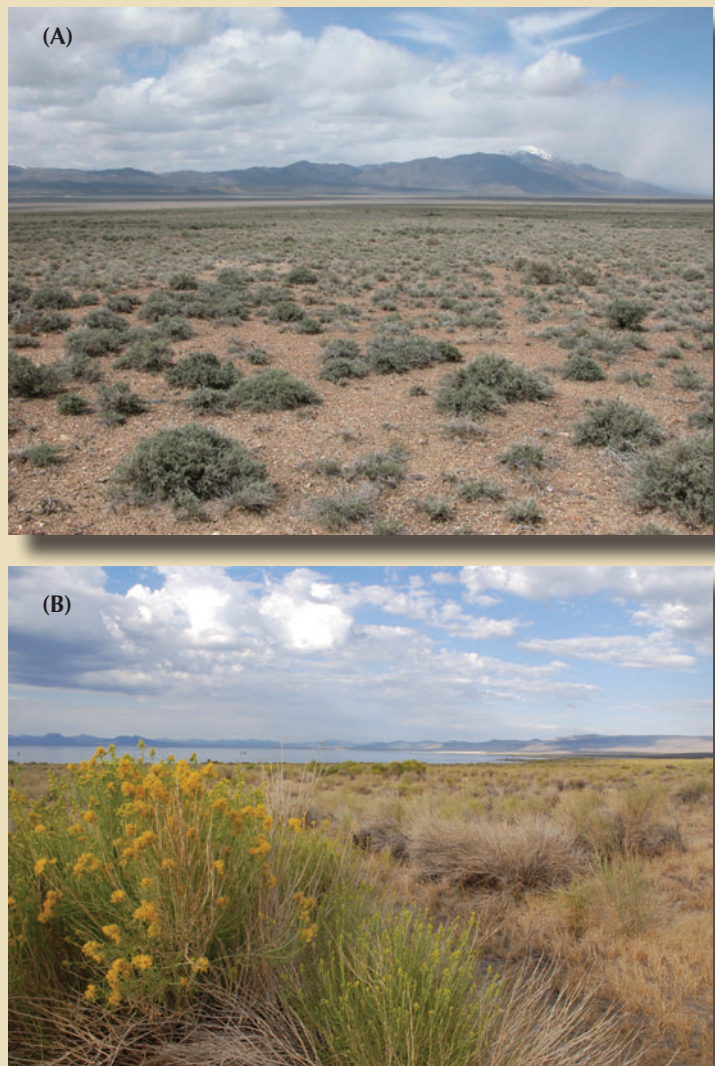
Figure 7. (A) Desert shrub dominated by bud sagebrush on a cool/dry (frigid/aridic) site (White Pine Co., Nevada). (B) Salt desert community dominated by alkali grey rabbitbrush, greasewood, and saltgrass (Mono Lake, California). (Photos by Rick Miller.)
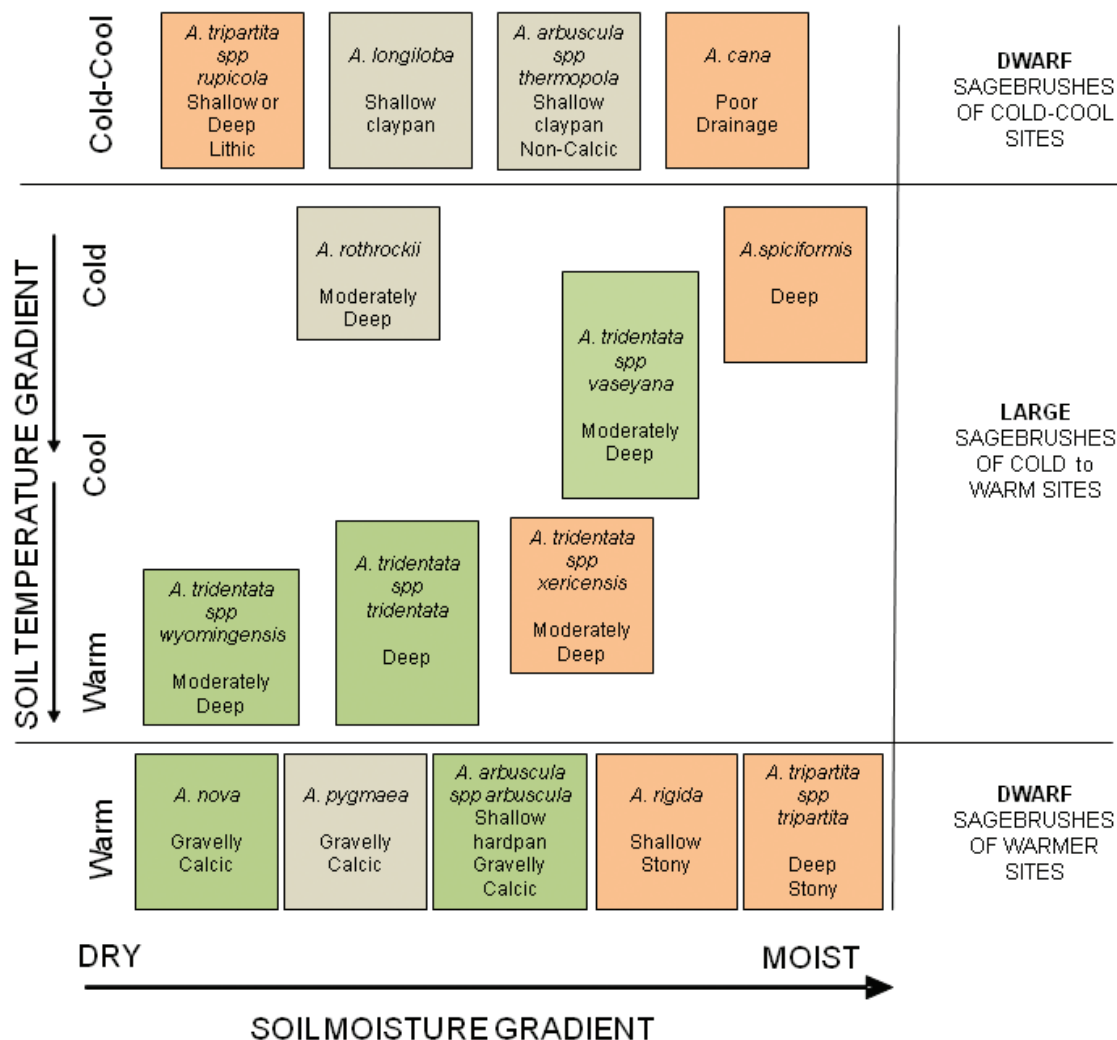
Figure 8. Major sagebrush taxa in the Great Basin and Columbia Basin positioned along gradients of soil temperature and soil moisture (adapted from Robertson and others 1966; McArthur 1983; West 1983; West and Young 2000; Rosentreter 2004; Shultz 2009, 2012). Key soil characteristics associated with each species are also included. Colors represents importance based on relative abundance among the sagebrush species and subspecies; tan = scarce, orange = common, and green = dominant species in the region.

(5) High elevation forest >20 inches (510 cm) PZ—Douglas-fir, whitebark pine, limber pine, ponderosa pine, lodgepole pine, white fir, grand fir, aspen, Sierra juniper, and bristle cone pine. Although these vegetation zones are closely associated with elevation, the elevation at which they occur changes with region (MLRA) and aspect (fig. 10).

An understanding of soil characteristics (e.g., soil temperature/moisture regimes) and species composition and structure are often important for evaluating site productivity, **resilience** to disturbance, and **resistance to invasive species** (as suggested by Passey and Hugie 1962, 1963a). Species can act as indicators of a land unit or ecological site characteristics such as soil temperature and moisture regimes. One of the most prominent examples in the region is the variation in ecological site characteristics and differences in resilience and resistance to invasives among the big sagebrush subspecies—Wyoming, mountain, and basin big sagebrush (figs. 5, 8). A second example is the subtle change in species composition from a bluebunch wheatgrass-Idaho fescue community to an Idaho fescue dominated community, which indicates a shift toward cooler soil temperatures (Daubenmire 1972). In the absence of diagnostic species on severely degraded sites, soil map unit components and their associated **ecological sites** can help provide the necessary clues.

Figure 9. Wyoming big sagebrush communities usually occupy warm/dry (mesic/aridic) sites at lower elevations and have lower recovery potential (resilience) and resistance to invasive species like cheatgrass than cool/moist (frigid/xeric) mountain big sagebrush communities at higher elevations. As a result, the outcome of fire versus mechanical treatments may differ over these elevation gradients. Mechanical treatments are often less severe than burn treatments and can result in significantly smaller increases in exotic species at lower elevations (Monitor Range, central Nevada; photo by Jeanne Chambers).

Figure 10. A high elevation (8000-10,000 ft [2400-3000 m]) cold/moist (cryic/xeric) mountain big sagebrush community intermingling with white bark pine and limber pine. Cold temperatures result in high resistance to invasive species; however, the harsh environmental conditions and short growing season can result in low resilience (White Mountains, California/Nevada; photo by Rick Miller).

Disturbance, Resilience, Resistance, and State-and-Transition

Disturbance

Disturbance is defined as any relatively discrete event in time that disrupts ecosystem, community, or population composition and structure and changes resources, substrate availability, or the physical environment (Pickett and White 1985: 7). Fire, a common disturbance in the Great Basin Region, removes large amounts of biomass; alters plant species composition and structure; dramatically changes nutrient, energy, and water cycles (at least in the short term); and alters the microenvironment. In addition, the short- and long-term impacts of fire are also influenced by other disturbances that have occurred in the past, are ongoing, or will occur following the fire event. Disturbances that commonly interact with fire are: herbivory by livestock, large wild ungulates, small mammals and insects; fungi (snow mold, etc.); and diseases; all of which also interact with climate (particularly climatic extremes) (Evers 2010; Evers and others 2013a, 2013b). However, very little research has evaluated the interaction of fire with other disturbances on vegetation. The potential combined impacts of fire and other disturbances can be clarified by carefully developing and answering questions that relate to:

- What past and present disturbances are impacting the site to be treated?
- Have any of these disturbances increased the probability of a plant community or **phase** crossing an ecological **threshold** (at-risk community or phase) to another steady **state**?
- How do these disturbances interact with fire?
- Are climate trends influencing both fire risk and recovery potentials of the ecological site?

To evaluate the potential interaction of these disturbances with fire, the following two questions can be addressed:

- How is the secondary disturbance(s) influencing current composition and structure of vegetation, resilience, and resistance to invasives of the ecological site before and after a fire event?
- How does the interaction influence **fire severity** (often confused with **fire intensity**)?

Resilience to Disturbance and Resistance to Invasive Species

An ecological site's resilience to disturbance and resistance to invasives are measures of its ability to recover from disturbance and stay within its **historic range of variability**. We define *resilience* of arid and semi-arid ecosystems as the capacity of an ecosystem to regain its fundamental structure, processes, and functioning when subjected to stressors or disturbances such as drought, herbivory, or wildfire (e.g., Holling 1973; Walker and others 1999; Allen and others 2005; Groffman and others 2006; Chapin and others 2010). In this context, resilience is a function of the underlying ecosystem characteristics and processes that determine ecosystem recovery rather than the amount or magnitude of stress or disturbance that an ecosystem can withstand before changes in characteristics and processes result in new, alternative states (e.g., Gunderson 2000; Carpenter and others 2001; Briske and others 2008). Simply stated, resilience is the ability of an ecological site to recover, which enables managers to use it in a management context to evaluate and predict potential outcomes of either planned or unplanned disturbances. *Resistance* is commonly defined as the capacity of an ecosystem to retain its fundamental structure, processes, and functioning (or remain largely unchanged) despite stressors or disturbances (e.g., Folke and others 2004). Because of the ever-increasing effects of invasive species on ecosystems and management options, invasive species ecologists have been using the term resistance in the context of species

invasions for over a decade. Here, we adopt that interpretation, using resistance in a more narrow focus related to invasive species. We define *resistance to invasion* as the abiotic and biotic characteristics and ecological processes of an ecosystem that limit the population growth of an invading species (modified from D'Antonio and Thomsen 2004; Chambers and others in press).

Ecological Sites, States, Phases, and State-and-Transition

Plant community dynamics are reflected through changes in plant community composition and structure resulting from disturbance and variation in climate. Ecological site descriptions (ESDs) attempt to depict the variation in plant community dynamics from natural disturbances for a specific land area using state and transition models (fig. 11). An *ecological site* is a unit of land with specific physical characteristics that differs from other units of land in its ability to produce distinctive kinds and amounts of vegetation and its response to management and disturbance. Ecological sites have specific characteristics that determine the level of resilience to disturbance and resistance to invasives (fig. 12). A *state* is defined as a suite of plant community successional (**seral**) stages, called phases, that occur on similar soils and climate and produce resistant functional and structural characteristics with a characteristic range of variability such that they are maintained through self-repair mechanisms (Bestelmeyer and others 2003, 2009; Stringham and others 2003; Briske and others 2008). The reference state describes the historical range of variability of an ecological site (figs. 11, 13). *Phase* is a plant community within a state that is predicted to replace other communities along traditional succession-retrogression pathways. Succession from one community phase to the next within a state is readily reversible over short time periods (years to decades) without management intervention because the phases are not separated by **thresholds**. However, an **at-risk phase** is unstable as a result of a decline in resilience and is vulnerable to threshold crossing (changes that trigger a new plant phase in a different state) and subsequent transition to an alternative state. The change from one state to another may occur rapidly with a single disturbance, particularly if the phase is at-risk, or gradually with a chronic single disturbance or with multiple disturbances over a long period of time. Figure 11 represents a generalized state and transition model for both a warm and cool sagebrush-bunchgrass community in which fire is a major disturbance factor. Within the reference state, fire simply shifts the community phase toward bunchgrass dominance. However, if past and/or current disturbances have resulted in an at-risk phase (e.g., depleted perennial grasses and forbs, increase in invasive species, and/or change in fuel structure and biomass resulting in an increase in **fire severity** or frequency), fire can be the trigger that results in the crossing of a threshold to a new steady-state. The resilience and resistance of the reference state is a function of the characteristics of the ecological site and current vegetation composition and structure (phase) (fig. 12). Since climate is one of the primary components of an ecological site, changes in climate will impact soil moisture and temperature regimes, resulting in changes in resilience and resistance. In addition, climate change will impact potential vegetation, biomass, and fire weather patterns, which are key components in driving **fire regimes**.

Interactions with Global Change Factors

Changes in global climate will potentially be a major disrupter to Great Basin Region plant communities. Climate is a primary component of **ecological sites** that influences thermal regimes, water availability, soil processes, nutrient availability, potential vegetation, **resilience** and **resistance to invasives** (fig. 12). Climate change models predict more variable and severe weather events, higher temperature, drier summer soil conditions, and wetter winter seasons for the mid-latitude-arid regions (Schlesinger and others 1990; Schneider
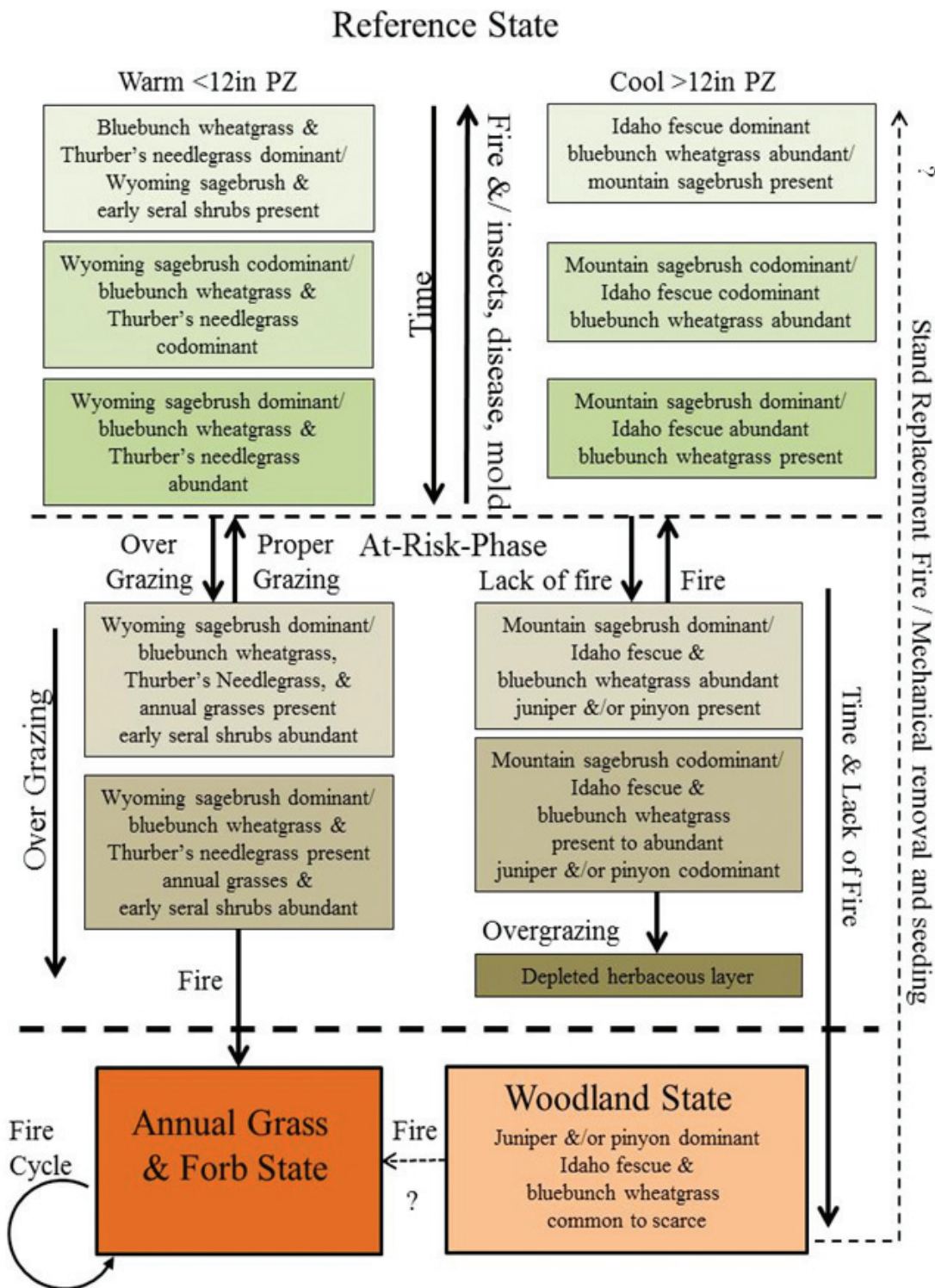
Figure 11. State-and-transition model for the warm/dry (mesic/aridic) loamy 10 to 12 PZ and cool/moist (frigid/xeric) loamy 12 to 16 PZ ecological site types. Overgrazing on the warm/dry site will result in a depleted understory and an increase in sagebrush and cheatgrass. Fire acts as the trigger to move the at-risk-phase to the annual grass and forb state. Return to the reference state will take one to several treatments of herbicides and seeding. In the cool/moist loamy 12 to 16 inch PZ (30-40 cm), length of fire-return intervals will be the primary variable that determines succession towards woodland if a seed source is available. Shrubs will decline as a result of tree competition. On sites with moderately deep to deep soils, depletion of understory herbs is primarily a result of overgrazing. However, on ecological sites with soil retrictive layers at around 16 inches (40 cm) or less, the herbaceous layer will decline regardles of grazing due to competion from the trees. A stand replacement fire may or may not return the woodland to its historic state. This will depend on fire severity, post-fire density of deep-rooted perennial grasses, and site characterisics, especially the soil thermal regime.
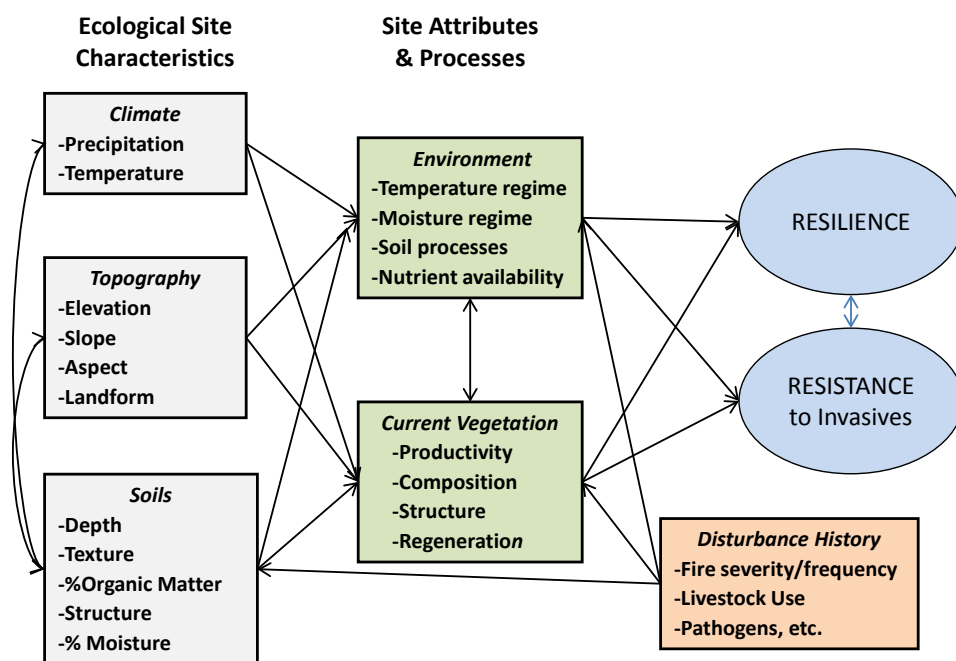
Figure 12. Ecological site characteristics (on the left) are the primary drivers that influence soil temperature/moisture regimes and potential vegetation. These regimes are identified in soil maps as mesic, frigid, cryic; aridic, xeric (warm, cool, cold; dry, moist). Potential vegetation + disturbance history + time since disturbance = current vegetation. If all components are intact on a site, then levels of resilience to disturbance and resistance to invasives are near full potential. However, if the site is not at potential because several components are below potential or missing, e.g., perennial grasses are severely depleted or invasive species are abundant, resilience and/or resistance levels will be lower than potential. Adapted from Chambers and others in press.

Figure 13. This Wyoming big sagebrush/Thurber's needlegrass community is an example of a late seral community in a reference state (non-degraded). The site is dry (8 to 10 inch [200 to 250 mm] PZ) and at the transition from a warm to cool (mesic to frigid) soil thermal regime (northwestern Nevada; photo by Rick Miller).

1993; Christensen and others 2007; IPCC 2007). These potential changes will alter fire weather and fuels, plant composition, competition between native and invasive species, and soil moisture and thermal regimes, resulting in large changes in the interaction between fire and vegetation. The increase in the length of wildfire seasons around the year 2000 may be partially attributed to the sudden increase in large wildfires in forests (Westerling and others 2006) and rangelands (Miller and others 2011) in the West. However, other anthropogenic factors also have likely contributed to increased wildfire occurrence and size, including recreation (e.g., all-terrain-vehicles), changes in fuel characteristics (e.g., introduced annual grasses), and wilderness designations, which limit fire suppression.

Climate models predict the loss of 33,600 mi^2 (87,000 km^2) of sagebrush habitat with each 1.8 °F (1 °C) increase in temperature (Neilson and others 2005). Increased temperature also may exert stresses on sagebrush through increased annual transpiration rates coupled with changes in precipitation patterns resulting in increased loss of soil moisture (Shafer and others 2001). Increased stress will decrease resilience to disturbance and resistance to invasives. As the climate warms, lower elevations may no longer have the environmental characteristics to support persistent populations of cheatgrass while higher elevation communities may become less resistant to invasion as they become increasingly climatically suitable (Bradley and others 2009). However, at the lower elevations, more drought-tolerant introduced species such as red brome may invade.

Increased levels of atmospheric carbon dioxide (CO_2) have also contributed to changes in plant composition in the region. Atmospheric CO_2 has increased from preindustrial levels of 280 µmol/mol to 2012 levels of 394 µmol/mol and are projected to reach 420 µmol/mol in 2020 (Bazzaz and others 1996; Ziska and others 2005; NOAA 2012). The continual increase in greenhouse gasses is predicted to alter global climatic regimes. Climatic models predict that average temperatures may warm by as much as 5 °F (2.8 °C) to >11 °F (>6 °C) by the mid-21st Century for the Great Basin and much of the adjacent areas that support sagebrush (Bachelet and others 2001; Neilson and others 2005; Christensen and others 2007). Temperature decreases with rising elevation at about 5.5 °F/1000 ft (or ~1 °C/100 m) based on the adiabatic lapse rates. In the Great Basin Region, elevation is a primary driver that influences soil moisture and temperature regimes. Increasing temperatures will shift soil thermal regimes at the mid to higher elevations from cool (frigid) to warm (mesic), which will significantly reduce site resilience to disturbance and resistance to invasive species (fig. 14). Based on current criteria for soil mapping, a 5.5 °F (3 °C) rise in temperature would change the boundary in elevation that separates warm (mesic) and cool (frigid) soils from 4000 to 5000 ft (1220 to 1525 m) in the Malheur High Plateau **MLRA** (fig. 2). In the Great Salt Lake and Central Nevada Basin and Range MLRAs (fig. 2), the warmer-cool (warm-frigid) soils will shift from 6000 to 7000 ft (1830 to 2135 m) and from 7500 to 8500 ft (2290 to 2690 m) for the cooler-cool (cool-frigid) soils. As soil temperature/moisture regime boundaries increase in elevation, the proportion of available land area falling into the cooler regimes declines.

Increases in temperatures and CO_2 levels are predicted to facilitate the spread of invasive annuals across the Great Basin Region resulting in an increase in frequency and size of reoccurring fires (D'Antonio and Vitosek 1992; D'Antonio 2000; Ziska and others 2005; Westerling and others 2006). Cheatgrass responds more positively to elevated CO_2 than many native, perennial grasses (Smith and others 1987). Decreased precipitation, particularly summer precipitation, in areas that currently receive a high proportion of summer precipitation (which is a small portion of the Great Basin Region) may increase suitable areas for cheatgrass expansion by 45% (Bradley 2009). However, cheatgrass biomass and cover will likely become more sporadic at the lower elevations (Bradley and others 2009) as a result of decreasing amounts of available moisture and increasing variability of precipitation.

Summary

Possibly the first documented impacts of global change in the Great Basin Region have been: (1) an increase in competition from some invasives due to elevated CO_2 levels; and (2) a lengthening of the fire season and increase in large fires. Increasing temperatures will alter the soil temperature/moisture regimes of cooler-wetter sagebrush communities at mid and upper elevations, lowering their resilience to disturbance and resistance to invasives.
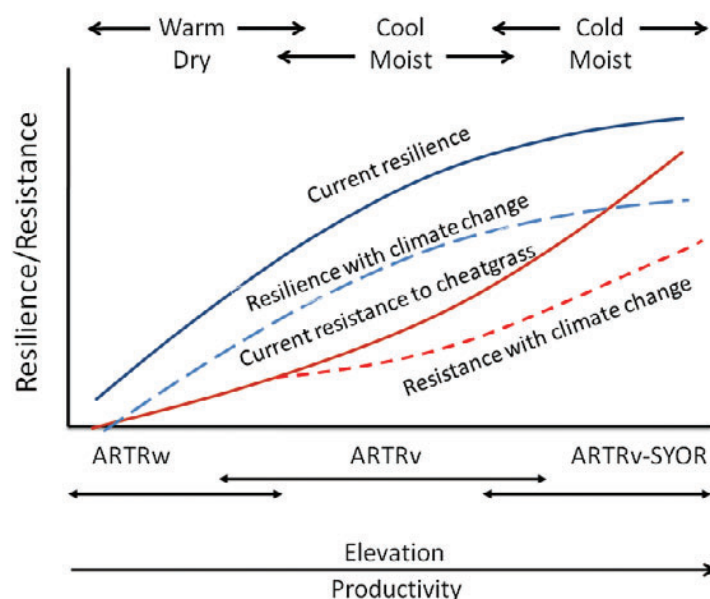
Figure 14. The relative resilience (solid blue line) of Great Basin ecosystem types over an elevation/productivity gradient and the potential shift that will occur with increasing temperatures (dashed blue line) as a result of climate change. Resistance to cheatgrass of those same ecosystem types along a gradient representing current conditions (solid red line) and potential decline with climate change (dashed red line) (modified from Chambers and others in press). Although lower elevation systems may become more resistant to cheatgrass due to drier conditions (Bradley and others 2009), climate shift may cause other invasive species such as red brome and ripgut brome to shift northward in the Great Basin Region.

The literature indicates that resource managers will be confronted with an increasing invasive weed problem resulting from increasing temperatures throughout the Great Basin Region. Although a decrease in climate suitability for common invaders such as cheatgrass at lower elevations may result in restoration opportunities (Bradley and others 2009), it may also result in the invasion of more drought-tolerant genotypes of cheatgrass (Merrill and others 2012) or other exotic species.

Evaluating Ecological Site Characteristics: Their Role in Influencing Resilience to Disturbance and Resistance to Invasive Species

Ecological site characteristics that influence soil moisture and thermal regimes and vegetation composition and structure are key components in determining ecological site **resilience to disturbance** and **resistance to invasives** (fig. 12). These biotic and abiotic ecological site characteristics operate across environmental gradients (fig. 5) at multiple scales ranging from **MLRA**s to plant communities (Chambers and others in press). These gradients are typically associated with changes in productivity and can be related to dominant plant functional groups (West 1983b; Burke 1989; Whitford 2002; Davies and others 2009b; Brooks and Chambers 2011; Chambers and others in press). Differences in the responses of landscapes and ecological sites to disturbance are predictable based on their abiotic and biotic characteristics. These are often highly dependent on local conditions such as aspect, slope, and soils (e.g., texture and depth). Topographic features, including elevation, slope, aspect, and landform are dominant features that affect environmental gradients and influence temperature, effective precipitation, and soils (figs. 5, 12). As temperatures become cooler and moisture increases along these gradients, ecological site resilience and resistance usually increases (Koniak 1983, 1985; West 1983b; Everett and Sharrow 1985a; Chambers and others 2007; Miller and others in review). However, as temperatures decline to levels supporting alpine ecosystems, resilience also declines, largely due to a shortened growing season. At the dry end of the environmental gradient that supports salt-deserts and desert shrub communities, soil water availability becomes limiting, which can limit reproduction and establishment of invasive species.

Landscape Sub-Units and Scale

A high degree of heterogeneity exists across the Great Basin Region. To evaluate the applicability of research findings and observed management responses, and to determine differences in resilience to disturbance and resistance to invasives from one location to the next, it is helpful to separate this large region into sub-units. Sub-units used in this synthesis are MLRAs, ecological sites (USDA-NRCS 2011), and their associated plant communities.

Major Land Resource Areas (MLRAs)

MLRAs are geographically associated land resource units that are characterized by particular patterns of soils, geology, climate, water resources, and land use (table 1; fig. 2). These units may be one continuous area or several separate nearby areas. Vegetation differences among MLRAs in the Great Basin Region are not always clear. For example, Wyoming and mountain big sagebrush occur across the High Malheur Plateau, Central Nevada Basin and Range, and Great Salt Lake MLRAs. However, western juniper only occurs in the High Malheur Plateau, and Utah juniper and singleleaf pinyon only occur in the Central Nevada Basin and Range and Great Salt Lake regions. MLRAs are further subdivided by ecological sites. When evaluating the applicability of an observed response from either a management action or research project from one location to another, either within or among MLRAs, it is important to evaluate the characteristics of the two areas (fig. 4). These include soil temperature/moisture regimes and the differences in plant species composition and their potential responses to disturbance. Ecological site response to disturbance will be more strongly related to environmental gradients and ecological site characteristics than to specific MLRAs. For example, although parent materials may differ between two ecological sites located in two different MLRAs, characteristics that influence water capture and storage are usually more important to compare when evaluating ecological site resilience. **Soil temperature regimes** are also important characteristics to compare, although they may occur at different elevations among MLRAs.

Ecological Sites

Ecological sites are both described and used only within boundaries of the appropriate MLRA, although very similar ecological sites may occur in other MLRAs in the Great Basin Region. Ecological site descriptions provide important information about soils, topographic position, climate, and potential vegetation that influence resilience and resistance to invasion of a site (fig. 12). These subdivisions also describe plant community dynamics, including processes, actions, and events, that can result in **thresholds** being crossed and plant communities (**phases**) changing to undesired alternative **states** (fig. 11).

Plant Community (Phase)

Plant community or phase reflects the current vegetation (**seral** stage) on the ecological site and is a function of climate, soils, topography, land use, disturbance history, and available and adapted organisms. Plant communities may grade from one to another through succession and retrogression.

Section 2: Predicting Outcomes—Asking the Right Questions

Predicting short- and long-term ecological response and successional trajectories following disturbance is not an exact science but a game of probabilities. However, asking the right questions can significantly increase the probability of closely estimating the short- and long-term plant successional patterns. Key questions are associated with the primary variables driving succession:

- What are the characteristics of the ecological site, including current vegetation?
- What is, or will likely be, the severity of disturbance(s)?
- What was the pre-disturbance weather, and how will various scenarios of post-treatment weather influence ecological site recovery?
- What is the disturbance history of the area to be treated?
- What will the post-fire disturbance scenarios likely be (e.g., drought, wild or insect herbivory, or livestock grazing) (fig. 15)?

Each variable is discussed in more detail below. Addressing these questions requires a combination of assimilating the available science, personal experience, and observations of others. Section 3 summarizes the available science, and Section 5 presents a framework of questions and considerations for predicting plant successional patterns, identifying risks, and selecting the appropriate ecological sites.

Ecological Site Characteristics, Ecological Condition, and Disturbance

Characteristics of an ecological site (e.g., topography, climate, and vegetation, or fuels) influence how a fire will burn (figs. 15, 16), the potential **fire regime**(s), ecological site **resilience**, **resistance to invasives**, and the potential vegetation (figs. 5, 12, 13). Moisture availability,
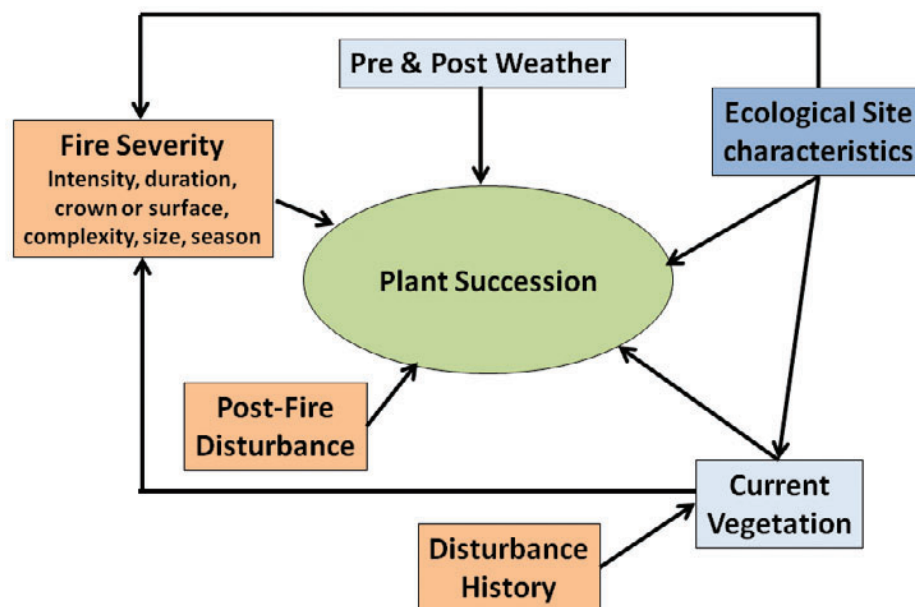
Figure 15. A simple conceptual model of the key components that drive plant successional trajectories following fire. These components are the basis for a series of questions to be addressed when predicting site response and plant succession and evaluating the potential of the site to recover (similar to the model for resilience to disturbance and resistance to invasives; see fig. 43).
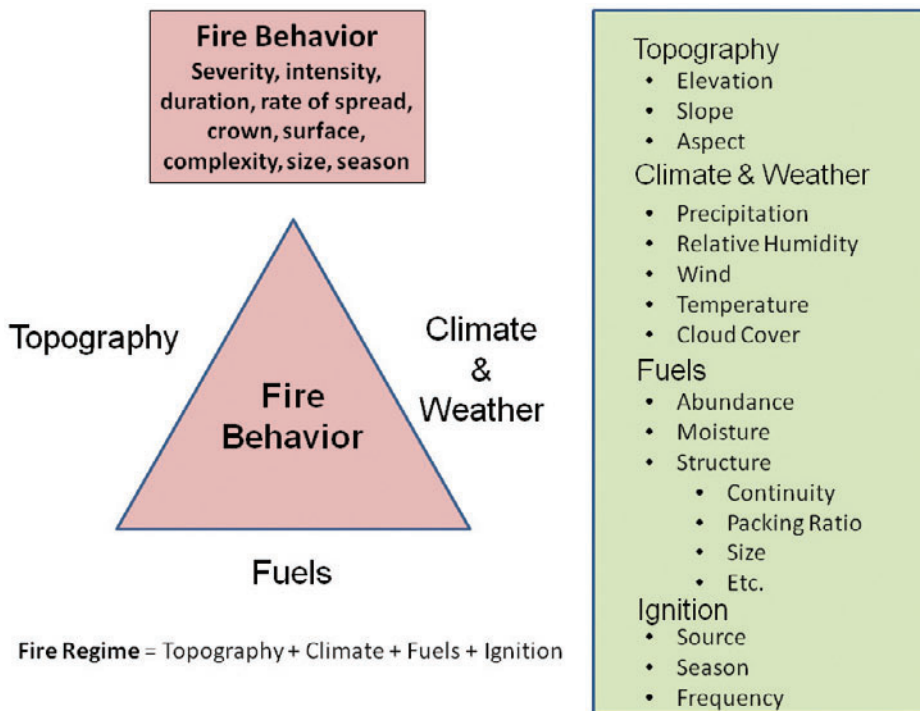
Figure 16. Fire behavior triangle. Variables that influence fire behavior are similar to ecological site characteristics and influence a site's fire regime.

temperature, and current vegetation are major characteristics in determining how an ecological site will respond following fire (all three are addressed in Section 3) (figs. 17A and B). Current vegetation is a function of an ecological site's biological, physical, and chemical characteristics that facilitate development of the potential vegetation and the disturbance history (human and non-human) that has been imposed upon the ecological site. Characteristics of current vegetation determine fuel characteristics and reflect the relative abundance of native and non-native species, composition of fire tolerant and non-tolerant species (related to life histories and morphology), and how the ecological site is functioning. These parameters will influence ecological site response to future disturbance events or changes in ecological processes. Two important questions to address regarding the ecological site to be treated by fire or a fire-surrogate are: (1) what do the ecological site characteristics indicate about the resilience to disturbance and resistance to invasives, and (2) is current vegetation at risk of transition to another steady-**state**? Post-fire disturbances and interactions will also influence successional trajectories. For example, drought alone may only influence the annual dynamics and rate of recovery of a community but not the successional trajectory. However, the combination of drought and repeated heavy grazing during the growing season may redirect the successional trajectory (Evers and others 2013b).

Fire Severity

Important questions to address about the characteristics of a fire (wild or prescribed) relate to *fire severity*, which is defined as the effects of fire on ecological processes, soil, flora, and fauna, or the degree to which an ecological site has been altered or disrupted by fire. Fire severity can have a large effect on ecological site resilience to disturbance and resistance to invasives through its influences on the mortality or vigor of desirable species, soil biota, erosion, and resource availability (e.g., water and nitrogen). The characteristics of a fire that influence severity are fire intensity (the amount of energy released), duration or residence time, complexity (patchiness), size, time of year, and time since the last fire event (interval). These characteristics are driven by topography, weather, and fuels (fig. 16), which are closely associated with ecological site characteristics (vegetation, topography,

and climate). Evaluating the characteristics in fig. 16 is important for evaluating potential fire severity on an ecological site that will receive a prescribed burn. Evaluating post-fire severity requires measurements and observations collected during and after the fire. Accurate measurements during a fire (e.g., intensity, rate of spread, residence-time, and soil temperatures at the surface and subsurface) are usually limited or non-existent. However, even in the absence of measurements collected during a fire event, knowledge of weather conditions during the event and on-site observations or measurements taken immediately following the event (table 2) enable managers to attain some estimate of fire severity on vegetation (fig. 18). Observations include the amount of foliage and fuel size diameter classes consumed (1 hr = fuels with a diameter <0.25 inch [6.4 mm]; 10 hr = 0.25 to 1 inch diameter [6.4 to 25 mm]; and 100 hr = 1 to 3 inches diameter [25 to 76 mm]). For example, different levels of fire severity are indicated by:

(1) percentage of tree canopy foliage killed but not consumed;

(2) percentage of foliage and fine branches [<0.25 inch; 6.4 mm] consumed;

(3) percentage of tree bark consumed;

(4) proportion of sagebrush skeletons remaining compared to main stems burned to or below ground level;

(5) patchiness or complexity of the burn and spatial arrangement of fuels consumed across the landscape; and

(6) amount of stubble remaining on perennial bunchgrasses (table 2).

Pre- and Post-Fire Weather

Pre-fire weather conditions one to three years prior to the event influence fuels (biomass of live and dead, moisture, and structure), seed banks, and general vigor of plant species. Post-fire weather, an unknown at the time of the event, plays a key role in the growth of surviving plants, plant establishment, and reproductive effort. Although post-treatment weather conditions are unknown when planning a prescribed burn, it is important to ask: how will vegetation respond under relatively dry, average, and wet conditions? Post-fire weather may interact with other post-fire disturbances to increase the severity of the disturbance (e.g., drought + overgrazing and drought + insect herbivory).

Figure 17. (A) A warm/dry Wyoming big sagebrush/cheatgrass site (6 to 8 inch [150 to 200 mm] PZ) that has crossed a threshold to an alternative state. Fire will act as a trigger that will move the site to annual grassland. The ability of the site to recover without assistance is low to non-existent (east slope of the Pueblo Mountains, southeast Oregon). **(B)** A Utah juniper-singleleaf pinyon pine site that occupies a cool/moist (frigid/xeric) regime with a depleted but persistent stand of deep-rooted perennial grasses. This site is relatively resistant to invasive species, so potential recovery to a late seral non-degraded state should be good in the absence of inappropriate livestock grazing. Deferment of grazing following fire on this site should be longer than on a similar site with a less degraded understory (central Nevada). (Photos by Rick Miller.)

Table 2. Post-fire indicators of severity that can be evaluated following a fire. Soil and litter indicators are derived from Parson and others (2010).

Low severity	Moderate severity	High severity
>75% burned sagebrush skeletons remaining	15-75% burned sagebrush skeletons remaining	Sagebrush basal stumps remain or burned below the soil surface
<25% tree foliage dead, <15% foliage consumption	25-75% tree foliage dead, 15-50% foliage consumed	>75% tree foliage dead, >50% consumed
Tree duff blackened but little consumed	Majority of tree duff consumed surface blackened	White ash layer beneath tree canopy
>2 inch blackened stubble remains on burned grasses	0.25-1 inch blackened stubble remains on burned grasses	Grass crowns consumed to or below the surface
Unburned patches >50%	Unburned patches 15-50%	Unburned patches <15%
Interspace litter consumption <50%	Interspace litter consumption 50-80%	Interspace litter consumption >80%, white ash deposition
Shrub canopy litter consumption <50%	Shrub canopy litter consumption 50-80%	Shrub canopy litter consumption >80%, white ash deposition
No ash, ground fuels blackened & recognizable	Thin layer of black to gray ash, some litter recognizable	Layer of powdery gray or white ash >90% surface organics consumed
No fire-induced water repellency	Weak to medium water repellency at or just below the surface	Strong water repellency at or below the surface
Surface soil structure unchanged	Surface structure slightly to not altered	Aggregate stability reduced or severely degraded, surface loose and/or powdery

Figure 18. A warm/dry (mesic/aridic) Wyoming big sagebrush site that burned under severe conditions (hot temperature, dry, low humidity, and winds >10 mph). High fire severity indicators illustrated in the picture are sagebrush stumps burned level to and below the soil surface, nearly complete consumption of surface organic matter, absence of unburned patches, and lack of perennial bunchgrass crowns (that either did not exist prior to the fire or were completely consumed). The nearly complete consumption of fuels would result in a high amount of total aboveground nitrogen being lost from the site. Resilience is very low due to a combination of high fire severity and site characteristics, which may have included a severely depleted stand of perennial grasses (Milford Flat fire, Utah; photo by Rick Miller).

Section 3: Plant Response to Fire—Review of the Literature

Short- and long-term plant community response following fire is highly variable across plant communities and **ecological sites** in the Great Basin Region. Ecological response and successional trajectories following fire are a function of **fire severity** and ecological site characteristics, including disturbance history, climate, and vegetation present at the time of the fire, as well as post-fire disturbance and pre- and post-fire weather (fig. 15). The more knowledge about each of these key variables, the higher the probability of accurately predicting short- and long-term plant succession.

Plant communities form through interactions among plant species populations. How individual plant species respond to fire is dictated by the location of each plant's growing points, the susceptibility of these growing points to fire, and the ability of the species to re-establish from seed or resprout following fire (Pyke and others 2010). In a thorough review of the literature, we found a considerable number of studies that report short-term (<five years) vegetation response to fire. However, literature reporting long-term response (>10 years) is limited and often:(1) substitute space for time (multiple fires at different locations of different ages are evaluated instead of measuring vegetation at the same location over time), (2) lack unburned controls, and (3) do not have pre-burn plant composition data. Limitations of studies that substitute space for time include the assumptions that fire severity and ecological site characteristics were similar across the multiple burn sites. In addition, the lack of controls can make it difficult to evaluate potential post-treatment changes in vegetation due to the high degree of variation in annual productivity (e.g., as high as 300%; Passey and Huggie 1963a, 1963b) and cover (Anderson and Inouye 2001) over time. For example, perennial grass cover during two relatively long-term dry and wet periods between 1933 and 1975 fluctuated 13-fold (Anderson and Inouye 2001), and density varied from 0.5 to 6 plants/ft^2 (<5 to 60 plants/m^2). Variation over time is also illustrated by West and Yorks' (2002) work where percent cover of different plant functional groups varied above and below pre-fire cover over a 19-year period. And cheatgrass cover typically varies significantly between wet and dry years (West and Yorks 2002; Bates and others 2007a). Some studies evaluating wildfires used adjacent unburned areas to compare results with burned areas but often lacked the data to compare the sites (e.g., soil depth and texture and macro and micro topography). Only one study (West and Yorks 2002) evaluated the unburned site for similarity (soils, topography, and landscape position). However, studies that substituted space for time and/or lacked pre-fire data from controls can still provide insights to long-term vegetation response to fire.

Plant functional groups addressed in this synthesis are deep-rooted perennial grasses (such as bluebunch wheatgrass and Indian ricegrass), shallow-rooted perennial grasses (Sandberg bluegrass), perennial forbs, annual forbs, invasive annual grasses (primarily cheatgrass), invasive forbs, **biological soil crusts**, shrubs, and trees (singleneedle pinyon, Utah juniper, and western juniper). Ground surface cover groups include bare ground and litter (much of the older literature combined biological soil crusts with bare ground). How these functional groups respond is largely influenced by ecological site characteristics (illustrated in figs. 5 and 12) and the tolerance or avoidance to fire of individual species within each functional group.

Key Questions Related to Plant Response and Ecological Site Recovery

To predict successional trajectories when planning a prescribed burn or following a wild-fire, key questions to address are:

- How will pre-fire plant species composition be related to ecological site recovery?
- What are the short- (<5 years) and long-term (>10 years) responses of plant functional groups, species, litter, and bare ground following fire?
- What **ecological site** characteristics influence plant response, and how do they relate to **resilience** and **resistance to invasives**?
- How does **fire severity** influence post-treatment response?
- How does fire impact soil seed banks, and what role do seed banks have in post-fire recovery?
- How does post-fire grazing influence ecological site recovery?

These questions are addressed in the following sections.

Pre-Fire Species Composition

In the early 1950s, Egler (1954) was one of the early ecologists to recognize the importance of the composition and structure of pre-disturbance vegetation on the immediate post-disturbance composition and subsequent secondary succession (fig. 19). In the Great Basin, pre-disturbance plant species composition has been reported to be a good indicator of post-disturbance plant succession (Everett and Sharrow 1985a). Initial plant composition following fire or fire surrogate treatments is largely influenced by pre-treatment floristics. For example, early post-treatment vegetation will usually be dominated by native perennial grasses and forbs if they were the ecologically dominant species present prior to treatment (Bunting 1985; Everett and Sharrow 1985a, b; West and Hassan 1985; Bates and others 2000, 2007a, 2011; Baughman and others 2010; Rhodes and others 2010; Miller and others in press). In contrast, plant cover after treatment is likely to be dominated by invasives if

Figure 19. Pre-disturbance plant composition and structure is one of the most important characteristics when evaluating post-treatment plant succession, resilience, and resistance to invasives, especially on warm/dry (mesic/frigid) sites. This warm/dry site was in the early stages of juniper encroachment and had an excellent stand of bluebunch wheatgrass, one of the more fire-tolerant bunchgrasses, and native perennial forbs. Cheatgrass would be a major threat if this site was in an at-risk phase due to a depleted stand of deep-rooted perennial grasses or was disturbed by a high severity fire (Bridge Creek, Oregon; photo by Rick Miller).

native perennial grasses and forbs had low cover prior to treatment and invaders existed on the site (Young and Evans 1978; Koniak 1985; Hosten and West 1994; Young and Miller 1985; Wright and Chambers 2002; Chambers and others 2007, in press; Miller and others in review). However, as illustrated in fig. 15, although pre-treatment composition (labeled as "current vegetation") is a primary predictor of plant succession, an ecological site's ability to recover is also influenced by fire severity, pre- and post-fire weather, soil moisture and temperature regimes, and post-treatment disturbance—all of which affect mortality and recruitment.

Summary

Pre-fire plant composition can have a major influence on post-fire plant succession but is only one of several key variables, all of which can interact with one-another (e.g., fire severity, pre- and post-fire weather, post-fire disturbance, ecological site resilience, and resistance to invasives).

Short- and Long-Term Response of Vegetation

Shrubs

One of the largest changes in plant community structure and composition following fire is the immediate reduction of shrubs. This is largely because growing points for most Great Basin shrubs have no insulation to protect them from the heat of a fire. Re-establishment depends on:

- fire severity, intensity, complexity (fire patchiness), and size;
- composition of fire-tolerant and fire-intolerant species in the aboveground vegetation and soil seed banks prior to the burn;
- composition and abundance of seed sources after the burn;
- post-fire weather;
- and ecological site characteristics (including soil regimes).

In the Great Basin Region, sagebrush and associated shrub species and shrubs occupying desert shrub communities (both sodic and non sodic soils) can be grouped as sprouters, weak sprouters, and non-sprouters (table 3).

Sagebrush

The vast majority of research on the fire ecology of woody *Artemisia* species has been conducted on Wyoming and mountain big sagebrush (fig. 20). Literature is limited on fire effects on other species or subspecies of *Artemisia*, including basin big sagebrush, low (little) sagebrush, and black sagebrush. Sagebrush species in the Great Basin Region, with the exceptions of three-tip, silver, and snowfield (or snowbank) big sagebrush, are easily killed by fire (Blaisdell 1953; Pechanec and Stewart 1954; Wright 1972; Winward 1985). Re-establishment is entirely dependent upon post-fire seed dissemination from living sagebrush plants in unburned patches within the interior of the burn and/or edges adjacent to the burn, or seed that was present in the seed bank prior to the fire. Two important factors that constrain sagebrush establishment after fire are: (1) seed is usually disseminated only short distances from the mother plant; and (2) soil moisture is frequently limiting for successful germination and establishment in the spring. Post-fire re-colonization of big sagebrush subspecies is strongly influenced by seed source, ecological site characteristics, and post-fire weather.

Table 3. Potential response to fire of common shrubs in the Great Basin Region.

Tolerant	Moderately tolerant	Intolerant
Sagebrush steppe		
silver sagebrush (s)	rubber rabbitbrush (s)	low sagebrush (ns)
snowfield sagebrush (s)	three-tip sagebrush (ws)	black sagebrush (ns)
aspen (s)		big sagebrush (ns)
green rabbitbrush (s)		curl-leaf mountain mahogany (ws)
wax current (s)		antelope bitterbrush (ws)
desert gooseberry (s)		Mexican cliffrose (ws)
Woods' rose (s)		broom snakeweed (ws)
mountain snowberry (s)		
horsebrush species (s)		
serviceberry (s)		
Stansbury cliffrose (s)		
desert bitterbrush (s)		
Nevada Mormon tea (s)		
Desert shrub		
greasewood (s)		spiny hopsage (ws)
Torrey's saltbush (s)		budsage (ns)
Gardner's saltbush (s)		shadscale (ns)
		fourwing saltbush (ws)
		winterfat (ws)

s = sprouter; ws = weak sprouter; ns = non-sprouter. Derived from Blaisdell 1953; Mueggler and
  Blaisdell 1958; Nord 1965; Wright 1972; Wright and others 1979; West 1994.

Figure 20. Most of the fire-related research in the Great Basin Region has been conducted on Wyoming and mountain big sagebrush (central Nevada; photo by Rick Miller).

Big sagebrush can produce large seed crops (Goodwin 1956) with relatively high germi-nation rates (Harniss and McDonough 1976). However, re-colonization of big sagebrush from the edges of a burn is slow, potentially taking many years to fully re-colonize the interior of large burns (Mueggler 1956; Johnson and Payne 1968; Ziegenhagen and Miller 2009). Seeds are disseminated primarily by wind—most (90%) fall within 30 ft (9.1 m) of the parent plant and nearly 100% fall within 100 ft (30.5 m) (Blaisdell 1953; Mueggler 1956; Johnson and Payne 1968; Daubenmire 1975; Frischknecht 1978). Seedling mor-tality also can be very high under limited moisture availability in the first few weeks of

emergence (Boltz 1994; Board and others 2011). Seedling emergence is associated with moisture availability as influenced by soil surface texture, herbaceous competition (especially from cheatgrass), micro-topography, seedling year precipitation, exposure, and slope position (Boltz 1994). Wyoming big sagebrush recruitment was non-existent in monotypic stands of cheatgrass but occurred in stands dominated by squirreltail (Booth and others 2003). Failure of Wyoming big sagebrush to establish in cheatgrass stands was attributed to the earlier soil moisture withdrawal than occurred in stands of squirreltail.

Sagebrush soil seed banks can play an important role in the re-colonization of the interiors of large burns. Sagebrush seed matures and disperses in the fall and winter (Young and Evans 1989; Meyer and Monsen 1991; Stevens and others 1996). Wildfires and prescribed burns usually occur in summer and fall, prior to maturity and dissemination of the current year's seed crop; consequently, fires largely destroy the current year's seed crop. As a result of limited dispersal, the source of sagebrush seed is primarily from nearby unburned plants (if they are present) or the soil seed bank from seed crops of previous years. In northwestern Nevada, Ziegenhagen and Miller (2009) reported good establishment of mountain big sagebrush in the first two post-fire years (0.25 plants/10 ft^2; 0.25 plants/m^2) within the interior of a 10,000-acre (4047-ha) wildfire where surviving shrubs were rare to non-existent. In the absence of nearby plants (>300 ft [91.5 m]), the authors assumed the seed source was from the soil seed bank. The authors concluded that the presence of sagebrush seed in the soil seed bank and establishment conditions in the first and second years following fire are key variables in determining the rate of long-term recovery from large fires. Wijayratne and Pyke (2012) reported that 40 to 60% of Wyoming and mountain big sagebrush seed remained viable for two years when covered with a thin layer of soil. However, if buried more than a few millimeters, the small sagebrush seeds are unable to germinate (Chambers 2000). For seed located on the soil surface or in and beneath the litter, viability declined to 0 and <20%, respectively. Thus, soil heating and the amount of litter consumed (e.g., table 2) influence the amount of the residual seed pool remaining after fire.

Recovery time required for big sagebrush to approach potential cover on a burn site is variable among ecological sites and subspecies. Mountain big sagebrush canopy cover usually approached 20 to 30% within 20 to 35 years but can take up to 50 years (Bunting and others 1987; Ziegenhagen 2003; Miller and Heyerdahl 2008; Ziegenhagen and Miller 2009; Nelson and others in preparation). In northeastern California, mountain big sagebrush was the dominant plant cover 30 to 60 years following fire but then declined within 80 to 100 years to near 0% cover as a result of increasing juniper dominance (fig. 21) (Miller and Heyerdahl 2008). Ziegenhagen and Miller (2009) concluded that sagebrush recruitment within the first two years after large burns can determine the rate of long-term recovery. In patchy complex burns where a seed source is readily available, recovery can occur within 9 to 15 years (Pyle and Crawford 1996; Ziegenhagen 2003).

Compared to mountain big sagebrush, post-fire re-colonization of Wyoming big sagebrush is very slow to nearly non-existent (Ralphs and Busby 1979; Hosten and West 1994; Beck and others 2008; Rhodes and others 2010). In one study, West and Yorks (2002) reported sagebrush canopy cover was less than 3% 18 years post-fire compared to nearly 18% cover prior to wildfire. The predominant post-fire shrub was Mormon tea, a root-sprouter. The low success of re-colonization is largely attributed to ecological site characteristics. Soil moisture availability is a principal variable in determining establishment of sagebrush (Boltz 1994; Board and others 2011). The majority of Wyoming big sagebrush occurs on relatively warmer and drier ecological sites (warm/dry soil regimes), and cheatgrass is a more effective competitor for early growing season soil moisture than on the cooler-moister mountain big sagebrush sites (fig. 5). Boltz (1994) reported that post-fire survival
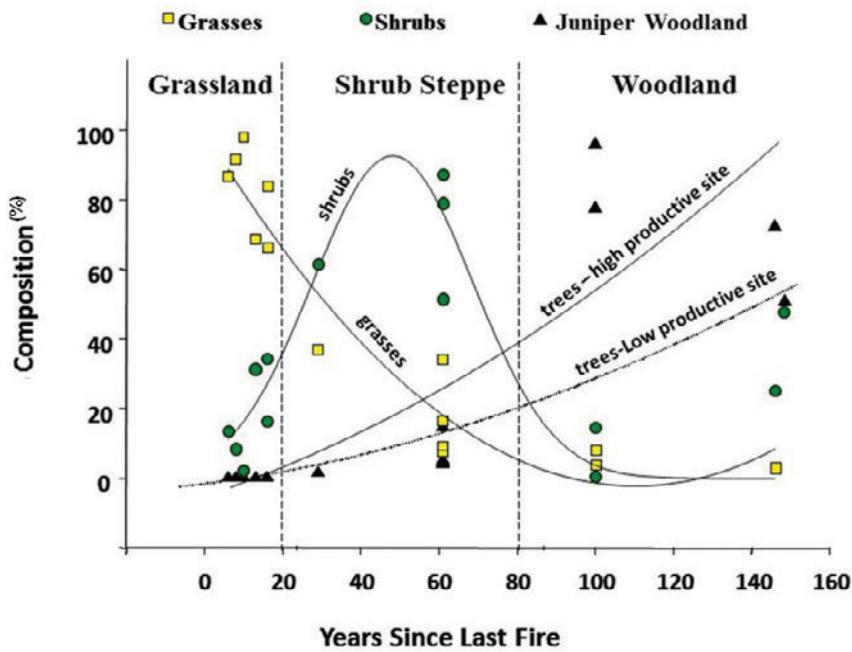
Figure 21. Sagebrush steppe successional model based on canopy cover data collected at the Lava Beds National Monument from warm-cool (warm-frigid) and cool-cool (cool-frigid) mountain big sagebrush communities. Both the degree of tree encroachment and site productivity are influenced by time since the last fire (from Miller and Heyerdahl 2008).

of sagebrush seedlings was considerably less on sites with <10 inches (250 mm) of precipitation compared to sites with >10 inches (250 mm). Recruitment within the first three years was significantly less on warm soils (primarily Wyoming big sagebrush) than cool soils (primarily mountain big sagebrush) (Booth and others 2003; Miller and others in preparation b; fig. 22).

Summary

There is good evidence to conclude that recovery of mountain big sagebrush to pre-burn levels usually takes between 20 to 35 years but can take more than 50 years. Rate of re-establishment is largely dependent on available seed and establishment conditions during the early post-fire years. There is very limited work following long-term succession of Wyoming big sagebrush, but studies strongly indicate re-establishment on ecological sites
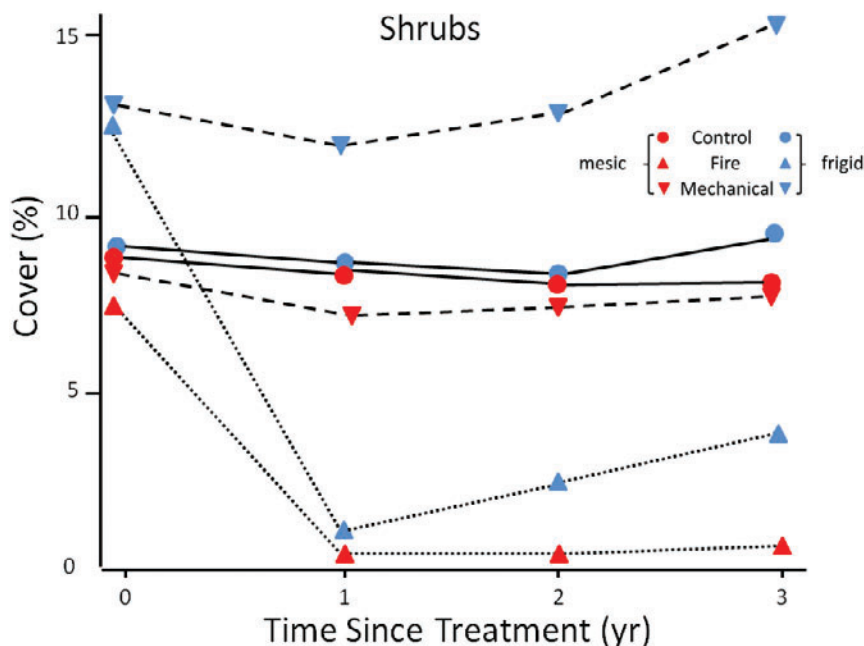
Figure 22. Average changes in shrub cover between two soil temperature regimes over a three-year period on 11 sites across the Great Basin Region following: (1) control, (2) prescribed fire, and (3) cutting and leaving juniper and pinyon trees. Red symbols = warm (mesic), blue symbols = cool (frigid) soil temperature regimes (from Miller and others in preparation b).

with warmer and drier soils will take longer than that exhibited on ecological sites with cooler-moister soils occupied by mountain big sagebrush. Establishment on areas dominated by cheatgrass may not occur at all. There is little documentation on annual seed crop dynamics and environmental variables that influence this variation for sagebrush species. We also have little to no information on post-fire re-establishment of other common species of sagebrush such as low and black sagebrush. Although it is not well documented, establishment of these species will likely be strongly associated with characteristics of the ecological site (e.g., soils, moisture availability, and temperature regimes) (fig. 12). Wyoming big sagebrush recruitment will likely be largely influenced by how cool and wet the ecological site is within the range of soils classified as warm/dry.

Antelope Bitterbrush

The literature related to fire response of bitterbrush, particularly as it relates to resprouting, can be very confusing (fig. 23). Antelope bitterbrush is considered a weak sprouter (Nord 1965) and has been reported to be severely damaged by fire (Hormay 1943; Billings 1952; Blaisdell 1953; Countrymen and Cornelius 1957). However, others have reported active resprouting following fire, which allows the plant to recover within 10 years (Blaisdell 1953). The ability of bitterbrush to resprout after fire is highly variable and has been attributed to genetics, plant age, phenology, soil moisture and texture, and **fire severity** (Blaisdell and Mueggler 1956; Clark and others 1982; Cook and others 1994; Nord 1959, 1965; Riegel and others 2006). Several studies have reported that the highest potential for resprouting occurs in plants older than 5 years but younger than 20 to 40 years (Blaisdell and Mueggler 1956; Martin and Driver 1982). In eastern Oregon, survival of bitterbrush sprouts was influenced by soil surface texture and fire severity, and higher survival rates occurred in spring compared to fall burns (Driscoll 1963; Clark and others 1982). High-intensity wildfires appear to do the most damage to bitterbrush (Wright and others 1979). However, intensity in combination with duration, or **residence time**, determines heat loads at the basal buds. In eastern Idaho, resprouting occurred in 49%, 43%, and 19% in cool, moderate, and hot burns, respectively (Blaisdell and Mueggler 1956). In Northeastern California, 5 to 25% of the bitterbrush successfully resprouted in 5 of 13 fires (Nord 1965). In central Oregon, Driscoll (1963) reported percent sprouting in bitterbrush-big sagebrush communities

Figure 23. Antelope bitterbrush is usually a weak sprouter in the Great Basin region. Although some or many will resprout during the first post-fire growing season, mortality is high during summer drought. Seed stored in the soil or disseminated from nearby mature plants by rodents have a large influence on re-establishment of the species (Mono County, California; photo by Rick Miller).

ranged from 3 to 80% across 10 separate burns. However, on sagebrush communities in northwestern Nevada, eastern Oregon, and western Wyoming, <1% of the antelope bitterbrush survived by resprouting following wildfire (Cook and others 1994; Ziegenhagen and Miller 2009). Resprouting plants can produce flowers and seed the first year after fire, but it usually takes 9 to 15 years to reach pre-burn seed production (Pechanec and Stewart 1954; Wright 1972).

Unlike big sagebrush, bitterbrush seed is produced prior to summer and fall fires and seeds are actively dispersed by rodents through caching activities (Nord 1965; West 1968; Sherman and Chilcote 1972; Vander Wall 1994). Rodent caches typically occur within 80 ft (25 m) of the mother plant and are important for seedling establishment (Vander Wall 1994). However, rodents can have a large impact on re-colonization by consuming the current year's seed crop and browsing young seedlings (Holmgren 1956; Clements and Young 1996, 2001). Within the interior of large burns with few surviving bitterbrush plants, regeneration from the soil seed bank can be an important factor determining the rate of long-term recovery (Ziegenhagen and Miller 2009).

Summary

The ability of antelope bitterbrush to resprout following fire is highly variable and documentation of differences in sprouting among ecotypes and of environmental determinants is limited. It also has been observed that first year resprouts may die during the late summer or winter. Soil moisture availability and fire severity appear to be the most consistent variables attributed to resprouting. So, **soil moisture regime** may be a key factor in determining recovery rate, although annual seed crops and seed predation are also likely to be important. The number of years to recover to near pre-burn levels is highly variable and dependent upon sprouting and temporal patterns of seedling establishment.

Green Rabbitbrush

The majority of research on fire effects for rabbitbrush species has been conducted on green rabbitbrush. It is a common associate with sagebrush in the Great Basin Region and can occur as a minor to sub-dominant in intact sagebrush communities (Young and Evans 1974; Anderson 1986). Green rabbitbrush is a vigorous resprouter and seed producer allowing it to rapidly re-colonize disturbed areas (Plummer 1977) (fig. 24). Its seed can be disseminated long distances by wind. Rabbitbrush cover usually decreases in the first post-fire year followed by large increases within the third year both from resprouting and seed (Blaisdell 1953; Countryman and Cornelius 1957; Chadwick and Dalke 1965; Young and Evans 1974). However, although rapid re-colonization occurred within the first three years on a late **seral** Wyoming big sagebrush ecological site, cover increased only slightly above pre-burn levels during the next four to six years (Rhodes and others 2010). In a long-term study in eastern Idaho, green rabbitbrush cover increased for up to 12 years following fire and then declined below unburned plots within 30 years (Harniss and Murray 1973). In Nevada, green rabbitbrush periodically re-established itself for the first 15 years and then gradually declined as sagebrush dominance increased (Young and Evans 1974). Green rabbitbrush is supported by a taproot and is shorter lived and less competitive than sagebrush in the absence of frequent disturbance. Seedling density, flower production, and shoot growth decline as competition from other species increases (McKell and Chilcote 1957; Young and Evans 1974).

Shrubs in the Salt-Desert and Desert Shrub

In Nevada, a survey of salt-desert communities five years after wildfire reported that recruitment of bud sage was very low and only occurred on 2 of the 24 sites (Haubensak

Figure 24. Green rabbitbrush in the first post-fire growing season that has re-sprouted from the crown and bloomed. New shoots **(A)** early in the growing season following a fall burn. Although this species can rapidly recover after fire, its abundance is usually determined by pre-fire plant composition and structure and post-fire grazing. Green rabbitbrush is supported by a taproot and is shorter lived and less competitive than sagebrush in the absence of frequent disturbance (central Nevada; photo by Rick Miller, insert by Jeanne Chambers).

and others 2009). Shadscale cover was about one-third of adjacent unburned communities. Groves and Steenhof (1988) reported 100% mortality of shadscale in Idaho following a wildfire. In eastern Washington, black greasewood vigorously resprouted following fire but spiny hopsage did not and was absent on the study site two years after the fire (Rickard and McShane 1984). Reports on the response of winterfat are mixed. Several authors reported that sprouting was enhanced by fire (Cottam and Stewart 1940; Blaisdell 1953; Chadwick and Dalke 1965). However, in southwestern Idaho, wildfires resulted in nearly 100% mortality of winterfat (Pellant and Reichert 1983; Groves and Steenhof 1988).

Other Shrub Species

Nevada Mormon tea is a vigorous resprouter following fire (West and Hassan 1985).

Horsebrush is a vigorous resprouter and seed producer that can rapidly recolonize sagebrush communities following fire (Blaisdell 1953; Pechanec and others 1954; Wright and others 1979).

Additional information for shrub and other species occurring in the Great Basin Region can be found on the USDA Forest Service fire effects information site (http://www.fs.fed.us/database/feis/plants/index.html).

Perennial Grasses

Deep-rooted perennial grasses (e.g., bluebunch wheatgrass, Thurber's needlegrass, Indian rice grass, and squirreltail) are an important component that influences ecological site **resilience** to disturbance, **resistance to invasives** (Chambers and others 2007; Davies 2008; Reisner and others 2013; Chambers in review a), and ecological processes. The dominant species associated with sagebrush vary across the Great Basin Region. Idaho fescue is

common in the northern portion of the region and at the higher elevations elsewhere (West 1983b). It is most commonly associated with mountain big sagebrush growing on cool to cold soils with a moist moisture regime (Shiflet 1994; USDA-NRCS 2011). Bluebunch wheatgrass is possibly the most widespread grass across the region and is frequently associated with Wyoming, basin big, and mountain big sagebrush (Shiflet 1994; West 1983b). It most frequently dominates warm/moist soils but is also commonly found on warm/dry and cool/moist soils (USDA-NRCS 2011). Bluebunch wheatgrass typically dominates the warmer aspects (west and south) in cool/moist soils and cooler slope positions (e.g., north aspects) in warm/dry soils. Dominance of bluebunch wheatgrass gradually gives way to Idaho fescue in cool soils as thermal regimes become cooler (Daubenmire 1972). However, on warm soils, it often dominates the cooler end of this regime, such as north aspects. Squirreltail, Sandberg bluegrass, and Thurber's needlegrass are also common throughout the region. Indian ricegrass and needle-and-thread grass also are found throughout the region, becoming more abundant to the south. Additional distribution and fire effects information on the more common grass species can be found online at the USDA Forest Service fire effects information site (http://www.fs.fed.us/database/feis/plants/index.html) and in the USDA Natural Resources Conservation Service plant characteristics database (http://plants.usda.gov/characteristics.html). However, there is little to no fire effects information for many grass species found throughout the Great Basin Region. Only the most common are addressed in this synthesis.

Important questions to address when evaluating the potential response of deep-rooted perennial grasses to fire are:

(1) How does fire influence their short- and long-term cover and productivity?
(2) What role will ecological site characteristics play in their response?
(3) Which species are more fire tolerant than others?

As described below, species morphology, **ecological site** characteristics, and **fire severity** all influence the resilience of this functional group.

Morphology and Fire Tolerance

Although information is limited for many species, the morphological characteristics are important in determining species fire tolerance (Pyke and others 2010). Broad-leaf grasses, including bluebunch wheatgrass, squirreltail, and Columbia needlegrass, are relatively tolerant of fire (Blaisdell 1953; Wright and Klemnedson 1965; Wright 1971; Bunting and others 1987). In contrast, fine leaf grasses such as needle and thread, Thurber's needlegrass, and Idaho fescue are considered less tolerant and can be severely damaged by fire. Increased abundance (denser **packing ratio**) of plant material that accumulates in the crown of fine-leaf grasses will increase the **residence time** of smoldering and the likelihood that growing points located in the crown will be exposed to lethal temperatures (Wright 1971). Location of the buds also influences the susceptibility of species to crown damage due to a fire. Conrad and Poulton (1966) observed (but did not measure) that the buds in Idaho fescue are located at or just above the soil surface compared to bluebunch wheatgrass where they are located just below the surface. However, the response of individual species to fire can vary greatly as a result of a complex of interacting variables that influence fire severity (intensity and duration), including plant size, phenology, and accumulation of litter in the crown. In eastern Oregon, the extent of bunchgrass consumption by fire seemed to be a good predictor of both early post-fire bunchgrass mortality and change in bunchgrass size. Where plants were scorched with >2 inches (51 mm) of stubble remaining on the crown, there was <4% mortality and <25% reduction in basal area the first year following fire (Conrad and Poulton 1966). However, where <2 inches (51 mm) of stubble remained, mortality was >50% and basal area was reduced >33%.

General Response to Fire

Cover of deep-rooted perennial grasses typically declines in the first post-fire year followed by recovery to pre-burn levels in the second or third year (figs. 25A and B). Decline in cover in the first year was reported in 86% of the studies reviewed (Blaisdell 1953; Countryman and Cornelius 1957; Harniss and Murray 1973; Young and Evans 1974; Everett and Ward 1984; West and Hassan 1985; Cook and others 1994; Hosten and West 1994; Pyle and Crawford 1996; Bates and others 2000; West and Yorks 2002; Seefeldt and others 2007; Davies and others 2008; Ellsworth and Kaufmann 2010; Rhodes and others 2010; Miller and others in review). Only two studies reported no change in the first post-fire year in productivity (Cook and others 1994) and cover (Pyle and Crawford 1996). No studies reported an increase in cover the first post-fire year, but the majority (90%) reported recovery to at least pre-burn levels within two to five years.

Only a minority of the short-term studies (<five years) reported an increase in either cover or biomass in the early post-fire years compared to adjacent unburned plots (Countryman and Cornelius 1957; Davies and others 2007; Bates and others 2009; Rhodes and others 2010). Most did not report an increase in deep-rooted perennial grasses in the early post-fire years compared to pre-burn levels or control plots. We found only six studies that evaluated the long-term (>10 years) response of perennial grasses to fire in the Great Basin Region, of which only two did not substitute space for time and collected both pre-treatment and control plot data (Blaisdell 1953; Harniss and Murray 1973 [a follow-up of Blaisdell's study]; West and Yorks 2002). Three studies substituted space for time and did not collect pre-treatment data (Barney and Frischknecht 1974; Koniak 1985; Stager and Klebenow 1987), and one study did not have controls or pre-treatment data but followed the same fire and mechanical treatments over time (Bristow 2010).

West and Yorks (2002) reported deep-rooted perennial grass cover was consistently greater 5 to 18 years after fire on burned-ungrazed plots compared to the burned-grazed and unburned-grazed plots. The authors found that deep-rooted grasses recovered to pre-burn levels on the burned-ungrazed plots by the fifth year and then fluctuated above and below (25 to 57% cover, average = 37%) pre-burn levels (33%) during the next 15 years.

(A)   **(B)**
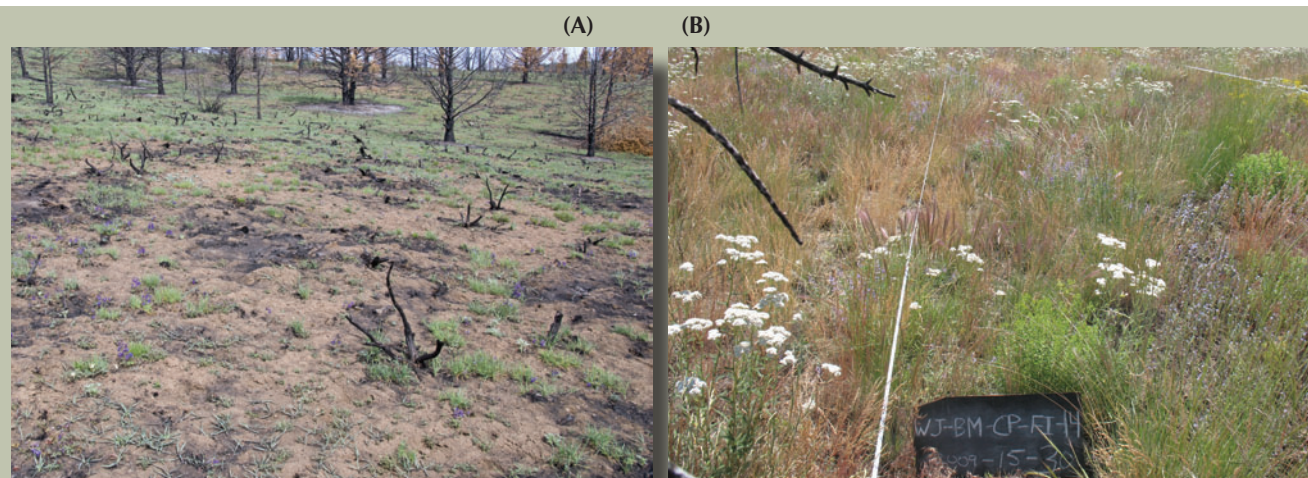
Figure 25. Fire usually reduces cover of deep-rooted perennial grasses in the first post-fire year on both warm/dry and cool/moist sites. However, deep-rooted perennial grasses usually recover or exceed pre-burn levels in the second or third growing seasons if the site was not degraded prior to fire. **(A)** First post-fire growing season on a cool/moist (12-14 inch PZ [30-35 cm]) site and **(B)** the same site in the second growing season. (Modoc County, California; photos by Rick Miller.)

On unburned-grazed and burned-grazed treatments (grazing began the next growing season after the fire), grass cover fluctuated 12 to 30% (average = 20%) and 5 to 33% (average = 20%), respectively, over the same period and never increased above the pre-burn levels (33%). Barney and Frischknecht (1974) reported a short-term increase in perennial grass cover and then a gradual decline starting about 35 years after fire as a result of increasing competition with juniper. Harniss and Murray (1973) measured a rapid increase (200%) in perennial grass cover in the first three post-fire years followed by a more gradual increase during the next nine years, then followed by a gradual decline. Blaisdell (1953) reported that post-burn cover of grasses 15 years after the fire were similar to pre-burn levels. However, in Nevada where deep-rooted perennial grass cover was low (<1%) prior to burning, cover remained low (<2.5%) 36 years (Bristow 2010) and 22 to 60 years (Koniak 1985) after treatment. Very little work has documented the minimum amount of deep-rooted perennial grasses necessary to allow an ecological site to recover. Bates and others (2007a) reported two to three plants/m^2 was sufficient for bunchgrasses to recover following the removal of a closed western juniper stand on the low elevation end of a cool **soil temperature regime** (southwest slope at 5000 ft [1525 m] in the Malheur High Plateau **MLRA** (fig. 2), where the elevation break between warm and cool soils is 4500 ft [1372 m]). However, in addition to soil temperature/moisture regimes, post-fire disturbance and threat of invasives likely will influence the amount of pre-fire deep-rooted perennial grasses necessary for recovery.

Ecological Site Characteristics

Across both warm/dry and cool/moist soil regimes, recovery of deep-rooted perennial grasses was closely associated with pre-fire composition. Six studies conducted on cool/moist mountain big sagebrush communities reported an increase in perennial grass cover above pre-burn levels within two to four years following fire (Blaisdell 1953; Harniss and Murray 1973; Cook and others 1994; Pyle and Crawford 1996). For Wyoming big sagebrush, three studies also reported increases in perennial grass cover above pre-burn levels following fire (West and Yorks 2002; Davies and others 2007; Bates and others 2009). All three study locations were in a non-degraded **seral** stage and very near the elevation break separating warm and cool soils. The short-term response of deep-rooted perennial grasses on ecological sites with different soil temperature/moisture regimes was usually similar when ecological sites were in a non-degraded seral stage and Wyoming big sagebrush occupied the cool end of warm soils (near the elevation break between warm and cool soils). However, the response of invasives was very different between the two regimes (discussed in Invasive Grasses section) when the abundances of deep-rooted perennial grasses were low. When deep-rooted perennial grass cover on cool soils was low pre- and post-fire, post-fire cover of cheatgrass also remained low (Koniak 1985; Bristow 2010). However, on a study site with <2% deep-rooted perennial grass cover and 2.5 plants/10 ft^2 (2.5 plants/m^2), there was an 8-fold increase in cheatgrass biomass on cut western juniper plots (used in place of fire) compared to uncut plots in the second year following treatment (Bates and others 2007a). On warm (**mesic**) soils, the density of medusahead was closely associated with the density of perennial grasses (Davies 2008). Warm/dry ecological sites with a low abundance of deep-rooted perennial grasses became dominated by cheatgrass (see Invasive Grasses section).

Summary

The literature consistently reports a decline of deep-rooted perennial grasses in the first post-fire year followed by recovery to pre-burn levels within two to three years. Only a few short-term studies reported an actual increase in deep-rooted perennial grasses above pre-burn or controls. However, long-term studies usually reported increases in abundance of

deep-rooted perennial grasses when they were not severely depleted. These studies generally reported an early increase and then gradual decline in deep-rooted perennial grasses 12 and 40 years after fire as a result of increasing woody plant cover. Response to fire on ecological sites with a depleted abundance of this functional group was usually very limited. The literature also suggests that increases in cover and biomass are more likely to occur on ecological sites with both cooler and moister soil temperature/moisture regimes and an intact perennial grass layer (figs. 26A, B, and C). Primary limitations we found were: (1) the lack of post-fire management information such as grazing and other disturbances, and (2) and need for more long-term studies.

Figure 26. Post-treatment plant response is largely a function of *site characteristics* (particularly soil temperature/moisture regimes) + *pre-treatment plant composition and structure + treatment type and severity*. Examples shown are **(A)** degraded warm/dry community, **(B)** degraded cool/moist community, and **(C)** late seral cool/moist community in the early stages of tree encroachment. **(A)** This community occupies a warm/dry soil temperature/moisture regime that is in a degraded state (near absence of deep rooted perennial grasses). The potential for recovery (resilience) and resistance to invasives is very low due to site characteristics and severely reduced deep-rooted perennial grasses. Without seeding, this community will likely become dominated by cheatgrass and snakeweed (Fossil Co., Oregon; photo by Rick Miller). **(B)** A hand cutting was used on this site where surface fuels were too limited to carry fire. This is a cool/moist site on the warm/dry end of the temperature/moisture gradient dominated by bluebunch wheatgrass. Bunchgrasses accounted for 2.5% of ground cover at 2 to 3 plants/10 ft^2 (2 to 3 /m^2) prior to treatment. The picture shows the release of perennial grasses in the first year post-treatment. Although cheatgrass persists on this site, especially in wet years, deep-rooted perennial grasses dominate the herbaceous layer (Steens Mountain, Oregon; photo by Rick Miller). **(C)** This mountain big sagebrush/Idaho fescue community occupies a cool/moist soil temperature/moisture regime and is in a mid-seral stage (Phase II) of woodland encroachment. In the early to mid-phases of woodland development, fire severity will still be moderate and recovery potential remains high if the understory remains intact. However, with the continued increase in juniper, fire severity will shift from moderate to high, substantially increasing mortality of the bunchgrasses (northeastern California; photo by Rick Miller).

Bluebunch Wheatgrass

Bluebunch wheatgrass is one of the most fire tolerant bunchgrasses in the Great Basin Region. Fire response of bluebunch wheatgrass is often limited in the first post-fire year, resulting in little change in cover. This is usually followed by an increase above pre-burn levels in the second or third year (Blaisdell 1953; Conrad and Poulton 1966; Uresk and others 1976, 1980; Hosten and West 1994). Blaisdell (1953) reported that cover doubled in low to moderate burns and increased 143% in a high severity burn. Uresk and others (1976, 1980) reported greater biomass production and reproductive effort in the first three post-fire years compared to pre-burn levels. Reproductive effort (reproductive shoots, flowering, and/or seed production) following fire often increases (Uresk and others 1976; Patton and others 1988; Sapsis 1990; Bates and others 2009; Ellsworth and Kaufmann 2010). Only a few studies have compared spring and fall burning effects. In a **burn barrel** experiment (where vegetation is burned inside a barrel to contain the fire), reduction of basal area was greater in a spring burn than fall burn (Britton and others 1990).

Idaho Fescue

Idaho fescue is typically thought to be sensitive to fire, but responses reported in the literature are highly variable during the early post-fire years. In Idaho, Blaisdell (1953) reported slight reductions in biomass for light to **moderate severity fires** but 50% reduction in **high severity fire**. In eastern Oregon and Montana, Idaho fescue cover, density, and frequency approached unburned levels within two to three years (Antos and others 1983; Jourdonnais and Bedunah 1990; Sapsis 1990). However, others have reported cover of Idaho fescue remained below that in unburned plots for over a decade after the burn (Harniss and Murray 1973). In Montana, Wambolt and others (2001) reported declines on four study locations but an increase on a fifth location. The confusion may partially be explained by the lack of information on fire severity and post-fire grazing by wild and/or domestic herbivores, which can have a large impact on post-fire response. For example in the first case, in eastern Oregon, the extent to which Idaho fescue was consumed by fire was a good predictor of both early post-fire mortality and change in basal area. Where plants were scorched but left with >2 inches (51 mm) of stubble on the crown, there was <4% mortality and <25% reduction in basal area the first year following fire (Conrad and Poulton 1966). However, where <2 inches (51 mm) of stubble remained, mortality was >50% and basal area reduction was >33%. To add to the confusion, Seefeldt and others (2007) reported 33% mortality of Idaho fescue following a **low severity fire** while a second study concluded that Idaho fescue was less sensitive to high severity fire than bluebunch wheatgrass (Robberecht and Defoss 1995). Another study found that Idaho fescue seed production was initially suppressed but increased above plants that were not burned in the fifth year following fire (Patton and others 1988). The authors reported reproductive shoot density and seed production following fire was generally greater on four mountain big sagebrush study sites compared to one basin big sagebrush study site. Bates and others (2009) reported seed production was greater on burned than unburned plots in the first two years following burning. When evaluating the potential response of Idaho fescue following fire, key characteristics to consider are **fire severity**, post-fire management and disturbance, and pre-treatment plant composition. Idaho fescue typically grows on cool to cold/moist ecological sites, but other key components (fig. 15) are also important to consider.

Thurber's Needlegrass

Thurber's needlegrass is a fine leaf grass and often is considered sensitive to fire (Wright and Klemmedson 1965). However, species response appears to depend on pre-fire plant size, season of burn, fire severity, and ecological site characteristics (e.g., soil temperature/moisture regime). Large plants appeared to be more prone to mortality than smaller

plants in a burn barrel experiment. Season of burn also appeared to be a factor influencing response. Britton and others (1990) reported 90% mortality for plants burned in May and June compared to no mortality when burned in summer or fall. On a Wyoming big sagebrush study site with 11.5 inches (290 mm) precipitation and cool soils (just above the elevation break between warm and cool), seed production was greater on the burn in the first two post-fire years (Bates and others 2009). On a similar ecological site, cover was reduced in the first year but recovered by the second year after fire, while reproductive effort was greater in the first year but similar to unburned plots in the second year (Davies and Bates 2008). This species commonly grows in both warm/dry and cool/moist soil temperature/moisture regimes, which may contribute to its variable response to fire. However, no work has compared response across these soil regimes. One might expect fire tolerance to be greater on cool/moist ecological sites that historically had more frequent fires (e.g., Davies and others 2011). However, differences in ecological site factors influencing fire severity must also be considered.

Needle and Thread

Needle and thread, also a fine leaf grass, is considered sensitive to fire. However, the degree of damage is influenced by season of burning, fire severity, and plant size. Needle and thread is most commonly found on warm/dry soils in contrast to Thurber's needlegrass, which is found on both warm/dry and cool/moist soils. In one study, needle and thread was more severely damaged (in a burn barrel experiment) than Thurber's needlegrass (Wright and Klemmedson 1965). The same study reported that season of burn was the primary determinant of the extent of damage to needle and thread. However, size of plant became increasingly important in determining the effect of burning during the latter part of summer. Mortality was >90% for both large and small plants burned in June but only 20% for plants burned in July (Wright and Klemmedson 1965). Total basal area for large plants was reduced >95% in June and July and 68% in August. Basal area reduction of small plants was 100% in June (total mortality) and 82% in July. However, the authors also found that in a July wildfire, mortality was only 10% for large plants and 0% for small plants. Basal area reduction in the wildfire was similar (94%) for large plants but only 52% for small plants, significantly less than the reduction in the burn barrel experiment.

Indian Ricegrass

Little research has been conducted evaluating fire effects on Indian ricegrass. However, observations suggest that although basal area may be reduced, survival is high (Wright and others 1979). Recovery has been reported to range from slow (Pechanec and Stewart 1944) to rapid (Barney and Frischknecht 1974).

Other Needlegrasses

Limited work has been conducted on other needlegrass species common throughout the Great Basin Region. Therefore, one must take into account the conditions (ecological site, fire severity, and post-fire weather) under which the species burned when considering the results of only one or two studies. Western needlegrass decreased the first year after an August wildfire in northeastern California, but increased by the third post-fire year, nearly doubling in basal area (Countryman and Cornelius 1957). In Idaho, Blaisdell (1953) reported Columbia needlegrass was not significantly affected by fire severity 12 years after a burn.

Basin Wildrye

Few research studies have evaluated fire effects on basin wildrye. However, observations typically suggest high survival rates and vigorous resprouting following fire (Young 1987).

In eastern Oregon, fall and spring burning increased total shoot and reproductive shoot densities in the first year, although live basal areas were similar between burn and unburned plants. By the second year, there was little difference between burned and control treatments.

Bottlebrush Squirreltail

Squirreltail is considered an early **seral** species and is highly competitive with cheatgrass (Booth and others 2003). Squirreltail, a broad-leafed grass, is tolerant to fire if it has senesced, and responds positively and rapidly in biomass, cover, and reproductive effort (Wright and Klemmedson 1965; Young and Miller 1985; Blank and others 1994b; Bates and others 2009). Prior to fire, squirreltail had half the amount of dead plant material per unit crown area in contrast to needle and thread (Wright 1971). Biomass of shoots, roots, and inflorescence were greater in the first year following a fire in July (plants were fully senesced at the time of the burn) than unburned plants (Young and Miller 1985). Reproductive culms accounted for 74.8% of all shoots in the burn compared to 14.3% in unburned plots. Shoot biomass increased 5.8 times/unit of crown area and root biomass was 49% higher for plants in burned plots compared to unburned plots. Bates and others (2009) also reported increased seed production in the first two post-fire years. However, Wright and Klemmedson (1965) reported some damage to plants in a July burn when foliage was still green and fire conditions severe (air temperature at 100 °F [38 °C] and relative humidity 11%). Large plants appeared to be less resistant. High **fire intensity** (800 °F [427 °C] measured at the soil surface) resulted in 25% mortality compared to no mortality at moderate intensity (400 °F [204 °C]). They reported heat tolerance of the tissue increased from May to September (Wright and Klemmedson 1965).

Shallow-Rooted Perennial Grasses

The majority of studies reported little effect of fire on the shallow-rooted perennial grass, Sandberg bluegrass (Blaisdell 1953; Wright and Klemmedson 1965; Young and Evans 1978; Akinsoji 1988; Miller and others in press). Only one study reported a reduction in biomass but not density in the first post-fire year (Davies and others 2008). Sandberg bluegrass, which primarily roots in the upper 3 to 4 inches (70 to 100 mm), has an open crown with limited biomass, therefore limiting fire intensity and duration.

Summary

Bluebunch wheatgrass and bottlebrush squirreltail are two of the more fire-tolerant bunchgrasses in the Great Basin Region. Fine leaf grasses are usually less tolerant, sustaining more damage than the broadleaf grasses. The response of Idaho fescue to fire is highly variable. Key variables that influence post-fire response are fire severity at the crown level, accumulation of biomass, plant size, season of burn, weather conditions at the time of the burn, and post-fire disturbance. The majority of studies did not monitor fire severity nor take into account post-fire disturbance, especially herbivory by wild and domestic herbivores (see Post-fire Grazing section).

Perennial Forbs

Species and Morphology

High annual variation and often limited abundance of individual perennial forb species make it difficult to evaluate fire impacts on this functional group on a species basis. Perennial forbs usually account for the largest number of species in sagebrush steppe communities although they typically comprise <10% of plant cover and biomass (Daubenmire 1970; Harniss and Murray 1973; Barney and Frischknecht 1974; Blaisdell and others 1982; Passey and others 1982; Jensen 1989; Miller and others 2000; West and Yorks 2002).

Perennial forb abundance is closely related to moisture availability. Abundance is highly variable from year to year due to fluctuations in precipitation and along environmental gradients due to continually changing moisture availability. In a survey of 106 big sagebrush study sites (including both mountain and Wyoming subspecies) in southeast Oregon, mean perennial forb biomass and species richness was 4.5- and 1.9-fold greater in mountain big sagebrush than Wyoming big sagebrush communities, respectively (Davies and Bates 2010). However, the authors did not report the relative abundance (e.g., percent composition) of perennial forbs based on total herbaceous biomass. In response to variation in annual precipitation from 50 to 185% of average in a two year period, perennial forb biomass varied from 20 to 80 lbs/acre (21.5 to 86 kg/ha) on a Wyoming big sagebrush/ Thurber's needlegrass study site, respectively (Miller and Eddleman 2001). In Utah, perennial forb cover varied between <1 to 8% over a 19-year period (West and Yorks 2002).

What we know about perennial forb sensitivity to fire on a species basis primarily relates to growth forms. **Fire intensity** and duration in combination with location of buds are important characteristics influencing mortality. Species with buds located below the soil surface (corms, bulbs, rhizomes, and basal or crown buds) are typically tolerant to fire (table 4) (Fites-Kaufman and others 2006). This growth form is classified as Cryptophytes in the Raunkiaer life form classification system, which is used in the USFS Fire Effects Information System (FEIS; see Appendix 2). Mat-forming perennial forbs, shrubs, and half-shrubs that retain their buds on the soil surface (Hemicryptophytes) or within 20 inches (510 mm) of the soil surface (Chamaephytes) can be severely damaged by fire. Examples of fire-sensitive perennial forbs, all of which have their buds located above ground, are *Eriogonum* spp., *Antennaria* spp., *Arenaria* spp., and *Pholx* spp. (fig. 27A) (see Appendix 1 for common names). Fire-tolerant species, including arrowleaf balsamroot, tailcup lupine, silvery lupine, mules-ears, longleaf phlox, largehead clover (fig. 27B), and thorn skeleton weed (Cryptophytes), usually increase following fire (table 4) (Young and Evans 1978; Goergen and Chambers 2009). The increase in species such as silvery lupine in mountain big sagebrush communities has been attributed to both resprouting and reproduction from seed (Goergen and Chambers 2009). Species in the Cichorieae (also called Lactuceae) tribe (Cryptophytes), an important food source for sage-grouse, have been reported to increase in the first or second year following fire (Pyle and Crawford 1996; Wrobleski and Kauffman 2003). The FEIS (available at: http://www.fs.fed.us/database/feis/) provides Raunkiaer life forms for all species under the Botanical and Ecological Characteristics heading in relation to fire susceptibility.

Short-term studies reporting change in perennial forb cover or biomass following fire are mixed. A number of studies reported no change in perennial forb biomass during the first one to five years following fire compared to levels in pre-burn or nearby unburned plots (Cook and others 1994; Hosten and West 1994; Fischer and others 1996; West and Yorks 2002; Beck and others 2008; Rhodes and others 2010; Bates and others 2011), while other studies have reported increases (Stager and Kelbenow 1987; Martin 1990; Pyle and Crawford 1996; Wrobleski and Kauffman 2003; Bates and others 2011). Ecological site characteristics appear to be an important factor driving the response of this functional group. The lack of response following fire is most consistent on soils with warm/dry temperature/moisture regimes occupied by Wyoming big sagebrush (Young and Evans 1974; West 1986; Cook and others 1994; Fischer and others 1996; Hosten and West 1994; Rhodes and others 2010; West and Yorks 2002). However, on mountain big sagebrush ecological sites with cool/moist soil regimes, perennial forb production increased in 70% of the studies we reviewed (fig. 28) (Stager and Klebenow 1987; Martin 1990; Pyle and Crawford 1996; Wrobleski and Kauffman 2003; Dhaemers 2006; Davies and Bates 2010; Bates and others 2011), while 30% reported no change (Nelle and others 2000;

Table 4. Examples of some common fire tolerant and fire intolerant perennial forbs related to their growth form in the Great Basin Region.

Tolerant (damage–none to slight)	Intolerant (damage–moderate to severe)
Buds below ground	*Buds above ground*
common yarrow (*Achillea millefolium*)	pussytoes (*Antennaria* spp.)
mountain dandelion (*Agoseris* spp.)	sandwort (*Arenaira* spp.)
onion (*Allium* sp.)	matted buckwheat. (*Eriogonum caespitosum*)
aster sp. (*Aster* sp.)	Douglas buckwheat (*Eriogonum douglasii*)
milkvetch sp. (*Astragalus* sp.)	parsnip buckwheat (*Eriogonum heracleoides*)
arrowleaf balsamroot (*Balsamorhiza* spp.)	slender buckwheat (*Eriogonum microthecum*)
mariposa lilly (*Calochortus* spp.)	rock buckwheat (*Eriogonum sphaerocephalum*)
hawksbeard (*Crepis* spp.)	sulfur-flower buckwheat (*Eriogonum umbellatum*)
fleabane (*Erigeron* spp.)	spiny phlox (*Phlox hoodii*)
sticky purple geranium (*Geranium viscosissimum*)	
old man's whiskers (*Geum triflorum*)	
biscutroot (*Lomatium* spp.)	
lupine sp. (*Lupinus* spp.)	
bluebells sp. (*Mertensia* spp.)	
woolly groundsel (*Pakera* cana)	
penstemon spp. (*Penstemon* spp.)	
longleaf phlox (*Plox longifolia*)	
lambstongue ragwort (*Senecio integerrimus*)	
largehead clover (*Trifolium macrocapum*)	
death camus spp. (*Zigadenus* spp.)	
mule's ear (*Wyethia amplexicaulis*)	

Derived from Blaisdell 1953; Pechanec and others 1954; Mueggler and Blaisdell 1958; Lyon and Stickney 1976; Klebenow and Beall 1977; Wright and others 1979; Volland and Dell 1981; Bradley and others 1992; Pyle and Crawford 1996; Riegel and others 2006; Fire Effects Information System.
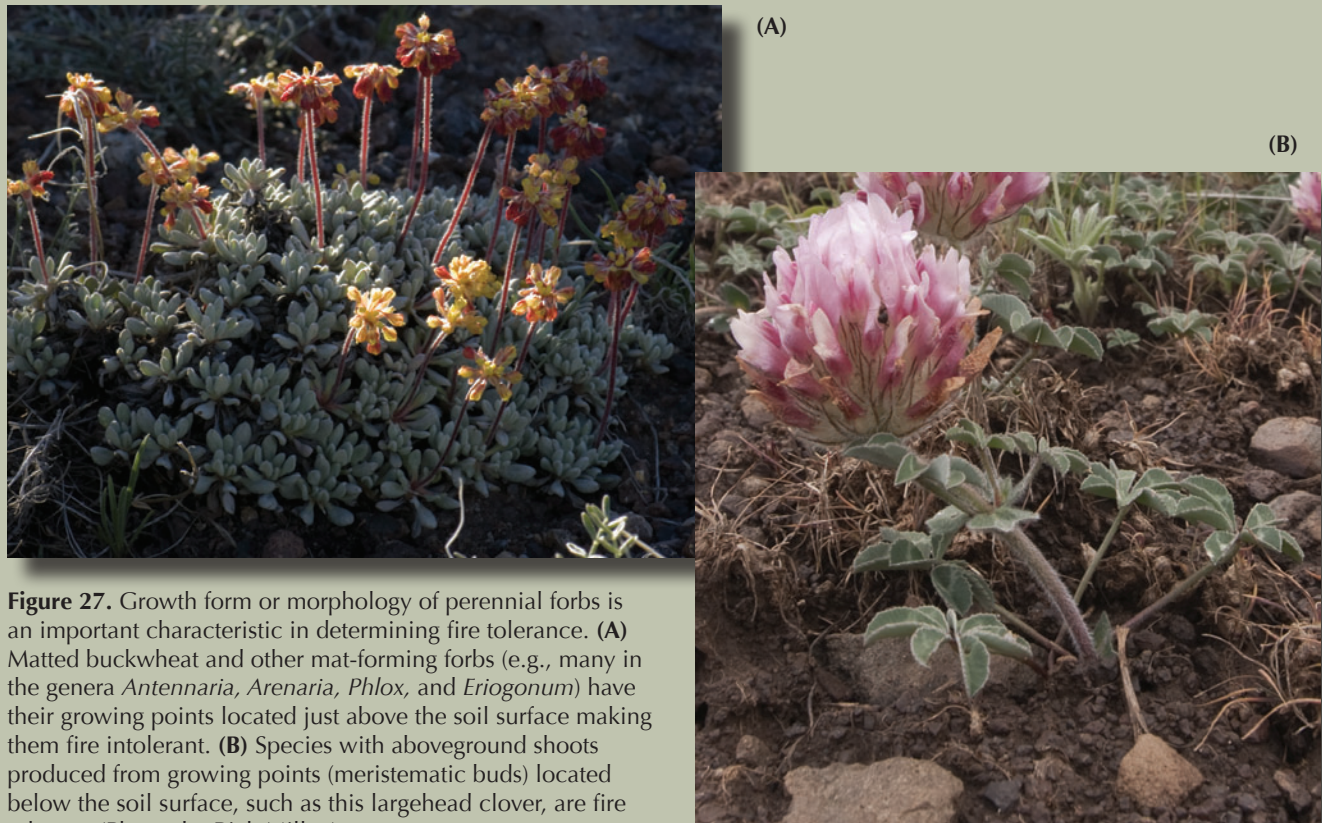
Figure 27. Growth form or morphology of perennial forbs is an important characteristic in determining fire tolerance. **(A)** Matted buckwheat and other mat-forming forbs (e.g., many in the genera *Antennaria, Arenaria, Phlox,* and *Eriogonum*) have their growing points located just above the soil surface making them fire intolerant. **(B)** Species with aboveground shoots produced from growing points (meristematic buds) located below the soil surface, such as this largehead clover, are fire tolerant. (Photos by Rick Miller.)

Bates and others 2011; Miller and others in preparation b). Unfortunately, there is not enough information in these studies to separate the role of post-fire precipitation and temperatures on either warm or cool soils. Seedling survival, reproductive effort, and crown volume of largeflower hawksbeard, tapertip hawksbeard, and woollypod milkvetch—three important sage-grouse food forbs—were higher in burned than unburned mountain big sagebrush communities (Pyle and Crawford 1996; Wirth and Pyke 2003). Nitrogen concentration in tapertip hawksbeard, sulphur-flower buckwheat, and silvery lupine were higher in burned than unburned mountain big sagebrush communities, but individual plant biomass was determined by seasonal precipitation (Rau and others 2008). Forb succulence during the growing season was also extended for about two weeks on burn plots (Wrobleski and Kauffman 2003). Others have reported no increases in important perennial food-forbs for sage-grouse following fire on either Wyoming or mountain big sagebrush study sites (Fischer and others 1996; Nelle and others 2000; Beck and others 2008; Miller and others in preparation b).

Studies reporting long-term effects of fire on perennial forbs are limited and the majority substitute space for time (different aged fires are evaluated), lack paired controls, and/or lack pre-fire plant composition data. Only two studies followed perennial forb cover over time on the same location (Harniss and Murray 1973; West and Yorks 2002). Over a 68-year period on a mountain big sagebrush study site, Harniss and Murray (1973) reported an increase in perennial forb cover during the first 12 post-fire years, followed by a decline. However, on a Wyoming big sagebrush study site, perennial forb cover did not increase following fire (West and Yorks 2002). Substituting space for time across 21 different fires in California and Nevada, Koniak (1985) reported a higher cover of perennial forbs on study sites in earlier successional stages (<8 years) than later stages (22 to 60 years). These study sites were primarily located on cool soils with dry and moist **soil moisture regimes**. However, in Utah, perennial forb cover did not relate to time since fire across 28 different burns varying in age from 3 to 100+ years (Barney and Frischknecht 1974). Post-fire disturbance, which could significantly impact the long-term response of perennial forb abundance and composition, was not described in any of these studies, with the exception of West and Yorks (2002).

Figure 28. The response of perennial forbs following fire is highly variable. In the studies reviewed, increases in perennial forbs were rare on warm/dry sites but occurred on 70% of the cool/moist sites (northeastern California; photo by Rick Miller).

Summary

Perennial forbs make the largest contribution of any functional group to the species diversity of sagebrush steppe plant communities, yet the variety of post-fire responses documented to date still lacks satisfying explanations. Although the potential for increasing perennial forb cover and biomass on soils with warm (mesic) **soil temperature regimes** is not supported by the literature, cover did increase in the majority of studies on soils with cool/moist (frigid/xeric) regimes. It may also be reasonable to expect that responses may be closely tied to post-fire precipitation, but there is little literature to support or dispute this. Importantly, increases in sage-grouse perennial food forbs were inconsistent, limited to the cool/moist sagebrush ecological sites, and often only applied to the annual forb functional group.

Annual Forbs

Annual native and exotic annual forbs usually increase in cover and biomass during the first few years following fire, decline sometime after the fifth year, and remain low during the later stages of succession. Unfortunately, most studies that evaluated post-fire response of annual forb cover and biomass did not separate natives from invasives. Annual forb cover usually increases in the first one to five years post-fire followed by a decline within three to five years (fig. 29) (Barney and Frischknecht 1974; Everett and Ward 1984; Koniak 1985; Cook and others 1994; Wright and Chambers 2002; Dhaemers 2006; Bates and others 2007a, 2011; Davies and others 2007; Rhodes and others 2010; Miller and others in review). Bates and others (2007a) reported annual forb cover increased >3-fold in the first year following fire but dropped below pre-burn levels in the fifth post-fire year. Annual forb biomass in unburned and burned plots was 36 and 100 lb/acre (39 and108 kg/ha) in the first post-fire year and 56 and 175 lb/acre (60 and 188 kg/ha) in the second year, respectively (Davies and others 2007). Several studies that evaluated individual species reported a large portion of the response was from invasive forbs, including pale alyssum, tansy-mustard, and tumble mustard, on warm ecological sites or ecological sites in a degraded low-**seral** stage (Barney and Frischknecht 1974; Koniak 1985; Young and Miller 1985; Rhodes and others 2010). Long-term studies reported total annual forb (often invasive and native) cover was low in late seral stages (Harniss and Murray 1973; Barney and Frischknecht 1974; Koniak 1985; West and Yorks 2002). Large differences in post-fire response between warm and cool soils have been reported across the Great Basin Region (Miller and others in

Figure 29. Native annual forbs, such as the ground smoke (*Gayophytum* sp.) shown in the photo, can be the dominant herbaceous species in the first post-fire growing season following a high severity fire. Native annual forbs can be particularly abundant where native perennial grasses and forbs were in low abundance prior to fire. However, native annual forbs can be outcompeted by exotic annual grasses and forbs in the second post-fire year given a seed source (Milford Flat, west central Utah; photos by Rick Miller).
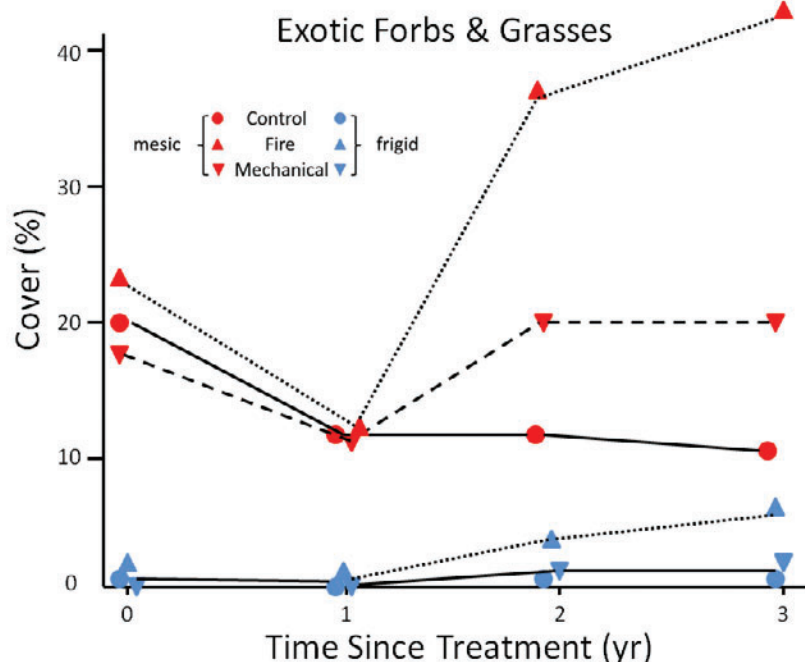
Figure 30. Average exotic annual forb and grass cover between two soil temperature regimes over a three-year period on 11 sites across the Great Basin Region following: (1) control, (2) prescribe fire, and (3) cutting and leaving juniper and pinyon trees,. Red symbols = warm (mesic) and blue symbols = cool (frigid) soil temperature regimes (from Miller and others in preparation b).

preparation) (fig. 30). Exotic forb cover increased from 10 to 30% in the third year following fire on warm soils compared to an increase from <1 to 1.4% on cool soils (Miller and others in preparation b). Although large increases in annual forb cover is the most common response following fire, several studies have reported limited increases in cover (Everett and Ward 1984; West and Hassan 1985).

In a study that focused on annual forbs known to be found in the diet of sage-grouse, Miller and others (in press) reported a significant increase (2.5 to 6%) in the second and third post-fire years. Rhodes and others (2010) reported increases in sage-grouse annual food forbs the first year following fire followed by declines to unburned levels by the second year, which persisted through the fifth year. Biomass and cover of this functional group is highly responsive to annual moisture and competition from perennials.

Summary

Annual native and exotic forbs usually increase in cover and biomass during the first few years following fire, decline sometime after the fifth year, and remain low during the later stages of succession. However, the magnitude of increase and their persistence is likely influenced by the early post-fire response of perennial herbs (grasses and forbs). We suspect that elevated levels of exotic annual forb cover will persist on ecological sites in a degraded early-seral phase. However, there are no long-term studies to confirm this. It also appears that invasive forbs are more abundant on warm than cool soils following disturbance (Chambers and others in review), although studies are limited.

Invasive Annual Grasses

Cheatgrass

Cheatgrass has been a very successful invader in the Great Basin Region. Its ability to invade and/or dominate following fire or other disturbances is closely related to the soil temperature/moisture regime and the abundance of deep-rooted perennial grasses prior to and after the fire. In 1994, cheatgrass and medusahead were estimated to dominate or have a significant presence (>10% composition based on biomass) on at least 17.3 million

Figure 31. Many warm/dry sites in the Great Basin Region have crossed thresholds to alternate states dominated by exotic annuals such as cheatgrass. Potential historic vegetation on the lower slopes of this site was likely Wyoming big sagebrush with an understory of deep-rooted perennial grasses shifting to mountain big sagebrush communities with increasing elevation. The cool/moist sites support Jeffrey pine and western juniper and have a higher resistance to exotic annuls than the lower elevations, which are currently dominated by cheatgrass (Long Valley, northeastern California; photo by Nolan Preece).

acres (7 million ha) in the Intermountain West (Pellant and Hall 1994). Surveys in 2005 using combinations of field data and remote imagery estimated that cheatgrass dominated >5 million acres (2.02 million ha) in the northern Great Basin (Bradley and Mustard 2005, 2006; Peterson 2005) (fig. 31). An earlier estimate of nearly 100 million acres (40.5 million ha) (Whisenant 1990) was a misinterpretation of a paper by Mack (1981), who stated that cheatgrass now dominates on many rangelands "within" 100 million acres (40.5 million ha) of potential steppe vegetation in the Intermountain West, United States (Miller and others 2011). The ability of cheatgrass to readily invade following fire, especially on heavily grazed rangelands, was reported in the 1920s and 1930s (Pickford 1932; Stewart and Hull 1949).

The capability of cheatgrass to invade and dominate depends on a number of interacting factors, including amounts and timing of available moisture, seasonal temperatures, plant community composition, competitive abilities of resident species, and characteristics of the disturbance (Chambers and others 2007) (fig. 12). The temporary increase in resource availability in the early post-fire years (Hobbs and Schimel 1984; Blank and others 1994b; Young and Allen 1997; Stubbs and Pyke 2005; Davies and others 2007; Rau and others 2009) can temporarily lower a community's resistance to invasive species in areas with favorable climates (Beckstead and Augspurger 2004; Johnson and others 2011). This is partially due to the ability of annuals to more quickly take up these resources. The abundance of cheatgrass is also closely associated with annual precipitation, especially on warm/dry ecological sites, increasing during wet years and declining during dry years (West 1994; West and Yorks 2002; Bates and others 2007a). The increase in cheatgrass and other invasive annuals has been reported to increase regional fire activity (Balch and others 2013), especially during wet years (Knapp 1995; Littell and others 2009). Based on a predictive model, the risk or likelihood of fire was <60, 60 to 86, and >86% when cheatgrass cover was <20, 20 to 40, and >40%, respectively (Link and others 2006). Elevated CO_2 levels have also been associated with significant increases in cheatgrass (Smith and others 1987, 2000; Ziska and others 2005).

Post-fire year 1–In sagebrush communities, cheatgrass cover usually does not increase and often declines the first year following fire (Young and Evans 1978; West and Hassan 1985; Young and Miller 1985; Hassan and West 1986; Akinsoji 1988; Cook and others 1994; Pyle

and Crawford 1996; Bates and others 2000; Davies and others 2007; Mata-Gonzalez and others 2007; Rowe and Leger 2011; Miller and others in press). However, a study in Utah reported an increase in cover in the first year from about 8% pre-burn to 36% post-burn (Hosten and West 1994). The limited response of cheatgrass that usually occurs in the first year may be a result of reduced micro-relief or micro-sites conducive to germination and establishment (Hilty and others 2003; Davies and others 2009a), reduced litter cover, and/ or consumption of seed by fire (Young and Evans 1975; Hassan and West 1986; Humphrey and Schupp 2001; Allen and others 2008). The majority of seed is usually located beneath shrub canopies (Young and Evans 1975; Allen and others 2008) where fire burns the hottest and the majority of surface organic matter is usually consumed (Davies and others 2009a). In Utah, on a Wyoming big sagebrush study site, nearly 90% of the seed in the litter layer beneath sagebrush canopies was consumed by fire (Hassan and West 1986). Litter (see Litter section) also plays a role in the establishment of invasive annuals (Chambers 2000). Evans and Young (1972) reported micro-relief, and litter cover increased soil moisture in the top 3.5 inches (90 mm) and enhanced germination of both cheatgrass and medusa-head. The one-year delay in increase of cheatgrass cover following fire emphasizes the importance of seeding communities depleted in native perennial grasses and forbs prior to the first growing season to improve the chances of plant community recovery. Evans and Young (1970, 1978) reported better seedling establishment success when seeding occurred prior to the first post-fire growing season compared to seeding after the first or second post-fire growing seasons.

Post-fire years 2-4–Cheatgrass cover usually increases in the second and third post-burn years on warm/dry ecological sites (West and Hassan 1985; West and Yorks 2002; Mata-Gonzalez and others 2007; Miller and others in press). In a central Utah study, pre-fire cheatgrass biomass was 13.5 lb/acre (14.5 kg/ha) (West and Hassan 1986) compared to 680 lb/acre (732 kg/ha) in the second and third post-fire years. West and Yorks (2002) also reported large increases in cheatgrass cover (6% pre-burn to 60% post-burn) in the first few years following fire. Both studies were located in communities with warm soils but good perennial grass cover (>30%). In west-central Nevada, although fire reduced cheatgrass seed densities to $1/\text{ft}^2$ ($10/\text{m}^2$) in the first year; they rapidly increased to nearly $1000/\text{ft}^2$ ($10,000/\text{m}^2$) in post-burn years two to four (Young and Evans 1978). The study area was located on a cool/dry ecological site (8 to 10 inches [200 to 255 mm] of precipitation) in a degraded low-**seral** stage. Studies conducted on cool/moist soils also reported increases in cheatgrass cover during the second and third post-fire years; however, post-fire cheatgrass cover usually remained below 10% (Everett and Ward 1984; Cook and others 1994; Pyle and Crawford 1996; Chambers and others 2007; Davies and others 2007, 2008, 2009b; Miller and Heyerdahl 2008; Miller and others in preparation b). On a cool/moist Wyoming big sagebrush study site with 20 to 30% cover of native **herbs**, density and cover of cheatgrass were lower on the burned compared to unburned plots in the fourth year following disturbance (Davies and others 2008).

In the salt-desert, the adaptability of cheatgrass in this arid environment and its relationship with fire are less clear. Cheatgrass was rarely mentioned in the salt desert literature prior to the early 1980s (West 1994); it apparently became more common during the very wet years of the early 1980s. Also, elevated CO_2 levels may have facilitated an increase in cheatgrass abundance in these dry desert systems (Smith and others 1987; Ziska and others 2005). Meyer and others (2001) reported low resistance of salt-desert shrub to cheatgrass invasion. However, the process of invasion is slow and cheatgrass annual cover and biomass are highly variable, due to low and sporadic annual precipitation and limited seed carryover. On a shadscale community in western Utah, cheatgrass biomass increased with increased nitrogen availability and decreased with soil compaction and competition (Beckstead and

Augspurger 2004). In low density cheatgrass patches, disturbance that reduced soil compaction (authors created 0.2 inch [10 mm] holes in the soil surface) increased cheatgrass biomass 270% and density 85%. A major concern is the presence of cheatgrass genotypes that appear to be better adapted to salt-desert environments than more broadly adapted generalist genotypes that occupy historically invaded communities (Merrill and others 2012).

Cheatgrass significantly increased following wildfire in greasewood and black sagebrush/shadscale communities in northwestern Utah (Jessop and Anderson 2007) and fourwing saltbush and bud sage communities in northwestern Nevada (Haubensak and others 2009). Cheatgrass cover and density were low prior to the fire but substantially increased in all four communities following fire. However, cheatgrass density and cover were highly variable during the three years following fire, with abundance largely paralleling spring precipitation. In the black sagebrush/shadscale community cheatgrass cover was 3% in the dry spring increasing to 10.5% in the wet spring. Cheatgrass cover in the unburned plant community varied from <1% to just over 1% in the dry and wet spring, respectively. On the burned greasewood community, cheatgrass cover varied from 11% in the wet year to nearly non-existent in the dry year.

Long-term response–Annual fluxes in precipitation are associated with large parallel fluctuations in cheatgrass cover (West 1994; Anderson and Inouye 2001; West and Yorks 2002; Bates and others 2007a; Jessop and Anderson 2007). This variation over time makes it difficult to separate the effects of climate and time since fire on cheatgrass cover in the absence of control plots. One of the few long-term studies conducted at a single location that also included pre-treatment measurements reported that cheatgrass cover fluctuated between <1% to 20% between 6 and 19 years after a fire (West and Yorks 2002). During this period, cheatgrass cover increased in the wetter periods (up to 20%) and declined to near trace levels in the dry years. A second example of annual variation with precipitation occurred in a study where western juniper was removed by cutting. Cheatgrass cover increased from <1% prior to cutting to 15 and 20% in the **duff** and debris areas in the fifth year (1997), followed by a decrease to <5% 12 years (2003) after treatment (Bates and others 2007a). In this study, crop-year precipitation was above average during 1995 to 1997 and below average during 2001 to 2003, making it difficult to determine if the decline of cheatgrass was a result of time since treatment or available moisture. Therefore, it is important to consider annual precipitation when evaluating the cover of cheatgrass over time. With only one long-term study (West and Yorks 2002) that was conducted on the same location and that had both pre-treatment data and control plots, we have little information about the long-term trends of cheatgrass following prescribed burns. This is especially true with the effects of repeated burns on long-term trends in cheatgrass abundance. Long-term model simulations in central Washington indicated annual species, primarily cheatgrass, decreased in the first year following fire, rapidly increased 2.5- and 4.5-fold in the second and third years, and remained dominant up to 13 to 15 years, after which abundance declined due to increased dominance by rubber rabbitbrush, big sagebrush, and crested wheatgrass (Mata-Gonzalez and others 2007). However, the authors based the long-term response on studies conducted on considerably wetter ecological sites (mountain big sagebrush) in Montana than the warm/dry ecological sites in eastern Washington. Limited long-term studies suggest the response is highly variable. Melgoza and others (1990) reported that competition for moisture from cheatgrass remained high on warm (**mesic**) soils 12 years after fire in western Nevada. This pattern of succession is supported by several studies that replaced time for space (Barney and Frischknecht 1974; Skousen and others 1989) and the long-term study at a single study location (Hosten and West 1994; West and Yorks 2002). Across 28 burned study sites in west-central Utah, cheatgrass cover averaged 13% in the young burns and <3% in the older burns (>50 years), which had become dominated by pinyon and juniper

(Barney and Frischknecht 1974). However, in eastern California, cheatgrass was the most abundant plant 45 years after fire (Billings 1994). And on study sites with limited perennial grass cover (<3%), cheatgrass cover remained high up to 60 years post-burn (Koniak 1985). On warm soils with moist or dry **soil moisture regimes,** it has been proposed that communities dominated by invasive annuals are new steady **states** (Pellant and Hall 1994; Holmes and Miller 2010; Miller and others 2011). The persistence of cheatgrass will likely depend on ecological site characteristics, including composition and structure of perennial herbs and post-fire disturbance.

Other Grasses of Concern

There is considerably less information on fire effects on other invasive annual grass species. Most are highly flammable, competitive with seedlings of native grasses, and well adapted to fire (Young 1992; Mazzola and others 2011). In the early 1960s, medusahead was first reported in the Great Basin Region in Verdi, Nevada, and Susanville, California. There is an indirect relationship between medusahead density and density of perennial grasses (Davies 2008). Slow decomposition of medusahead litter results in an accumulation of litter and an increase in the potential of medusahead infested areas to burn (McKell and others 1962). Repeated fires favor the dominance of medusahead (Young and others 1971, 1972). Stubbs (2000) documented medusahead seedling emergence, survival, and reproduction on four Central Oregon western juniper/basin big sagebrush study sites. Medusahead seedlings growing on one- and two-year-old burns produced 43 seeds per plant while those on control plots produced 22 seeds per plant. However, seedling emergence and survival did not differ on burned or unburned sites (60 to 75% emergence; 85 to 95% survival). North Africa grass (also known as wire grass) (*Ventenata dubia*) has recently expanded into the Great Basin Region, and is most noticeable in recent burns. It was first identified in Idaho in 1957 and by the mid-1980s it was present throughout the Pacific Northwest (Prather 2009).

Ecological Site Characteristics and Resistance to Invasive Annual Grasses

Possibly the most important ecological site factors that determine the resistance to invasive annual grasses in the Great Basin Region are cover of perennial grasses prior to and immediately following a burn, thermal regime, and water availability (figs. 12, 32A and B) (Beckstead and Augspurger 2004; Chambers and others 2007; Davies 2008; Davies and others 2012; Blank and Morgan 2012; Miller and others in press). Chambers and others (2007) concluded that the resistance of an ecological site to the invasion of cheatgrass was negatively related to temperature at higher elevations and positively related to soil water availability at lower elevations. The cover of invasive species also is negatively correlated with native plant cover but not species richness (Anderson and Inouye 2001; Chambers and others 2007; Condon and others 2011; Blank and others 2012).

Chambers and others (2007) concluded that **soil temperature regimes** and water availability are two of the most important ecological site characteristics in determining resistance to invasive annual grasses, followed by plant composition on the ecological site at the time of disturbance (fig. 12, 14). The studies we reviewed strongly support their conclusions. Research conducted on cool or cold soil temperature regimes report limited increases in cheatgrass cover in the first few years following fire (Everett and Ward 1984; Cook and others 1994; Pyle and Crawford 1996; Chambers and others 2007; Davies and others 2007, 2008, 2009b; Miller and Heyerdahl 2008; Miller and others in preparation b). However, warmer exposures can be a concern even on cool soils (Koniak 1985 Cook and others 1994; Condon and others 2011). In Wyoming at 7900 ft (2400 m), cheatgrass cover three years after a fire was considerably greater (20%) on the southwest aspect compared to
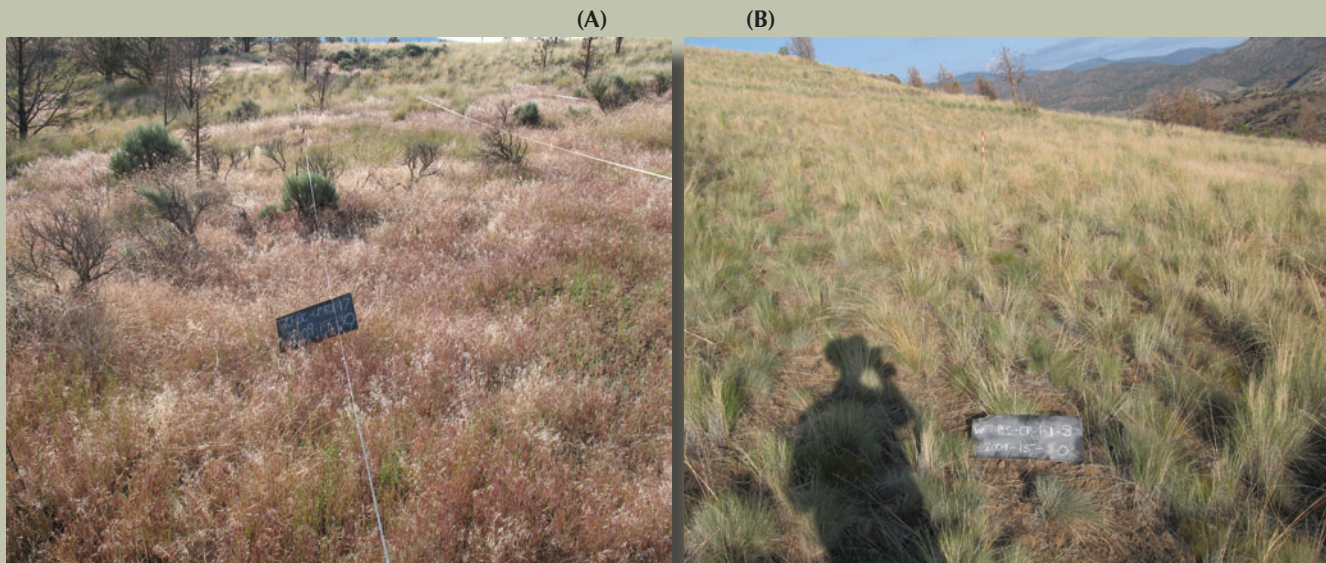
Figure 32. These two warm/dry (10 to 12 inch [250 to 300 mm] PZ) sites were burned by the same prescribed fire and are located approximately 300 yards (275 m) from one another. Potential vegetation is basin big sagebrush and bluebunch wheatgrass. Prior to the burn **(A)** the west aspect was dominated by Sandberg bluegrass and **(B)** the north aspect was dominated by bluebunch wheatgrass. Both sites had < 1% cover of cheatgrass prior to the burn. The prescribed fire was low to moderate severity. In the second post-fire growing season, **(A)** was dominated by cheatgrass and **(B)** was dominated by bluebunch wheatgrass with <1% cheatgrass cover. The consequence of different site characteristics (e.g., aspect and soil texture) and species composition resulted in different levels of resilience and resistance to invasives, which produced different successional trajectories (central Oregon; photos from the SageSTEP project).

easterly aspects (<3%) (Cook and others 1994). Wyoming big sagebrush was present on the southwest aspect and mountain big sagebrush on the easterly aspects. Studies conducted on warm/dry and warm/moist soils frequently reported large increases in cheatgrass following fire (Young and Evans 1978; Koniak 1985; Young and Miller 1985; Hosten and West 1994; West and Yorks 2002; Miller and others in preparation b).

Composition of native perennial grasses and forbs is also a primary characteristic that influences the post-fire response and persistence of invasive species. On a warm/dry ecological site in a relatively non-degraded **seral** stage, cheatgrass cover increased to 40 to 50% during the second, third, and fourth post-fire years (West and Yorks 2002). However, during the next 12 years, cheatgrass cover declined and fluctuated between <1 to 20% with annual precipitation. In central Nevada, ecological sites in a degraded seral stage prior to tree removal resulted in moderate increases in cheatgrass on cool soils following treatment (Baughman and others 2010) and large increases on warm-cool soils in western Nevada and northeastern California (Evans and Young 1985, 1987). Many studies reported that the increased cover of perennial **herbs** had an inverse relationship with cheatgrass cover and biomass (Beckstead and Auspburger 2004; Chambers and others 2007; Condon and others 2011; Mazzola and others 2011). Although ecological site characteristics (thermal and moisture regimes) had greater effects on cheatgrass establishment than fire or reduced competition from resident perennial grasses, cheatgrass biomass and seed production were affected by both fire and removal of native perennial grasses (Chambers and others 2007). Removal of native perennials alone resulted in a 2- to 3-fold increase in cheatgrass; fire alone produced a 2- to 6-fold increase; and the combination of perennial herb removal and fire resulted in a 10- to 30-fold increase in cheatgrass.

Summary

The response of cheatgrass following fire in sagebrush communities varies from no response to large increases. The largest increases most frequently occur in the second and third year following fire and on soils with warm temperature regimes. Increased abundance is closely associated with warm soils and the degree of competition from resident perennial grasses. Thermal regimes, moisture availability, and plant composition at the time of the burn are possibly the three most important ecological site characteristics that determine resistance to invasion. Increases in cheatgrass on cool soils following fire or fire surrogate treatments is usually low compared to increases on warm soils. When selecting management units to be treated and the type of treatment to be used, managers should carefully consider elevation, aspect, and composition and structure of current vegetation.

Literature evaluating long-term succession of invasive annual grasses and the permanency of new steady states dominated by these grasses is limited. In addition, the majority of these studies: (1) substituted space for time, (2) did not collect pre-treatment measurements, or (3) did not have legitimate controls (where adjacent study sites were closely evaluated for similarity). And no studies have looked at the effects of repeated fires on long-term succession. Limited work indicates that cheatgrass increases and remains abundant from 5 to 12 years after fire followed by a gradual decline. However 15 years after fire, cheatgrass was still a major component of the plant community (10 to 15% cover) during wet years in Utah even though perennial grass cover remained high. The long-term trend in cheatgrass abundance following fire will likely depend on: (1) the ecological sites thermal and moisture regimes, (2) the cover of perennial grasses at the time of the burn, (3) the absence or occurrence of a second fire, (4) the re-establishment of woody species, and (5) post-fire grazing management.

Biological Soil Crusts

Introduction

Most biological soil crusts grow in the open interspaces between shrubs and herbs in sagebrush steppe and salt-desert communities; however, some species can be associated with the under-canopy environment of shrubs and grasses. Also known as cryptogamic, cryptobiotic, microbiotic, or microphytic soil crusts (Harper and Marble 1988; West 1990), biological soil crusts are a complex of green algae, lichens, mosses, microfungi, cyanobacteria, and other bacteria (West 1990; Belnap and others 2001). Biological soil crusts influence ecosystem function, including soil stability (Hilty and others 2004), nitrogen fixation (Terry and Burns 1987; Warren and others 2009), and infiltration (Belnap and others 2005, 2001) although effects vary with composition.

Biological soil crust cover typically increases along an environmental gradient of increasing aridity and is inversely related to vascular plant cover. Crusts are usually more abundant at lower elevations in Wyoming big sagebrush, salt-desert, and desert shrub communities (warm [mesic] communities) than in higher elevation mountain big sagebrush and mountain brush (cool [frigid] communities) (fig. 33) (Kaltenecker and others 1999; Miller and others in preparation b). In east-central Idaho, Kaltenecker and others (1999) reported 60% cover of biological soil crusts in Wyoming big sagebrush compared to 35% in mountain big sagebrush. In the Great Basin (across a four-state area), cover of biological soil crust was 25% on warm and 4% on cool soil temperature regimes (Miller and others in preparation b).
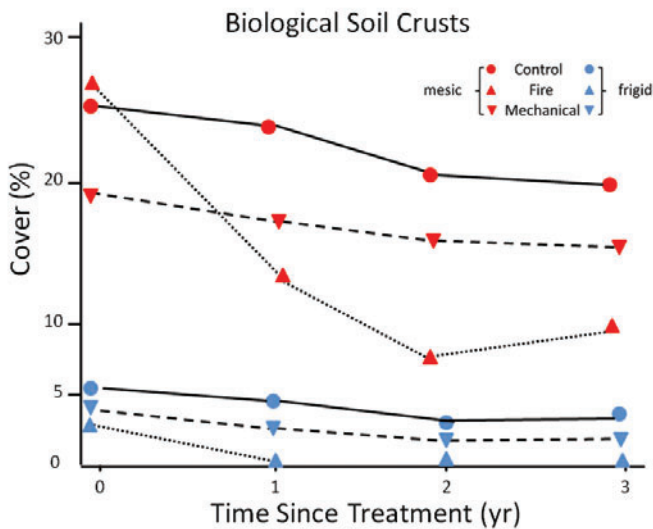
Figure 33. Average changes in biological soil crust (mosses and lichens only) cover between two soil temperature regimes over a three-year period on 11 sites across the Great Basin Region following: (1) control, (2) prescribed fire, and (3) cutting and leaving juniper and pinyon trees. Red symbols = warm (mesic) and blue symbols = cool (frigid) soil temperature regimes (from Miller and others in preparation b).

Fire Effects

Fire generally results in a significant and immediate decline in biological soil crust cover in sagebrush steppe and salt-desert communities in the Great Basin Region (table 5; fig. 33) (Johansen 2001; Johansen and others 1984, 1993; Johansen and Rushforth 1982; West and Hassan 1985; Terry and Burns 1987; Hilty and others 2004; Ponzetti and others 2007; Haubensak and others 2009; Miller and others in press). In the Snake River Plain, burning resulted in substantially reduced diversity and richness of crust taxa (Hilty and others 2004). Although cover of short mosses increased, cover of lichens and tall mosses growing on shrub hummocks was reduced. The number of species of lichens across treatments was 35 in the control, 25 in the burned and seeded, and 5 in the burned and unseeded. There was no difference in the number of mosses among treatments. Algae composition did not change following fire; however, there was a significant reduction in density (Johansen and Rushforth 1982; Johansen and others 1993).

The amount of reduction in crust cover is related to fire intensity (Johansen and others 1993; Warren and others 2009) and composition of the biological soil crusts (Hilty and others 2004). Low intensity fires only lightly burned biological soil crusts, which continued

Table 5. Differences in % cover of biological soil crusts between burned and adjacent unburned plots.

Location	Site	Unburned (%)	Burned (%)	Reference
NW NV	Salt-desert	40	13	Haubensak and others 2009
	Salt-desert	23	5	
OR, UT	Mesic soils	27	10	Miller and others in press
OR, NV, UT	Frigid soils	3	<1	
ID	Wyoming big sage all crusts	66	35 (unseeded) 46 (seeded)	Hilty and others 2004
	mosses	35	1.5	
UT	Wyoming big sage & Utah juniper (interspace-bryophytes)	13	2	Warren and others 2009
UT	Wyoming big sage & Utah juniper (interspace-lichen)	6	4	Warren and others 2009
UT	Wyoming big sage & bluebunch wheatgrass	11	<1%	West and Hassan 1985

to function following fire (Johansen and others 1993). Warren and others (2009) found no change in lichen cover in the interspace where fire intensity was low and the burn spotty. However, lichen cover was significantly reduced following a high intensity fire in Montana grasslands (Antos and others 1983). The common tall moss tortula, which is located beneath shrub canopies, burned at high intensities resulting in a near total loss (Warren and others 2009).

Recovery

The rate of recovery for biological soil crusts following fire is highly variable, ranging from only a few years to more than 30 years. This wide range in variability can likely be attributed to fire intensity, composition of biological soil crusts and vascular plants at the time of the burn, and post-fire climate conditions. Although nitrogen-fixation by biological soil crusts was significantly lower in the second year following fire compared to adjacent unburned plots in salt-desert communities in Utah, there were no differences in post-fire years three, four, and five (1982 to 1984) between burned and unburned (Terry and Burns 1987). The authors attributed the rapid recovery of nitrogen-fixation by biological soil crusts to very wet conditions in 1982-1984. Warren and others (2009) reported reductions in bryophyte cover immediately following a prescribed fire in 2006 but found no differences between burned and unburned plots in 2007 and 2008. However, others have reported limited recovery of biological soil crusts three or more years following fire (Johansen and others 1984; West and Hassan 1985; Haubensak and others 2009; Miller and others in press). Algal communities in two separate burns in Utah had not recovered within three years of fire, although recovery was complete in five years in one study (Johansen and others 1984). In salt-desert communities, biological soil crust cover was five-fold greater in unburned compared to adjacent burned plots (Haubensak and others 2009). However, cover of exotic grasses also was five-fold greater on the burned plots. Although we found no long-term studies evaluating recovery of biological soil crusts, Callison and others (1985) reported that biological soil crusts had not recovered in some blackbrush (*Coleogyne ramosissima*) communities in Utah 30 years after fire.

Fire, Exotics, and Biological Soil Crusts

There is very strong evidence that cover and species richness of biological soil crusts are inversely related to cheatgrass cover (Rosentreter 1994; Kaltenecker 1997; Burnham 2001; Ponzetti and others 2007; Haubensak and others 2009). In addition, recovery of biological soil crusts following fire is slower to non-existent on ecological sites where cheatgrass is co-dominant or dominant (Ponzetti and others 2007). As previously stated, cheatgrass cover was five-fold greater and biological soil crusts 80% less in burned compared to unburned plots five years after fire in a salt desert community (Haubensak and others 2009). In a Wyoming big sagebrush community in the Columbia River Basin, Ponzetti and others (2007) reported good recovery of biological soil crusts in communities with good native perennial grass cover and low cover of cheatgrass. In poor condition areas in the Snake River Plain where fire resulted in an increase in exotic annual grasses and forbs, cover of biological soil crusts was significantly greater on burned plots that were seeded to perennial grasses (species not reported) compared to plots that were burned and not seeded (Hilty and others 2004).

Summary

Although the number of studies evaluating the impacts of fire on biological soil crusts is limited, there is strong evidence that the short-term effect of fire reduces cover and species

richness. There is fairly strong evidence that there is a negative relationship between biological soil crust cover and richness with cheatgrass cover (and likely other exotic annuals). However, we have limited information on recovery after the first five years post-fire, and we could find no studies evaluating fire effects on biological soil crusts 10 or more years after fire in sagebrush or salt desert communities. Limited information suggests recovery is largely linked to fire intensity, vascular plant composition at the time of fire (especially exotic annuals), post-fire climate, composition of the biological soil crusts, and ecological site characteristics.

Bare Ground

Bare ground increases immediately following fire but usually decreases to pre-burn values within two to four years (Barney and Frischknecht 1974; Bates and others 2009, 2011; Miller and others in press) (fig. 34). The increase in bare ground in the first post-fire year results from reduction of woody plants, deep-rooted perennial grasses, biological soil crusts, limited response of invasives, and consumption of litter. In Wyoming and mountain sagebrush communities encroached by pinyon and juniper, bare ground increased from 28% (pre-burn) to 43% in the first post-fire year but returned to near pre-burn levels by the second post-fire year (Miller and others in press). In two mountain big sagebrush communities that were burned to control western juniper, Bates and others (2011) reported bare ground increased from pre-burn levels of 20 to 30% to 60 to 75% in the first year after the burn. Although bare ground remained higher on burned than control plots in the third post-fire year, it had declined to 45% due to increases in perennial grasses, perennial and annual forbs, and litter. In a mountain big sagebrush community in northwestern Nevada, bare ground increased from 8% to 98% immediately after fire (Pierson and others 2008a). By the first post-fire growing season, bare ground declined to 66% and by the third year it declined to 22%.

Figure 34. A cool/moist (12 to 14 inch [300 to 350mm] PZ) site once dominated by mountain big sagebrush, bluebunch wheatgrass, and Idaho fescue. Pre-fire vegetation was a woodland in late Phase II to early Phase III (late-mid to late woodland succession) with high fuel loads that resulted in a moderate to high severity fire. Mortality of perennial grasses is often >80% in dense patches of trees compared to perennial grass survival of >80% in more open stands. However, recovery potential and resistance to invasives may be relatively high due to site characteristics (cool/moist soil regime) and residual but partially depleted native perennials (southwest Idaho; photo by Rick Miller).

Summary

Bare ground can be expected to increase immediately after a fire event and remain significantly higher than pre-burn levels during the first post-fire growing season. However, bare ground usually declines rapidly, approaching pre-burn levels by the fourth growing season. The primary characteristics determining the decline in bare ground are those that drive plant succession and productivity (fig. 15).

Litter

Litter is significantly reduced immediately following fire but litter cover increases to pre-burn levels within two to three years (Beck and others 2008; Davies and others 2009a; Bates and others 2011; Miller and others in press). However, we know little about the rate of recovery of litter biomass to pre-burn levels. Litter can decrease surface soil temperatures and increase surface soil moisture, which enhances seedling germination and establishment of both native and invasive species (Evans and Young 1970; Chambers 2000). From a hydrologic prospective, litter thickness influences surface water storage, runoff, and infiltration (Pierson and others 2010). Litter build-up occurs primarily beneath shrub and tree canopies, which can be consumed during a **high severity fire**. However, accumulation of litter also can have negative impacts on plant community recovery. Some invasive grasses can produce rapid buildup of dense litter mats (e.g., medusahead), which can reduce recovery of biological crusts (Hilty and others 2003) and increase the potential for more frequent fires (McKell and others 1962). The reduced recovery of biological crusts may be attributed to reduction of light resulting from the accumulation of litter (Ponzetti and others 2007). On a mountain big sagebrush community in northwestern Nevada, litter cover increased from <1% immediately after fire to 18, 29, and 36% in post-fire years 1, 2, and 3, respectively (Pierson and others 2008a). By year three, litter was near pre-burn levels (39%).

Summary

Although only a limited number of studies have evaluated litter, it appears that litter cover quickly returns to pre-burn levels within the second or third year following fire.

Woodlands

Expansion and Development

Juniper and pinyon woodlands occupy approximately 44.6 million acres (18 million ha) in the Intermountain West (Miller and Tausch 2001). These woodlands are commonly associated with sagebrush communities forming a mosaic of shrublands and woodlands across the landscape. Juniper is more widespread throughout the region than pinyon, probably due to its higher tolerance to drought and cold (Malusa 1992; Miller and Wigand 1994). The most common juniper and pinyon species in the Great Basin Region are Utah and western juniper and singleleaf pinyon (Charlet 1996; West 1999; Miller and others 2005). Rocky Mountain and Sierra juniper (subspecies of western juniper) also occur in the region but are not as abundant. Prior to Eurasian settlement in the mid to late 1800s, these woodlands were dynamic, expanding and contracting throughout the Holocene (last 10,500 years) largely a result of changing climate (warm/dry to cool/wet periods) and fire (Miller and Wigand 1994). Pinyon dynamics also were affected by insects and disease, which are often linked to climate. Since Eurasian settlement, juniper and pinyon woodlands have been

rapidly expanding into sagebrush plant communities throughout the Great Basin Region (Cottam and Stewart 1940; Tausch and others 1981; Knapp and Soulé 1998; Tausch and Nowak 1999; Miller and Tausch 2001; Azuma and others 2005; Johnson and Miller 2006; Weisberg and others 2007; Miller and others 2008). Rapid expansion in the mid to late 1800s coincided with the introduction of livestock and less **surface fire** (Burkhardt and Tisdale 1976; Tausch and others 1981; Miller and Heyerdahl 2008; Miller and others 2008; Miller and Rose 1999; Miller and Tausch 2001). Others have reported similar results for increases in density of ponderosa pine (primarily infill defined in the following paragraph) and density and area of juniper and pinyon in the Southwest (Savage and Swetnam 1990; Touchan and others 1995; Swetnam and Baisan 1996; Baisan and Swetnam 1997; Swetnam and others 2001). Expansion also coincided with the end of the Little Ice Age (Miller and Wigand 1994) and relatively wet conditions during the late 1800s (Arnold 1964; West and others 1978). During the 20th Century, woodland expansion in the Great Basin Region has been greatest in cooler and/or moister portions of the landscape (Johnson and Miller 2006; Weisberg and others 2007). This largely coincides with soil temperature/moisture regimes that are cool-warm/moist to cool/moist, which includes portions of black sagebrush and Wyoming big sagebrush communities occupying the cooler and moister end of their range. It also includes cool/moist mountain big sagebrush and low sagebrush communities (with moderately deep soils).

Today, we have three types of juniper and pinyon woodlands that can be differentiated by age structure and density: old-growth woodlands, infilled woodlands, and expansion woodlands. If management goals are to restore plant communities to their **reference state,** it is extremely important for managers to be able to identify the type of woodland to be treated. *Old-growth* woodlands that have persisted over centuries are found where ecological site conditions (soils and climate) and disturbance regimes are inherently favorable for pinyon and/or juniper (fig. 35) and where trees are the major component of the vegetation unless recently disturbed by a stand-replacement fire, mechanical clearing (Romme and others 2009), or a combination of drought, insects, and fire. *Infill* is the increase in tree density within an existing shrub-steppe where pre-settlement trees were previously of lower density (fig. 37). *Expansion* is the establishment of trees in places that were formerly non-woodland (Romme and others 2009) (fig. 36). Persistent or old-growth woodlands are typically found on shallow soils, which limit the production of understory fine fuels (West 1988; Waichler and others 2001; Miller and Heyerdahl 2008). These three types of woodlands can be identified based on the morphological characteristics of the juniper and pinyon trees (Miller and others 2005, 2007; Tausch and others 2009) and the presence or absence and density of large standing dead trees, logs, and stumps.

Although woodland expansion into sagebrush communities began in the second half of the 1800s, it is an ongoing process that is not synchronous and that varies spatially and temporally across the Great Basin Region (Miller and others 2008). In the Great Basin, woodland succession in shrub communities being encroached or infilled vary from early (**Phase I**, see Glossary for definition) to tree dominated phases (**Phase III**) (Miller and others 2005). The stage of woodland development has large implications for the capacity of an ecological site to recover (resilience) and to resist invasives following a fire event, **fire severity**, and the ability to use prescribe fire (figs. 38 A and B). Sagebrush cover and other associated shrubs decline with increasing tree dominance (Adams 1975; Tausch and West 1995; Miller and others 2000; Roberts and Jones 2000; Miller and Tausch 2001; Schaefer and others 2003). Perennial forb cover also has been reported to decline with increasing tree cover (Dhaemers 2006) as has perennial grass cover (Miller and others 2000). In western juniper, the decline or persistence of herbaceous vegetation as

Figure 35. This site is occupied by 500 to 1000+ year-old trees with an understory of bluebunch wheatgrass. Located on moderately deep to deep but very rocky soils, this site has a high density of cavity nesting birds. Morphological characteristics of the trees and community structure (large woody debris, lichen in the tree canopies, etc.) make it easy to distinguish this site from post-settlement woodlands (central Oregon; photo by Rick Miller).

trees becomes dominant (in the absence of over-grazing) is related to soil depth (Bates and others 2000; Miller and others 2000). Herbaceous vegetation is more persistent in moderately deep to deep soils (>20 inches [510 mm]). Although we believe this to also be true for Utah juniper and singleleaf pinyon pine, we could find no literature that has evaluated this relationship for woodlands with these two tree species.

The consequences of increasing tree dominance in sagebrush communities are: an increase in fuel load and change in fuel structure (Miller and Tausch 2001; Sabin 2008; Tausch 2009), changes in plant community composition and structure (Adams 1975; Tausch and West 1995; Miller and others 2000; Dhaemers 2006), an increase in aboveground carbon and nutrient pools (DeBano and Klopatek 1988; Everett and Thran 1992;

Figure 36. An old growth western juniper stand located on a ridge-top with shallow soils and limited fine fuels. A 500+ year-old tree is in the foreground. A young stand of trees (<100 years) is in the background and is expanding into a mountain big sagebrush community with moderately deep soils and fine fuels exceeding 600 lbs/acre (272 kg/ha). This type of encroachment often increases fire risk for old growth trees (northwestern Nevada; photo by Rick Miller).

Figure 37. Historic potential vegetation on this ecological site was mountain big sagebrush and Idaho fescue with a low density of scattered juniper trees. Infilling of western juniper during the past 100 years has changed the plant composition and community structure on this ecological site and has altered fire regimes (south-central Oregon; photo by Rick Miller).

Tiedemann and Klemmedson 2000; Rau and others 2009), and increased soil erosion (West 1999; Pierson and others 2007) (the impacts of fire on carbon and nutrient pools and hydrology are discussed in Section 4). As trees become dominant, changes in fuel biomass, structure, and continuity alter the historic **fire regime**. Aboveground biomass increases as much as 10-fold, primarily accounted for in the tree trunks and canopies (Miller and Tausch 2001; Sabin 2008; Tausch 2009). However, surface fuel abundance, aboveground fuel continuity, and **packing ratios** decline. This results in a shift in fire behavior from surface fires to **crown fires** that spread through the tree canopies and that require extreme conditions (high wind, high atmospheric instability, low humidity, and high temperatures). These conditions can result in **high severity fires** and lowered **resilience to disturbance** and **resistance to invasives**. Bates and others (in press) reported higher fire severity and significantly higher mortality of perennial grass in closed woodlands (Phase III) than in open stands where shrubs and **herbs** still either dominate (Phase I) or co-dominate with trees (**Phase II**). The decline in deep-rooted perennial grasses as a consequence of increasing tree dominance (Miller and others 2000) and/or mortality due to high fire severity (Bates and others in press) will reduce the ability of the ecological site to recover and resist invasive species (Chambers and others 2007; Davies 2008; Reisner and others 2013).

Post-Treatment Tree Establishment

An important management question following tree removal on an encroached sagebrush grassland community is how long it will take the trees to re-invade and become a co-dominant or dominant species. Specific questions related to re-colonization of trees are:

- Is there a seed source nearby?
- What are the ecological site factors that will influence the rate of re-colonization of trees?
- What is the density of trees that survived the treatment (this will differ between fire surrogate and prescribed fire treatments)?

Seed Dispersal and Establishment

Distance to a pinyon or juniper seed source is one factor that determines a communities' risk of tree encroachment (Suring and others 2005). Dispersal of juniper and pinyon seeds occurs by overland flow, gravity, and animal transport. Animal transport is the most important dispersal agent for moving seeds >6.6 ft (>2 m). Pinyon pine seeds are primarily dispersed by scatter-hoarding birds and mammals that cache seeds in the soil (Chambers and others 1999a, 1999b). However, juniper seed is primarily scattered by frugivorous birds and mammals that defecate the seeds. Most birds have limited gut retention times after feeding on juniper seed, which limits the distance of dissemination (Schupp 1993; Chambers and others 1999a, 1999b). A study in Idaho reported that berry-eating birds commonly had small home ranges. Juniper seed dissemination was approximately 165 to 230 ft (50 to 70 m) from the seed tree (Strand 2007). American robins, a primary consumer of western juniper fruits (Lederer 1977), fly an average of 145 ft (44 m) to a post-foraging perch where they defecate the seed (Chavez-Ramirez and Slack 1994). Seed sources within 300 ft (91.5 m) of a sagebrush community greatly increase the risk of tree encroachment. However, birds can disperse seeds up to 3 miles (5 km) (Vander Wall and Balda 1977, 1981), placing sagebrush communities with adequate moisture and temperature conditions for tree establishment at some risk (Suring and others 2005). Seed-caching rodents also play an important but little studied role in juniper dispersal. Rodents are important dispersers of Utah juniper (Schupp 1993; Chambers and others 1999a), and probably also disperse seeds of western juniper.

Juniper seeds are initially dormant immediately following their maturation (Johnsen and Alexander 1974). Breaking seed dormancy for Utah, Rocky Mountain, and possibly western juniper can occur with warm stratification (exposing seed to varying warm conditions) for 45 to 240 days followed by a similar period of cold stratification (Johnsen and Alexander 1974; Van Haverbeke and Comer 1985; Rietveld 1989). However, Young and others (1988) reported no response from Utah or western juniper from these treatments. These authors found that germination was enhanced by prolonged cool/moist stratification (subjecting seeds to both cool and moist conditions). The effects of stratification are often cumulative from year to year. Seeds of juniper are long lived with extended dormancy, resulting in highly persistent seed banks (Chambers and others 1999a, 1999b; Bonner 2008). In contrast, pinyon seeds are short-lived, exhibit little dormancy, and germinate primarily in the spring following seed dispersal (Chambers and others 1999b; Chambers 2001).

Greater juniper seedling establishment and growth have been reported to occur beneath shrubs compared to shrub-interspaces (Burkhardt and Tisdale 1976; Eddleman 1987; Miller and Rose 1995; Wangler and Minnich 1996; Soulé and Knapp 2000; Soulé and others 2004). Enhanced establishment and growth beneath shrub canopies may be attributed to a disproportionate amount of seed dropped by perching birds and/or more favorable growing conditions, especially on ecological sites with warm (mesic) soils (Chambers and others 1999b; Roberts and Jones 2000; Miller and others 2005). Pinyon seedlings require a nurse plant or protected micro-site for establishment and can exhibit significant predation from small mammals (Chambers 2001).

Treatment Longevity

Longevity of treatment before conifers begin to dominate a community depends on: (1) ecological site characteristics, especially productivity; (2) distance to seed source; (3) seed density in the seed bank, which is influenced by both outside seed sources and age structure of the stand treated; and (4) the density of live trees remaining following treatment. The optimal time to treat or re-treat will depend on stand structure (relative

abundance of trees, shrubs, and **herbs**), which influences fuel abundance and structure and community **resilience**. Re-establishment of trees often takes longer following a fire because fire is usually more effective at removing small trees than a fire surrogate treatment.

With a nearby seed source, juniper recruitment can be initiated within the first 10 years following fire (Miller and Heyerdahl 2008). Seed density in the seed bank also effects re-establishment, which is largely influenced by the presence of reproductive trees (>45 years old) in the pre-treatment community. However, establishment often accelerates in the second decade following fire. This may partially result from newly established shrubs that serve as nurse plants that can enhance tree establishment (Miller and Rose 1995; Wangler and Minnich 1996). As juniper trees mature (>50 years), they become more difficult to kill under moderate fire conditions (Burkhardt and Tisdale 1976; Miller and Rose 1999). However, pinyon is intolerant of fire at all life stages partially due to thin bark, which exposes the cambium to heat and pitch on the outer trunk and increases flammability of the tree at the base. As trees increase in size, shrubs (ladder fuels) immediately adjacent

(A)

(B)

Figure 38. (A) Woodland expansion and infill in the Monitor Mountains of central Nevada (photo by Jeanne Chambers). Changes in fuel structure, primarily a decline in fine surface fuels, decrease the potential to burn under moderate weather conditions. The increase in aboveground biomass and the requirement for severe weather conditions to carry a fire greatly increases the potential for a high severity fire. **(B)** The structure of closed woodlands (Phase III) typically requires extreme weather conditions to carry a fire. A high severity fire in the Phase III woodland shown resulted in nearly 100% mortality of perennial herbs, loss of soil organic matter in the upper surface, and formation of hydrophobic soils. High fire severity also resulted in a large loss of aboveground nitrogen through the near complete consumption of surface organic matter and tree foliage, which contains the highest concentration of nitrogen stored in the tree (Milford Flat fire western Utah; photo by Rick Miller).

to the tree canopy begin to thin out and die due to competition from the trees. This largely disconnects the tree canopy fuels from the fine surface fuels. As fine surface fuels decline and become disconnected from the tree canopies, more extreme weather conditions are required to spread a fire through the tree canopy. This shift in fuel abundance and structure (e.g., **packing ratio**) can change the **fire regime** on an ecological site to less frequent but more severe. Tree canopy growth for both western and Utah junipers is relatively slow the first 45 to 50 years followed by an accelerated rate of expansion for the next 50 to 70 years unless hindered by competition from neighboring trees (Barney and Frischknecht 1974; Tausch and West 1988; Miller and Tausch 2001; Miller and others 2005). In southeastern Oregon and southwestern Idaho, the development of western juniper woodlands from the time of tree establishment to dominance (shift from **Phase I** to **Phase III**) varied from between 60 to 80 years on cool/moist ecological sites to >125 years on warm/dry ecological sites (fig. 39) (Johnson and Miller 2006). In northeastern California, western juniper dominated mountain big sagebrush communities within 85 to 90 years (Miller and Heyerdahl 2008). The age and stand structure of pinyon and/or juniper woodlands in Nevada and Utah suggests that their stand development probably requires a similar amount of time (Miller and others 2008) on ecological sites with comparable soil temperature/moisture regimes. Encroached sagebrush communities in Utah developed into closed canopy woodlands within 85 and 90 years (Barney and Frischknecht 1974).

Miller and others (2011) used tree growth rates and changes in composition and structure of different vegetation layers (fuel composition and structure) to approximate the potential fire return intervals required to maintain shrub communities. Based on their assumptions, woodlands are more likely to develop when individual fire return intervals (not mean return interval) exceed 50 years (fig. 40). Once this occurs, a fuels **threshold** is crossed and tree dominance increases, altering surface fuel biomass and continuity to the point that surface fuels can no longer support a **surface fire**. This significantly lengthens the fire return interval. Persistent woodlands most frequently occupy ecological sites that cannot produce adequate surface fuels to carry a fire (usually a result of soil conditions). Other factors besides fuels that influence fire regime are topography, isolation (often a result of topography), ignition source (season and frequency), and climate (fig. 16). Fire return intervals in persistent woodlands are usually measured in centuries (Waichler and others 2001; Romme and others 2009).
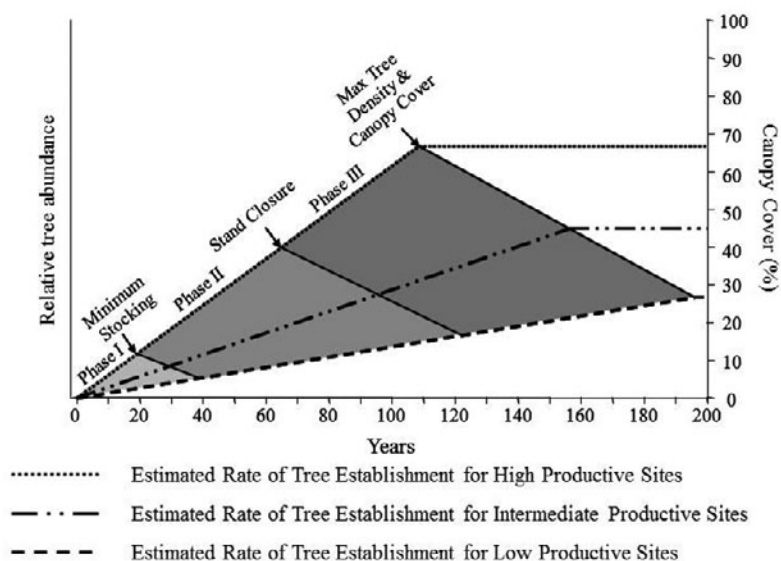
Figure 39. The hypothesized time periods from initial tree establishment (early phase I) to minimum stocking adequate for development of Phase III, and estimated maximum potential for tree density and cover for woodlands developing on ecological sites with varying productivity due to different moisture and temperature regimes (from Johnson and Miller 2006).
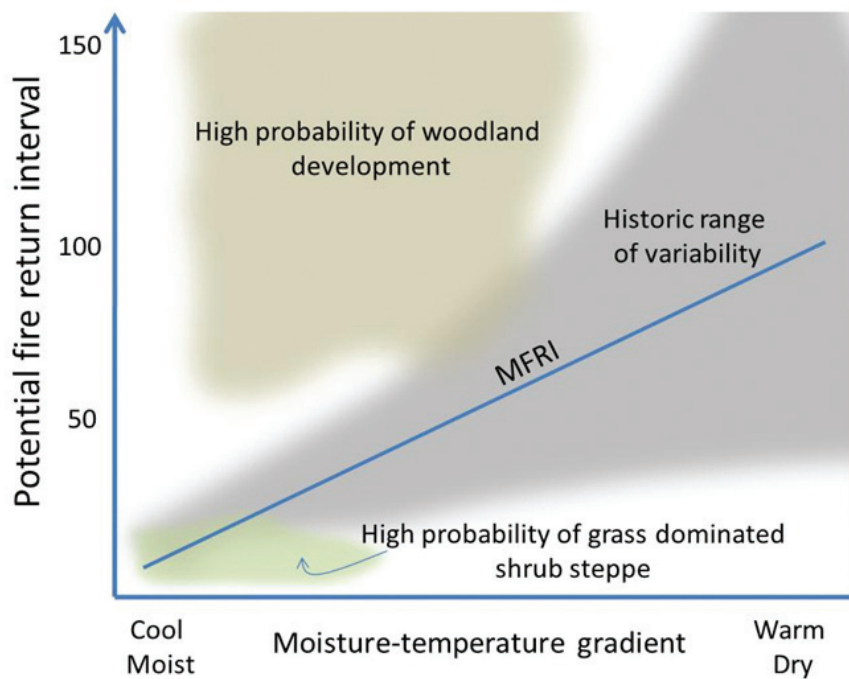
Figure 40. Conceptual model illustrating the range of historic potential mean fire return interval (MFRI) and historic range of variability (HRV) as it changes across a temperature and moisture gradient for sagebrush steppe (gray), persistent woodland (brown), and grassland (green). Fire frequency is a function of moisture and temperature, which influences fuel structure, composition, and biomass in addition to ignition. Persistent vegetation that occupies the gray area would likely be a sagebrush herbaceous mix; the relative abundance of each species is dependent on time since fire (from Miller and others 2011).

When fire or fire surrogate treatments kill only a portion of the trees, tree dominance can re-establish rapidly, often faster than previously reported estimates. In comparing woodland succession following chaining and prescribed fire at four different locations in Nevada, Bristow (2010) reported that 40 and 50 years following chaining tree canopy cover was 20 to 23% but it was only 0.3% in an adjacent 33-year-old burn. On chained plots, where survival of small trees was high, trees began to dominate the community within 15 years of treatment (Tausch and Tueller 1977). A large portion of small trees (<30 to 40 years old) will survive chaining (Vallentine 1971; Tausch and others 2009), and trees shorter than the sagebrush canopy (<3.3 ft [1 m]) are easily missed with cutting (Miller and others 2005). Fire is more effective for removing small trees, is usually cheaper than mechanical treatments, and can be more effectively applied to large land areas. However, mechanical treatments have several advantages over fire: (1) high degree of control of treatment boundaries and selection of specific trees for removal; (2) higher retention of the shrub canopy; (3) reduced risk in wildland-urban interface areas; (4) treatment can be applied nearly year-round if access is not constrained by weather, road conditions, special land designations, and fire conditions that restrict equipment or chain-saw use; and (5) less potential for releasing invasives (Miller and others 2005, 2007, in press; Tausch and others 2009; Chambers and others in review).

Summary

Rate of woodland re-establishment into sagebrush communities will be closely associated with ecological site characteristics, proximity to a seed source, and density and age of trees surviving the treatment. These variables will interact, resulting in a high level of variability in treatment longevity across landscapes. Where a juniper or pinyon seed source is >3 miles from a community, the probability of encroachment is low. Seed rain will be greatest within 300 ft (91.5 m) of the seed source. If a seed source is available, the literature indicates that it takes 60 to 80 years for the tree canopy to become the dominant layer on wetter ecological sites and 125 years on drier ecological sites. Comparisons (evaluation of paired plots) between fire and fire surrogate treatments are limited. However, limited work indicates woodland development will occur over a shorter period of time when fire

surrogates (chaining and cutting) are used compared to fire if small trees remain in the plant community.

Fire Severity

Fire severity is the magnitude of the effect that fire has on the total environment (van Wagtendonk 2006), including ecological processes, soil, flora, and fauna. It is the degree to which an ecological site has been altered or disrupted by fire. In forestry, it is usually measured as percent tree mortality and on rangelands sometimes as percent sagebrush kill. However, Keeley (2009) suggested fire severity is better measured as the consumption of organic matter and color of the ash. We believe Keeley's definition is a more useful measure of fire severity on rangelands. We use this approach in addition to other indicators of fire severity shown in table 2. Fire *intensity* is the amount of energy (heat) released in a fire and it is related to the amount and rate of surface fuel consumption. Fire *severity*, as it relates specifically to vegetation, is usually a function of intensity, residence time or duration, and plant morphology and phenology.

Fire severity has been closely related to mortality of perennial grasses in the Great Basin Region (Conrad and Poulton 1966; Bates and others 2011, in press). For this synthesis, we are interested in the fire characteristics that cause plant mortality. Unfortunately, very few studies in the Great Basin Region have addressed fire severity or collected fire behavior data.

Regeneration of aboveground shoots of perennial herbs occurs at the buds (meristematic tissue) located just above or below the soil surface. The amount of heat generated in this zone depends on the kind, distribution, and structure (e.g., compaction and size) of live and dead fuels, topography and micro-relief, weather, and soil moisture and texture (Wohlgemuth and others 2006). It also depends on the type of fire—head, backing, surface or crown fire—which influences residence time and the amount of heat near the soil surface. Maximum soil surface temperatures for fires in sagebrush shrublands typically range from 350 to 800 °F (178 to 293 °C) (Wright and Klemmedson 1965). On a fall prescribed fire in a mountain big sagebrush-bitterbrush community, temperatures at 2.5 and 0.3 inches (64 and 7.6 mm) below the soil surface peaked at 200 °F (93 °C) and 500 °F (260 °C), respectively. In contrast, soil temperatures at or near the surface were considerably cooler in two prescribed fires in eastern Washington (Beckstead and others 2011). They reported temperatures just below the surface (0.4 to 1.2 inches [10 to 30 mm]) of 122 to 167 °F (50 to 75 °C) and maximum surface temperatures reaching 250 to 300 °F (120 to 145 °C). Temperatures vary with heterogeneity of surface fuels, which is highly variable in shrubland and woodland communities (e.g., interspace versus beneath shrub or tree canopy) (Young and Evans 1977; Hassan and West 1986; Korfmacher and others 2003; Bates and others 2011). In the tree or shrub interspace, soil surface temperatures can range from <175 to 400 °F (<79 to 204 °C), and in tree canopy litter 1166 to 1300 °F (603 to 704 °C) (Bates and others 2011). Korfmacher and others (2003) reported significantly higher surface temperature beneath sagebrush, which averaged 718 °F (381 °C) compared to bare ground and under grasses, which averaged 590 to 585 °F (310 and 307 °C), respectively. Consumption of seeds in the seed bank is potentially high in the hot spots beneath shrubs (Young and Evans 1977; Hassan and West 1986). In two prescribed fires, Beckstead and others (2011) reported average soil surface temperatures between 248 to 293 °F (120 to 145 °C), which consumed 85 to 98% of seed in the seed bank.

Fire intensity is closely related to flame length; the longer the flame length the greater the intensity (Byram 1959). Fire intensity increases as shrubland communities shift to pinyon and juniper woodland communities. One study found that flame lengths in a pinyon and juniper **crown fire** can range from 12 to 36 ft (3.7 to 11 m), compared to less than 12ft (<3.7 m) in a shrubland fire (Dicus and others 2009). Woodlands also require more severe weather conditions to burn as a result of lower abundance and continuity of surface fuels. The combination of more severe weather and greater biomass in the tree canopy can potentially increase **fire severity** (figs. 38A and B). For example, Dicus and others (2009) found that winds >15 mi/hr (39 km/hr) were required to carry a fire through a singleleaf pinyon woodland canopy (or crown) that contained 3 t/acre (5.4 t/ha) of available canopy fuel load. (Note: Dicus and others [2009] did not provide percent tree canopy cover, but they mentioned that the woodland was made up predominantly of pinyon.)

The duration (residence time) that a point on the surface is exposed to heat by a flame and/ or smoldering fire has a large effect on soil temperatures just below the surface. DeBano and others (1998) reported significantly lower temperatures at 0.5 to 2 inches (12.5 to 50 mm) below the surface than occurred just above the surface (DeBano and others 1998). Most plant tissue dies at around 140 °F (60 °C) when exposed for 1 minute (Agee 1993). In addition to wind, duration is influenced by fuel biomass and structure, which affects the amount and rate of energy release. Large heavy fuels (e.g., tree logs or piled fuel), and **duff** layers where fine fuels are very compact (high **packing ratio**, g/cm^3), limit oxygen and will continue to burn or smolder long after the flaming front has passed.

Plant sensitivity to heat is influenced by phenology, type of tissue (meristematic), moisture content, and morphology. Location of the growing points is also an important attribute that influences the amount and duration of exposure to heat. Buds that produce reproductive and vegetative shoots on bunch grasses are located just above or below the soil surface, depending on the species. Greater mortality of Idaho fescue than bluebunch wheatgrass was attributed to the bud tissue being located at or just above the soil surface, compared to bud tissue in bluebunch wheatgrass, which is located just below the surface (observed but not measured; Conrad and Poulton 1966). Severity also has been associated with accumulation of plant material at the plant base (Wright 1971; Davies and others 2009b, 2010). Plant size also has been associated with fire sensitivity. Damage to the crowns of larger plants, which retain more fuel, is usually greater than damage to the crowns of smaller plants (Wright and Klemmedson 1965). Fire sensitivity of perennial forbs largely depends on the location of the growing points (meristematic tissue), which can be above or below ground, depending on the species (table 4; see Perennial Forbs Species and Morphology section).

Season of Burn

One important question is: do plants respond differently to spring, summer, fall, and winter burns? The majority of wildfires occur in the summer and early fall, and prescribed burning usually occurs in fall or winter in the Great Basin Region. However, there are only a few studies that compare season of burning. Most of the information that evaluates season of burn has been conducted in small plots using **burn barrels** (Wright and Klemmedson 1965; Wright 1971; Britton and others 1990) and the results are not always consistent (Wright and Klemmedson 1965). Important factors that influence fire severity on plants are phenology, species morphological characteristics, weather conditions at the time of the fire, and fuel characteristics. In the Great Basin Region, the sensitivity of live plant tissue to heat generally decreases from May to September (Wright and Klemmedson 1965).

Fires during the summer, compared to spring or fall, typically occur under hotter weather conditions and, compared to spring burns, have greater biomass of dead fine fuels. These conditions will increase fire intensity but not necessarily duration. Conditions that influence the amount of time living tissue is exposed to a heat source are some of the most important variables that influence mortality.

In several burn barrel experiments, bluebunch wheatgrass, Idaho fescue, squirreltail, Thurber's needlegrass, and needle and thread were found to be more severely damaged during the growth period (May, June, and early July) than in late summer or fall (Wright and Klemmedson 1965; Britton and others 1990). However, Wright (1971) reported higher mortality of needle and thread grass in summer compared to spring burns. Squirreltail that was still green in July was moderately damaged by wildfire (Wright and Klemmedson 1965), while another study found a significant increase in growth and reproductive effort in the first year following a July wildfire because the plants were completely dormant at the time of the fire (Young and Miller 1985). In comparing a spring and fall burn in eastern Oregon, Pyle and Crawford (1996) found no apparent differences between seasons of burn. However, Sapsis (1990) reported higher mortality of Idaho fescue in the fall burn (20.1%) compared to the spring burn (3.5%). The inconsistent results are likely due to the conditions at the time of the fire and specific characteristics of the fire, which can be highly variable.

Summary

There is very good information on the variables that influence fire behavior, including severity and intensity. However, the complexity of these variables at the time of fire across heterogeneous landscapes makes predicting severity an imprecise science. The effects of season of burning on plants are unclear. Highly variable conditions that influence fire severity have probably contributed to contradictory findings in the literature.

Seed Banks, Establishment, and Reproductive Effort

Two key questions related to seed banks and fire are:

- How do seed banks influence early succession following fire?
- How does fire influence seed density and composition in the seed bank?

Seed banks within ecosystems can be subdivided into two groups based on location, which influences their vulnerability to fire: (1) seed that exists within the litter at or above the soil surface; and (2) seed buried below the soil surface. Seeds located on the soil surface or in the litter have a high probability of being consumed or exposed to lethal temperatures. However, in some cases, such as fast moving head fires, fire may not consume all the litter, which contains a portion of the surface seeds. This means that fire behavior is important for understanding seed survival above the soil surface. Seeds located below the soil surface are often protected by the insulation properties of the soil that prevents lethal temperatures from reaching buried seeds. Timing of fires relative to seed maturation and dispersal is a major factor related to sagebrush re-establishment because sagebrush blooms in the late summer and seeds mature in the late fall and early winter. In a typical fire season, a fire event occurs prior to sagebrush seed maturity so any sagebrush seed that survives a fire in the seed bank must be from a previous season's seed crop.

Seed banks of native species are potentially important in recovery of sagebrush communities (West 1983b) and thus are a potentially important component of **resilience**. However,

composition of residual seed in the soil seed bank after fire often does not reflect what is growing in the community, and density of native perennial seed is often low to near absent (Koniak and Everett 1982; Everett and Sharrow 1985b; Koniak 1985; Everett 1986; Hassan and West 1986; Allen and Nowak 2008; Allen and others 2008; Duncan 2008; Mazzola and others 2011). As a result, the persistence of resident perennial **herbs** following fire is likely a more important driver of early and mid-successional trajectories following disturbance than the seed bank (fig. 41) (Koniak and Everett 1982; Everett and Sharrow 1985a; Allen and others 2008; Pekas 2010). Initial response of cover and biomass for perennial grasses and forbs in the first and often second year following fire is primarily the result of growth from plants that survived the fire (Everett and Sharrow 1985b; West and Hassan 1985; Hassan and West 1986; Wehking 2002; Bates and others 2000, 2009; Davies and others 2009a; Miller and others in press). Rarely is an increase in density of these two functional groups reported in the first two years following fire. However, *reproductive effort* of surviving individuals frequently increases within this early period (Uresk and others 1976; Young and Miller 1985; Patton and others 1988; Sapsis 1990; Wrobleski and Kaufman 2003; Davies and Bates 2008; Bates and others 2009; Ellsworth and Kaufmann 2010), potentially providing an on-site seed source during the early post-fire years (two and after) following disturbance.

Few studies have examined the ecology of seeds from maturation through dispersal and eventual emergence. Pyke (1990) tracked monthly seed production, retention, seed bank numbers, and seed removal of desert wheatgrass and bluebunch wheatgrass that grew together in a Wyoming sagebrush grassland community in northwestern Utah near Snowville. Retention of seed in inflorescences can be an important factor in how quickly seeds become part of the seed bank. Desert wheatgrass can retain hundreds of seeds per plant in inflorescences for at least nine months after production, whereas bluebunch wheatgrass disperses seeds immediately upon maturation (Pyke 1990). The elevated seed in desert wheatgrass has a higher proportion of seeds at risk of burning before dispersal than bluebunch wheatgrass; however, desert wheatgrass compensates by producing more seeds (Pyke 1990). Seeds of both species decline to pre-dispersal levels by snowfall, resulting from consumption by granivores or mortality before they had a chance to germinate (Pyke 1990). Granivory contributed significantly to seed bank losses and has the potential to remain high throughout the year (Pyke 1990).

The proportion of post-fire annual species in the seed bank is usually greater than perennial species (Everett 1986; Hassan and West 1986; Akinsoji 1988; Wehking 2002; Allen and others 2008; Duncan 2008). On a Wyoming big sagebrush community in Utah, cheatgrass seed made up 51% of the seed bank but only accounted for 17% of the plant cover (Hassan and West 1986). With the exception of Sandberg bluegrass and squirreltail, perennial grass seed density is often low to absent (Hassan and West 1986; Allen and Nowak 2008; Allen and others 2008; Duncan 2008; Pekas 2010). Relatively abundant seeds of annual species found in the seed bank were cheatgrass, western tansymustard, maiden blue eyed Mary, slender phlox, flatspine stickseed, desert alyssum, and curveseed butterwort. The most common perennial was Sandberg bluegrass. The density of sagebrush seed in the seed bank prior to fire varied from relatively high (Wehking 2002; Allen and others 2008) to very low (Hassan and West 1986; Duncan 2008). Residual sagebrush seed in the seed bank could be a primary driver in determining the rate of sagebrush recovery, although establishment conditions in years immediately following fire are also extremely important (Ziegenhagen and Miller 2009; Wijayratne and Pyke 2012). Sagebrush seedling mortality can be very high in the first few weeks of emergence (Board and others 2011) due to lack of available moisture (Boltz 1994).

Figure 41. Recovery of perennial vegetation on this site during the first post-fire year was predominately from plants that survived the fire and not from the seed bank. The seed bank of perennial grasses and forbs is seldom a key variable in post-fire succession during the first and second years following fire. Some annual forbs such as slender phlox (**insert A**, an important sage-grouse food plant), ground smoke, and maiden blue eyed Mary often persist in the seed bank and can increase in abundance given low competition from perennial herbaceous species and annual grasses. However, most seeds are consumed in high severity fires or in hot-spots such as duff and litter layers beneath trees and shrubs (south-central Oregon; photos by Rick Miller).

Seed Distribution and Loss by Fire

The pattern of seed distribution in the seed bank located on the soil surface and in the litter influences the potential for seed to be consumed or exposed to lethal temperatures. The highest density and species richness of seed in the seed bank were beneath the shrub canopy followed by the interspace (Pekas 2010) and then beneath the tree canopy (Allen and others 2008). Koniak and Everett (1983) reported that the highest seed densities and greatest species richness were located at the edge of the interspace and litter mats located beneath trees or shrubs. Micro-sites of seed accumulation beneath woody plant canopies are considered "hot" spots where most or all of the seed is consumed by fire (Young and Evans 1975; Hassan and West 1986; Wehking 2002; Allen and others 2008) (see Fire Severity section). In two prescribed burns, fire directly consumed 85-98% of the viable seed bank (Beckstead and others 2011). On a cheatgrass dominated community on warm/dry soils, cheatgrass seed density on burned plots was <3% of that on adjacent unburned plots (Humphrey and Schupp 2001). However, seed bank density of the invader recovered within two years.

Summary

Seed composition in the soil seed bank is not a strong predictor of plant succession following fire, especially for perennial herbs. If adequate amounts of residual perennial herbs are present, they are usually the primary drivers of early succession due to increased size and reproductive effort. Seed density in the soil seed bank is reduced by fire, particularly

beneath shrub canopies. Although seed density of invasive species such as cheatgrass is greatly reduced by fire in the first year, enough seed can remain to produce a limited number of plants, which can produce large seed crops. Seed banks, however can be very important for species such as sagebrush that do not survive fire and have a limited dissemination distance, especially on cooler and moister ecological site types.

Post-Fire Grazing

Standard practice on Federal land is to defer grazing for a minimum of two growing seasons following fire. There are very limited data evaluating the optimal time of the rest period. The length of time necessary for a plant community or ecological site to adequately recover before implementing grazing depends on a number of interacting variables. These are characteristics of the **ecological site,** including **resilience to disturbance** and **resistance to invasives**, **fire severity**, post-disturbance climate, plant composition of the community prior to disturbance, post-fire grazing management, and additional post-fire disturbances. In southern Idaho, early summer clipping during the growing season in the first and second years following burning of Idaho fescue, bluebunch wheatgrass, and squirreltail significantly increased mortality and decreased production, basal area, and culm numbers (Jirik and Bunting 1994; Bunting and others 1998). Defoliation during the growth period to a 1 inch (2.5 mm) stubble height in the first and second years resulted in cumulative mortality of 30 and 50% for Idaho fescue and 50% and 70% for bluebunch wheatgrass (Bunting and others 1998). However, defoliation of dormant plants in the fall in the first and second years following fire had little effect on plants.

Bates and others (2009) examined herbaceous species responses after a prescribed head fire conducted in late September and early October 2002 on a Wyoming big sagebrush/Thurber's needlegrass community with cool/moist soils that was in a non-degraded **seral** stage and had little cheatgrass. They reported no difference in recovery of herbaceous biomass between grazed-burn and ungrazed-burn plots when the community was moderately grazed in August (when plants were dormant) in the first, second, or third years post-fire. Spring grazing (just prior to or during reproductive shoot elongation for deep-rooted perennial bunchgrasses) in the second and third spring, or only in the third spring, after fire reduced the current year's growth by one-third relative to that of burned-ungrazed areas. Bruce and others (2007) reported similar results in Nevada. They found no difference in recovery of herbaceous cover between moderately grazed and ungrazed burned pastures when grazed from seed-set through dormancy (1 July to 31 August) in the first and second years following a wildfire. The pasture ranged in elevation from 5575 to 9850 ft (1700 to 3000 m) and contained basin big sagebrush, black sagebrush, Indian ricegrass, and Thurber's needlegrass at the middle elevations, and single-needle pinyon, Utah juniper, mountain big sagebrush, serviceberry, Idaho fescue, and bluebunch wheatgrass at the high elevations. Cheatgrass cover remained below 3% across grazed and ungrazed treatments. A third study evaluated delay and season of sheep grazing on vegetation recovery after a wildfire in a cool/moist three-tip sagebrush-mountain big sagebrush community in Idaho (Roselle and others 2010). Moderate grazing was initiated one, two, and three years post-fire in fall (September through October) and two and three years after fire in spring (May through June). The authors found no significant difference in perennial grass cover three years after fire regardless of grazing treatment. However, the increase of bluebunch wheatgrass cover (the dominant perennial grass) in the grazed treatment was less than the increase that occurred on the non-grazed pasture. Cheatgrass cover was not affected by grazing treatment. In Utah, under heavy grazing, lower perennial grass densities and higher weed cover occurred in burned-grazed pastures compared to burned-ungrazed pastures

(Pickford 1932). West and Yorks (2002) reported lower total vegetation and perennial grass cover and higher cheatgrass cover in grazed-burned than in ungrazed-burn areas in later successional **states** (12 to 18 years after wildfire) in Utah.

Summary

Only the Jirik and Bunting (1994) and Bunting and others (1998) studies evaluated the impacts of defoliation on perennial grasses during the active spring growth period in the first growing season following fire. They reported a significant increase in mortality and decline in production and basal area of perennial grasses. All other studies examined delayed grazing during senescence or dormancy in the first year following fire. Grazing during the dormant season following fire appears to be tolerated by most perennial grasses. Although post-fire grazing studies are limited and post-fire grazing (timing, intensity, frequency) is highly variable, the literature justifies deferring grazing during the first growing season following fire. Grazing effects during the active growth period in the second year are less clear and probably depend on a number of variables, including characteristics of the ecological site (e.g., productivity, soil moisture and temperature regimes, resilience, and resistance to invasives), fire severity, post-disturbance weather, plant composition of the community prior to disturbance, post-fire grazing management, and additional post-fire disturbances. Based on the principles of resilience, an ecological site that is in a non-degraded seral stage and occupies soils that are cool/moist will be more resilient to moderate grazing following fire compared to an ecological site in poor condition and/or located on warm/dry or moist moisture regimes. It should be noted that studies on the impacts of post-fire grazing have only looked at plant response and have been of short duration. Limitations are lack of information on effects of post-fire grazing on litter build up, soil stability, and micro-site modification, which may be especially important in warm/dry communities. Also, longer-term studies are needed to evaluate grazing effects on longer-term successional trajectories.

Fire Versus Fire Surrogates

Fire surrogate treatments are those vegetation manipulations designed to mimic changes in plant species composition or fuel load and structure in a similar way to fire. Many plant communities in the Great Basin are considered to be at risk of being replaced by cheatgrass or other highly invasive species that are tolerant to fire. These communities (**at-risk phase**) are under the greatest threat of shifting to alternative **states** (fig. 11). There is strong support in the literature to suggest that ecological sites with warm thermal regimes (e.g., mesic soils), low and highly variable precipitation, and reduced and spatially patchy perennial grass components have low resistance to invasion by exotic species. Both wild and prescribed fires most frequently result in large increases in cheatgrass and other non-natives on these ecological sites. The general perception, based on observation and limited research, is that invasive species will increase less using fire surrogate treatments than prescribed fire. However, there is surprisingly little research that directly compares vegetation response between fire and fire surrogates. A second consideration in using fire surrogates is retention of the shrub layer (fig. 42A), which can take considerable time to reestablish following fire. The shrub layer, when balanced with the herbaceous layer, provides important ecological functions including micro-site modification (e.g., soil surface temperature and moisture), snow retention, reduced wind erosion, nutrient cycling, and wildlife habitat, particularly for sagebrush obligate species. Fire surrogates are also appropriate when surface fuels (<3.3 ft [1.0 m]) are inadequate to carry **surface fire** under moderate weather conditions (fig. 42B).

Key questions to consider when selecting a fire or non-fire treatment are:

• Based on ecological site characteristics and plant composition and structure, are invasive species of concern?
• How will shrub recovery differ between fire and fire surrogate treatments?
• How will the rate of tree reinvasion differ between fire and fire surrogate treatments?

We know that soil resource availability temporarily increases following fire (Hobbs and Schimel 1984; Blank and others 1994b; Young and Allen 1997; Stubbs and Pyke 2005; Davies and others 2007) and can lower a community's resistance to invasive grasses (Blank and others 1994b; Beckstead and Augspurger 2004; Chambers and others 2007; Johnson and others 2011). Few studies, however, have directly compared changes in re-source availability between fire surrogates and fire. The assumption is that fire increases resource availability, especially soil nutrients in the short term, to a greater magnitude than fire surrogates.

There is good support in the literature to suggest that invasive species increase, at least temporarily, following fire surrogate treatments designed to reduce pinyon and or juniper.

Figure 42. **(A)** Using a fire surrogate allowed retention of the shrub layer on this cool/moist site in central Nevada. **(B)** Shrub fuels, which connect surface and tree canopy fuels, were absent on this cool/moist site in Phase III woodland. This community has relatively high resilience to disturbance and resistance to invasives due to ecological site characteristics and an understory with good cover of native perennial grasses in which Idaho fescue dominates (southeastern OR). Re-establishment of trees is usually more rapid where trees are removed by mechanical methods rather than by fire and where trees are of seed bearing age. The rate of re-establishment is largely influenced by the density of small trees remaining in the understory and of viable juniper and pinyon pine seeds in the seed bank in the duff layers. Many seeds and most seedlings of juniper and pinyon are consumed by fire but little mortality results from mechanical treatments. (central Nevada; photos by Rick Miller).

On cool soils, mechanical removal of pinyon and juniper resulted in a limited increase in cheatgrass (Bates and others 2000, 2007a; Baughman and others 2010). However, cheatgrass cover increased to 20% cover during wet years in areas directly beneath fallen trees but declined to <5% in dry years (Bates and others 2007a). Cheatgrass remained <5% in the interspace, even in wet years. In Nevada, the response of invasive annuals following mechanically thinning a stand of trees varied with the density of trees prior to treatment, approaching 8% cover in closed stands and <1% in open stands (Baughman and others 2010). Although results at times appear to be mixed, they can be at least partially explained by evaluating ecological site characteristics. On a warm/dry (8 to 10 inches [200 to 250 mm] **PZ**) ecological site with a depleted herbaceous understory, the removal of juniper using fire, mechanical, or chemical treatments all resulted in large increases in cheatgrass and medusahead (Evans and Young 1985, 1987). On cool soils, cheatgrass cover, although initially low in both treatments (<1%), was greater on the burn compared to chained treatment 30 years after treatment (Bristow 2010). Miller and others (in preparation b) reported that cheatgrass and exotic forbs increased in mechanical (warm soils only) and burned treatments (warm and cool soils), but the magnitude of increase was significantly greater for burned ecological sites occupying warm soils compared to cool soils (fig. 30).

Mechanical treatments usually have significantly less impact on the shrub layer compared to fire (fig. 42A) (Mueggler and Blaisdell 1958; Miller and others in press). In southeastern Idaho, sagebrush production was reduced 89% in the burn, 86% on rotobeaten, 57% on railed, and 50% on sprayed plots three years after treatment (Mueggler and Blaisdell 1958). Miller and others (in press) reported a significant increase in shrub cover immediately following cutting compared to adjacent control plots in contrast to a 10-fold decrease in shrub cover in burn plots (fig. 22). Retention of the shrub layer may be of greater importance on ecological sites with warm/dry soil regimes where re-colonization of shrubs is slower than on ecological sites with cool/moist regimes.

Summary

Although direct comparisons are limited between fire and fire surrogate treatments, the literature suggests that there will be a greater magnitude of increase in invasive annuals when using fire, at least in the first few years. The few studies that did compare fire and fire surrogate treatments strongly support this. However, ecological site characteristics that play a large role in determining the magnitude of increase and persistence of invasive species are: (1) plant composition and structure at the time of treatment, especially of the perennial herbaceous species; (2) ecological site thermal regimes (e.g., warm versus cool or cold); and (3) moisture availability. It is also likely that cheatgrass and other invasives will significantly increase on warm/dry ecological sites with a depleted herbaceous understory, regardless of the treatment selected. Primary considerations in deciding to use prescribed fire or a fire surrogate are: (1) characteristics of the ecological site, (2) risk of invasives, and (3) the importance of retaining the shrub layer if it is present, especially on warm/dry ecological sites where sagebrush establishment can be very limited. The use of fire is a high risk treatment on warm soils, particularly in dry moisture regimes. Sites in a degraded low-seral stage (e.g., low productivity, herbaceous native perennials depleted, increased bare ground, and co-dominance or dominance of invasive species) are in obvious need of seeding regardless of the treatment selected, especially on warmer, drier ecological sites. When selecting a treatment, addressing key questions related to the ecological site's characteristics will determine its ability to recover (resilience) and resist invasive species.

Ecological Site Characteristics, Resilience, and Resistance Summary

Ecological site characteristics and disturbance severity play a complex role in influencing **resilience to disturbance** and **resistance to invasives**. Evaluating ecological site resilience and resistance involves addressing questions that are similar to those asked when predicting plant successional trajectories (fig. 43). Possibly the most important characteristics, and ones that are relatively easy to identify and describe, are the thermal and moisture regimes of an ecological site (identified by elevation, aspect, and indicator species) and plant composition at the time of the disturbance (fig. 44). In the Great Basin Region, temperature and moisture regimes are largely a function of elevation, aspect, and soils (particularly characteristics related to capture and storage). These characteristics are described in ecological site and plant community descriptions (Briske and others 2008; USDA-NRCS 2013). For example, soil temperature/moisture regimes that identify warm, cool, or cold and dry or moist soils (**mesic**, **frigid**, and **cryic**; **aridic** and **xeric**) are included in ecological site and soil descriptions (usually in the soil family name). These regimes are typically identified and mapped using elevation and aspect. When mapped, the regimes appear to be separated by distinct boundaries. In reality, these regimes are continuums as described in the x-axis in figure 5, so it is important to determine if the ecological site is on the warm/dry (mesic/aridic) or cool/moist (frigid/xeric) end of a specific soil temperature/moisture regime. Although the variable landscape in the Great Basin Region creates a high degree of heterogeneity, predictable changes in temperature, precipitation, potential vegetation, and productivity occur along environmental gradients created by topographic features (West 1983a; Dahlgren and others 1997). As a result, both plant succession and resistance to invasives following disturbance change along these environmental gradients (Dhaemers 2006; Johnson and Miller 2006; Chambers and others 2007; Davies and others 2011). There are numerous examples in the various subsections supporting the concept that warmer ecological sites (mesic soils) are less resilient to fire and resistant to invasives than ecological sites with cooler soil temperature regimes (frigid or cryic soils). Evaluating these characteristics helps sort out the variable responses reported in the literature. Greater seedling establishment of native perennials and, typically, limited response of invasive species occur on cooler aspects (north to east) than on warmer aspects (south to west) (Koniak 1983; Everett and Sharrow 1985b; Sheley and Bates 2008; Bates and others 2011). There
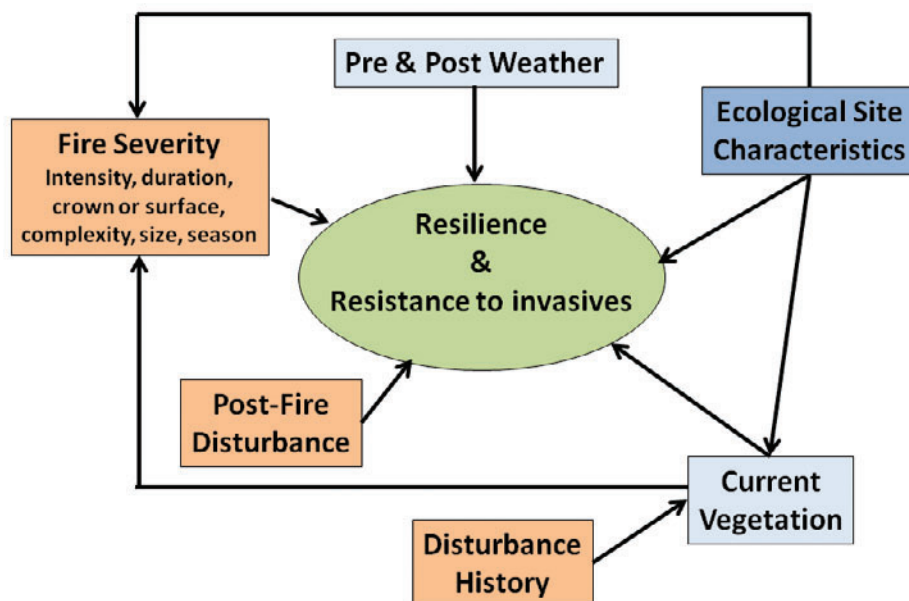
Figure 43. A simple conceptual model identifying the key components that influence resilience to disturbance and resistance to invasive species. These components and their attributes are important to consider when evaluating a site's resilience and resistance (similar to the model for resilience to disturbance and resistance to invasives; see fig. 15).
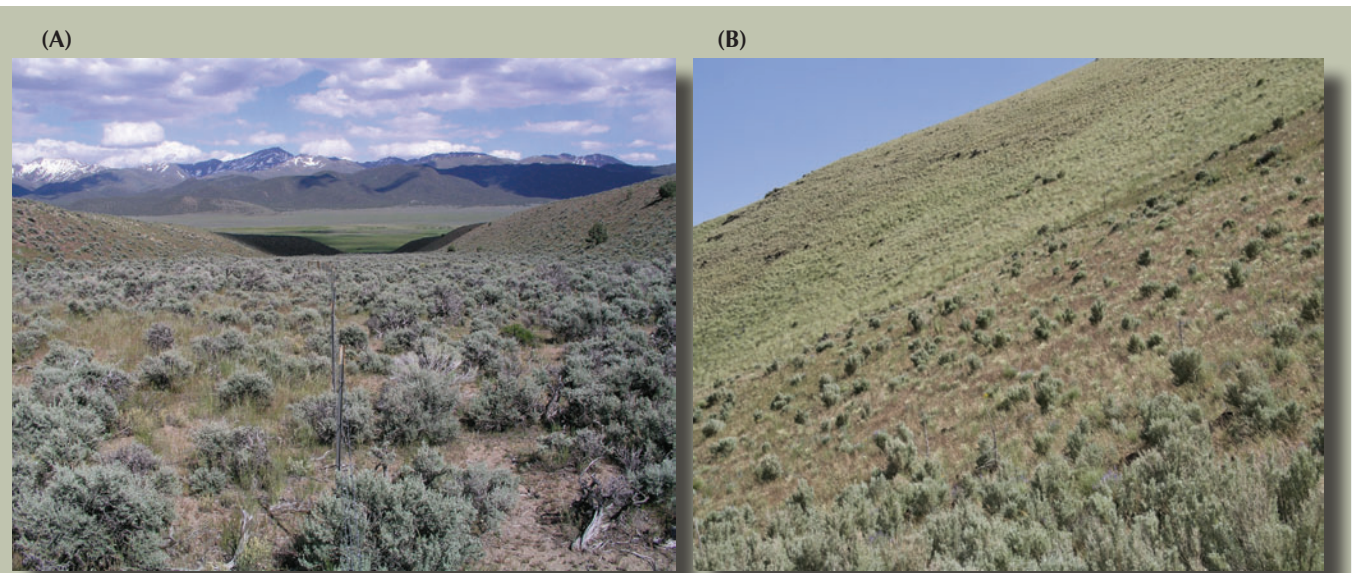
Figure 44. **(A)** A cool/dry Wyoming big sagebrush/needle and thread site in central Nevada (photo by Jeanne Chambers), and **(B)** warm/dry Wyoming big sagebrush/bluebunch wheatgrass site in southeast Oregon (photo by Rick Miller), both with degraded understories on the right side of the fence. On-site resilience to fire and resistance to invasives differs both between locations and sides of the fence. Location A has a cooler soil regime increasing its resilience and resistance compared to location B. And, the depleted perennial herbs on the left side of the fence, significantly lowers the resilience and resistances to invasives at both locations. Ecological site characteristics and plant composition are key components in determining post-fire response. In **(B)**, the site in the foreground may have the ability to recover (this site is on the cool and moist end of the warm/dry gradient and squirreltail is a co-dominant with cheatgrass). However, fire may also act as a trigger moving the site into an alternative state (annual grassland).

also is good support in the literature (discussed previously) that resilience to disturbance and resistance to invasives changes across soil temperature/moisture regimes (Miller and others in preparation b) and differs among Wyoming big sagebrush and mountain big sagebrush **cover types** (Cook and others 1994). The abundance of native herbaceous perennials present in a community prior to disturbance is also extremely important since they provide the primary plant material for post-disturbance recovery and resistance to invasive species (Chambers and others 2007; Baughman and others 2010).

Section 4: Fire Effects on Soils in the Great Basin Region

Introduction

Soils support nutrient, water, and energy cycles and act as a reservoir that captures and stores water and nutrients essential for plant growth. Fire can influence soils directly by altering **hydrophobicity**, texture, porosity and permeability, mineralization rates, and nutrient and organic matter content (Wohlgemuth and others 2006). Fire indirectly influences soils by changing plant composition and structure and litter cover, which influences microtopography, organic matter inputs, and the amount of bare ground and, thus, post-fire soil surface temperatures and evaporation. The degree to which fire affects soil's physical and chemical properties depends on **fire severity** and behavior (fig. 16), season of burn, ecological site **resilience**, and vegetation composition and structure prior to and following the burn. When determining or predicting fire effects on soils, key attributes to evaluate and questions to address are similar to those associated with plant succession in fig. 15.

The effect of fire (severity) on soils is related to intensity (energy released) and duration, which interact to determine consumption of organic material and temperatures at the soil surface and subsurface. Soil surface temperatures (a measure of intensity) are most frequently reported in the literature and often related to combustion of organic surface materials and tree mortality. However, belowground damage (a measure of severity) is largely influenced by duration (DeBano and others 1998; Neary and others 1999). Temperatures around 122 to 150 °F (48 to 65 °C) for 1 minute are lethal to living tissue (Pyne and others 1996; Neary and others 1999). However, changes in physical and chemical soil properties occur at much higher temperatures (Neary and others 1999).

Fire Soil Temperatures

Soil Surface Temperatures

Soil surface temperatures are spatially variable and differ according to vegetation type and fuel loads. Neary and others (1999) reported soil surface temperatures during fires of <440 °F (<225 °C) for grasslands, 480 to 1300 °F (250 to 700 °C) in shrublands, and 930 to 1300 °F (500 to 700 °C) beneath slash piles. However, soil surface temperatures in sagebrush steppe are spatially heterogeneous due to large differences in fuel abundance and structure for interspaces and beneath grass, shrub, or tree canopies. Soil surface temperatures typically ranged from <175 to 400 °F (<79 to 204 °C) in tree or shrub interspaces and from 1166 to 1300 °F (603 to 704 °C) in tree canopy litter (Bates and others 2011). Korfmacher and others (2003) reported significantly higher surface temperature beneath shrubs (718 °F [381 °C]) compared to bare ground and under grasses (590 to 585 °F [310 to 307 °C]), respectively. In Utah, soil surface temperatures reached 370 °F (187 °C) in open grassy areas and 1430 °F (777 °C) beneath pinyon and juniper debris piles (Gifford 1981). Beckstead and others (2011) reported cooler soil surface temperatures in their study, but temperatures were averaged across both interspace and canopy. Temperatures ranged

between 250 and 300 °F (120 and 145 °C) for two burns in sagebrush steppe in Washington and Utah. Soil surface temperatures between 248 and 293 °F (120 and 145 °C) consumed 85 to 98% of the cheatgrass seed in the seed bank (Beckstead and others 2011), and temperatures between 1400 and 1800°F (760 and 980 °C) beneath sagebrush canopies created water-repellent soils (Salih and others 1973). However, it is both temperature and duration that determine the effects of fire on soil processes and characteristics.

Soil Subsurface Temperatures

Soil insulative properties are important in reducing subsurface soil temperatures; however, soil water also contributes to the depths at which soil heat is transmitted. If surface temperatures exceed the boiling point of water, 212 °F (100 °C), all moisture must be evaporated before temperatures will increase in that layer. Fine textured soils with higher bulk densities will conduct heat to greater depths than coarser textured soils when dry, but provided surface temperatures do not exceed the boiling point, coarser soils will conduct heat better because water is a better conductor of heat (Trollope 1984 cited in Whelan 1995). Temperatures of 212 °F (100 °C) in the humus layer are lethal to most soil organisms (Pietikainen and others 2000). In chaparral vegetation, subsurface soil temperatures at 2 inches (50 mm) usually do not go above 212 °F (100 °C) regardless of duration and or **fire severity**, although they can approach 212 °F (100 °C) at >20 minutes duration (DeBano and others 1998). At 1 inch (25 mm) below the surface, temperatures remained <212 °F (<100 °C) with durations of <5 minutes, but approached 390 to 570 °F (200 to 300 °C) with durations >20 minutes, for moderate to high severity fires. In grasslands, subsurface soil temperatures rarely exceed 122 to 176 °F (50 to 80 °C). In cut pinyon and juniper, soil subsurface temperatures at 1 inch (25 mm) were <130 °F (<55 °C) in the interspace and 550 °F (288 °C) beneath tree debris piles (Gifford 1981). In sagebrush steppe in Washington and Utah, surface soil temperatures at 0.4 and 1.2 inch (10 and 30 mm) soil depths reached 140 to 167 °F (60 to 75 °C) and 122 to 140 °F (50 to 60 °C), respectively, compared to surface temperatures of 248 to 293 °F (120 to 145 °C) (Beckstead and others 2011).

Soil Nutrients

Nitrogen

The total amount of nitrogen (N) lost as a result of fire is influenced by fire severity, which is related to the amount of vegetation consumed, especially foliage (table 2, fig. 18), and the amount of N volatilized. N volatilizes above 390°F (200 °C) and over half of the aboveground N can be volatilized when soil temperatures are 930°F (>500 °C) (DeBano and others 1998). In pinyon and juniper woodlands, prescribed burning resulted in a 78% percent reduction in aboveground N. However, only a small portion of the total N pool is located above ground with 85 to 95% of ecosystem N stored in the soil (Rau and others 2010).

The amount of available N for plant growth after fire differs as a function of volatilization, soil temperature, erosion processes, and time since fire (DeBano and others 1998). Blank and others (1996) reported a slight increase in soil nitrate (NO_3_) at 300 and 480°F (150 and 250 °C) but significant reductions at 660 to 840 °F (350 to 450 °C). However, heat-induced formation of ammonium (NH_{4+}) did not occur until temperatures reached 660 °F (350 °C) for 5 minutes. At higher temperatures and duration (840 °F [450 °C] for 15 minutes), lower levels of NH_{4+} were released. Although total aboveground N is usually reduced after fire, available N for plant growth is increased immediately (Blank and others 2007; Rau and

others 2008, 2009). Increases in available nutrients occur due to deposition of ash onto the soil surface, release of phosphorus (ortho-P) and NH_{4+} from organic matter, decomposition of belowground biomass, and further oxidation of NH_{4+} to nitrogen dioxide (NO_2) and then NO_{3-} by bacteria (Hobbs and Schimel 1984; DeBano and Klopatek 1988; Covington and others 1991; Blank and Zamudio 1998). Levels of NH_{4+} are usually elevated immediately post-burn, but levels of NO_{3-} increase over time with mineralization of NH_{4+}. Elevated levels of available N (NH_{4+} plus NO_{3-}) have been reported to last for about one year (Stubbs and Pyke 2005; Rau and others 2009), two years (Blank and others 1994c, 2007), and four years (Rau and others 2007, 2008). Only Rau and others (2007) monitored levels of available N on burned and unburned plots for more than two years after fire, and there are no data reporting longer-term effects. Higher concentrations of N beneath shrub and tree canopies compared to the interspace and lower temperatures in the interspace may result in high spatial variability of available N (e.g., Rau and others 2009). No change in available soil N was found in the shrub interspace on a Wyoming big sagebrush community, but an increase did occur beneath the shrub canopies at 6 inches (150 mm) in years one and two (Davies and others 2009a). Halvorson and others (1997) also reported increased levels of N beneath burned sagebrush canopies but not in the interspace. This may be partially explained by higher initial concentrations of available N beneath shrub and tree canopies compared to interspaces and surface soil temperatures in interspaces that are too low to affect available nitrogen (Rau and others 2009). In Nevada, significantly more NH_{4+} and net N mineralized, but less NO_{3-}, were measured in heated soils compared to unheated soils (Blank and others 1994a). In fire surrogate treatments, Bates and others (2002) measured an increase in soil N in the first year following cutting of western juniper but no difference in the second year. In a second study, Bates and others (2007b) reported available soil N increased in the second year following cutting and leaving juniper in place. Concerns related to elevated available N following fire are higher rates of cheatgrass growth and reproduction (Monaco and others 2003; Beckstead and Ausgspurger 2004; Blank and others 2007; Chambers and others 2007).

Summary

Understanding the potential effects of fire on soil N requires knowledge of the soil type, species composition and abundance, fuel loads, and fire severity/soil temperatures during the burn. Amount of aboveground total N lost is directly associated with the amount of organic matter consumed and N volatilized (fig. 38). Literature strongly supports the immediate loss of total N and the increase in available soil N following fire. However, the length of time that elevated levels of available soil N persist needs additional documentation. We have little knowledge related to the long-term impacts of fire, especially repeated fires, on ecosystem N.

Carbon

Carbon (C) volatilization begins at 356°F (180 °C) with total organic matter consumption usually occurring at 842 °F (450 °C) (DeBano and others 1998). In Nevada, a community with 30% Utah juniper and singleleaf pinyon cover averaged 44,000 lb/acre (40,000 kg/ha) in aboveground biomass and 88,000 lb/acre (80,000 kg/ha) in total ecosystem C (above and belowground) (Rau and others 2010). Total aboveground C accounted for 25% (22,000 lb/acre [20,000 kg/ha]) of total ecosystem C. Prescribed burning resulted in 68% or 18,260 lb/acre (13,600 kg/ha) reduction in aboveground C. The majority of oxidized C was from the combustion of foliage and 1-hr fuels (organics <1/4 inch [6.4 mm] in diameter), of which 90% were consumed. The majority of 10-hr (1/4 to 1 inche [6.4 to 25 mm]), 100-hr (1 to 3 inches [25 to 76 mm]), and 1000-hr (>3 inches [>76 mm]) woody fuels remained

on site after the fire. Klopatek and others (1991) reported similar results (13,860 lb/acre [12,600 kg/ha]) in aboveground C loss for pinyon-juniper woodlands in the Southwest.

In a mountain big sagebrush community encroached by Utah juniper and singleleaf pinyon, soil C increased immediately following fire at 0 to 1.2 inches (0 to 30 mm) soil depth beneath shrub and tree canopies, but differences were not detectable six years after fire (Rau and others 2009). No differences were detected between burned and unburned tree and shrub interspace or below 3.1 inches (80 mm) soil depth beneath the shrub or tree canopies during the six year post-fire period. Soil surface temperatures in the upper 0.8 inches (20 mm) beneath woody canopies ranged between 390 and 572 °F (200 and 300 °C), which is adequate to volatilize C. However, temperatures in the interspace and below 0.8 inches (20 mm) soil depth in all three micro-sites were 176 °F (<80 °C), which is not hot enough to volatize C. Halvorson and others (1997) also reported no differences in total C in burned and unburned interspaces in big sagebrush/bluebunch wheatgrass communities in central Washington. They reported soil C was still higher within 20 inches (510 mm) of burned sagebrush stumps compared to shrub interspaces nine years after burning. However, levels did not differ beneath burned and unburned sagebrush canopies. Rau and others 2010 also noted that fire had little to no impact on soil C in a pinyon and juniper woodland in central Nevada. Woodland expansion has some potential for increasing ecosystem organic C. However, Rau and others (2011b) stated that the limited increase should be weighed against the risk of loss through high intensity fires and possible transition to exotic grasslands. The transition from shrubland or woodland to cheatgrass-dominated communities could result in decline of soil organic C below 24 inches (610 mm) (Rau and others 2011a).

Fire surrogate treatments (cutting and leaving) increased soil C in the upper 0 to 4 inches (0 to 100 mm) during the first two years following treatment (Bates and others 2002). There were no differences detected in the interspace or tree **duff** layer between cut and uncut. Rau (personal communication) speculated that although there is an immediate loss of above-ground C through volatilization from fire, gradual loss of C following cutting will occur through oxidation unless organic material is incorporated into the soil. In addition, C remaining on site in the form of ash or charcoal is a very resistant (long lasting) form of C.

Summary

It appears that only limited amounts of total ecosystem C is lost during fire because most ecosystem C is stored below ground. However, we have little information on the long-term response of ecosystems following multiple fires or fire surrogate treatments and transition to new steady states (e.g., annual grasslands).

Water-Soluble Anions and Cations: Ca^{++}, Mg^{++}, Na^+, K^+, $SO4^{--}$, ortho-P

Total cation exchange capacity (the sum of the exchangeable cations found largely on humus and clay particles) usually decreases following fire and often remains low for one year (Wohlgemuth and others 2006). However, available water-soluble nutrients usually increase. In sagebrush communities, burning resulted in the increase of available sulfate ($SO4^{--}$), calcium (Ca^{++}), sodium (Na^+), magnesium (Mg^{++}), and zinc (Zn^{++}) in the upper 4 inches (100 mm) of soil for at least one year following fire (Blank and others 1994b, 1994c, 1996, 2007; Wohlgemuth and others 2006; Rau and others 2008). Levels of Ca^{++}, Mg^{++}, and Na^+ can influence availability of some nutrients important for plant growth (e.g., potassium [K^+]) (Blank and others 2007).

In a laboratory experiment, the magnitude of increase of most water-soluble anions and cations was influenced by fire intensity (temperature) ranging from 300 to 840 °F (150 to

450 °C) and durations ranging from 1 to 15 minutes (Blank and others 1996). The highest concentrations of water soluble anions generally occurred at temperatures of 660 °F (350 °C) for 15 minutes but were significantly reduced at temperatures of 840 °F (450 °C) for 15 minutes, suggesting soil nutrient concentrations are more likely to increase beneath shrub canopies than in interspaces due to higher temperatures. Phosphorus (P) content in soils and leaf contents of big sagebrush and squirreltail were greater on burned soils than unburned soils (Blank and others 1994c, 2007). However, in another study in central Nevada on a different soil type, P did not differ between burned and unburned plots during post-fire years one through four (Rau and others 2007). In pinyon-juniper woodlands, 98% of the P pool was located belowground (Covington and DeBano 1988).

Summary

Burning usually increases the availability of Ca^{++}, Mg^{++}, Na^+, K^+, and $SO4^-$ although fire intensity and duration and changes in soil physical and chemical characteristics influence amounts. Work conducted in the Great Basin Region is generally supported by work in other ecosystems (e.g., dry forests) in the West. However, the response of soil nutrients to a single or multiple fires over the long term in different soil types is limited to unknown.

Post-Fire Soil Water

Burning or use of fire surrogate treatments produces variable results in near-surface soil water, but often increases deeper soil water in the first few years following treatment. Soil water content following removal of western juniper in closed woodlands was higher at both 0 to 8 and 8 to 16 inches (0 to 200 and 200 to 400 mm) in post-cut years one and two (Bates and others 1998, 2000; Roundy and others in press, in preparation). In the Bates and others study, pre-dawn and mid-day leaf water potentials of sapling trees also remained higher throughout the summer in tree removal plots, indicating greater available soil water and a longer growing season in cut plots. In Nevada, soil water at 6 and 12 inches (150 and 300 mm) was also significantly greater in tree harvested plots for the first two years post-treatment, but there were no differences by year five (Everett and Sharrow 1985c). In the above studies, soil water availability declined in the upper soil layer and both groups of researchers speculated that the decline was due to increased transpiration from released understory vegetation. In sagebrush shrubland in central Nevada and western Utah, burn plots had six more days of available water for plant growth than unburned plots at 6 and 12 inches (150 and 300 mm) in the first growing season but there was no difference in the second post-fire year (Chambers and others 2007; Whittaker and others 2008). They measured no difference in soil water above 1.2 inches (30 mm) between burned and unburned plots. Across multiple study locations in the Great Basin, soil water was still available for an additional two weeks in the summer four years after burning or mechanical treatments in phase III juniper and pinyon woodlands compared to untreated sites (Roundy and others in press). Following prescribed fire in basin big sagebrush in central Nevada, soil water was lower at depths less than 12 inches (300 mm) on burned than unburned plots but higher at depths of 12 to 24 inches (300 to 610 mm) (Chambers and Linnerooth 2001). Lower soil water in surface soils was attributed to snow removal, desiccation by wind, and increased soil temperatures resulting in higher evaporation, while higher soil moister with depth was attributed to removal of deep rooted sagebrush. In **aridic** shrub-steppe in Washington, soil water in the upper 4 ft (1.25 m) did not differ between burned and unburned plots but was significantly greater on burned plots when measured to an overall depth of 9 ft (2.75 m) during the first four years post-fire (Link and others 1990). These results were attributed to a larger leaf area index (LAI) for herbaceous species but low LAI for shrubs, which are deeper rooted, on the burned compared to unburned plots. Also, Davies and others

(2009) did not find a difference in a burned and unburned Wyoming big sagebrush/Thurber needlegrass community at depths of 0 to 6 and 6 to 12 inches (0 to 150 and 150 to 300 mm).

Summary

Removal of woody vegetation can result in a short-term increase in soil water in the upper soil profile. However, following fires or fire surrogate treatments that remove surface vegetation or leave exposed soils, soil water availability can decrease in near surface soils and may result in reduced seedling establishment. Soil water is more likely to increase at depths >6 to 12 inches (150 to 300 mm) and persist longer at depths >12 inches (>300 mm). Longer-term decreases in near surface soil water (e.g., <12 inches [<300 mm]) and increases in deeper soil water (>12 inches [>300 mm]) can be attributed to an increase in herbaceous leaf area and the reduction of deeper rooted woody species.

Mycorrhizal Fungi and Microbial Communities

Vesicular-arbuscular mycorrhizal (VAM) fungi are easily killed by fire (DeBano and others 1998) and the magnitude of loss is related to increasing **fire severity** (Klopatek and others 1988, 1994). Soil insulative qualities (including soil water) in addition to fire intensity and duration are key variables influencing the effect of fire on VAM populations. Klopatek and others (1988, 1994) reported a greater decline in VAM fungi on dry than wet soils immediately following fire in pinyon-juniper woodland. A larger reduction (80%) occurred on dry soils beneath Utah juniper canopies on soils beneath singleleaf pinyon and interspaces (48 and 40%, respectively). Fire had little effect on VAM fungi populations on wet soils in tree interspaces. Soil temperatures of 200 °F (94 °C) completely killed VAM fungi on dry soils. In a sagebrush-steppe community, soil microbial populations were lower in burned than unburned plots during the first three years following fire, but no differences were detected in burned sagebrush communities 7 and 39 years after fire (Dangi and others 2010).

Summary

There is very limited literature evaluating the effects of fire on VAM fungi. Only one study measured the long-term effects of fire and none measured the effects of multiple fires. The few existing studies strongly suggest that fire results in a significant decline in VAM fungi, although the degree of impact is closely related to soil characteristics and **fire severity**.

Hydrology and Erosion

Hydrologic Processes in Relation to Resistance and Resilience of Vegetation Communities

Rangeland hydrologic and/or erosion responses to storm events are a function of the resisting forces (surface protection, soil properties, surface roughness, and topography) (fig. 45) and the driving forces (rainfall or wind) acting on the landscape (Davenport and others 1998; Sankey and others 2009, 2012a; Pierson and others 2011, Pierson and Williams in preparation; Williams and others in press). Sparsely vegetated or bare soil locations (source areas) generate runoff and erosion by overland flow and wind, whereas areas with ample vegetation and surface protection (sink areas) safely capture and store water and soil resources (Blackburn 1975; Pierson and others 1994, 2009, 2010). Ecological site characteristics (e.g., hillslope position and angle, soil characteristics, and precipitation and **soil moisture regimes**; fig. 12) and land use and disturbance dictate the spatial arrangement of source and sink areas for water on a particular ecological site. Degraded

Figure 45. The effects of fire on capture and storage of water and on erosion are largely a result of changes in surface roughness due to the combustion of vegetation and potential for increasing soil water repellency. The recovery of surface vegetation is important in decreasing soil erosion and capturing and retaining moisture (Five Creeks, Steens Mountain, Oregon; photo by Rick Miller).

Figure 46. (A) Common physiognomy shifts in sagebrush steppe with advancing woodland encroachment, **(B)** associated degradation-induced shift in dominant erosion processes, and **(C)** representative increase in erosion magnitude associated with changes in site/ground surface conditions. Erosion from stable sagebrush steppe communities occurs primarily by rainsplash and sheetflow and is typically low. Erosion increases exponentially with site and ground surface degradation where bare soil increases beyond 50-60%. High rates of erosion typically occur where sagebrush communities transition to Phase II-III woodlands. The exponential increase in soil loss (C, red line) with site/ground surface degradation illustrates the effect of concentrated flow. Concentrated flow is the dominant erosion process at the transition from Phase II-III woodland encroachment and signals a transition from a stable to degrading landscape. Concentrated flow has higher velocity than sheetflow and thereby exhibits greater sediment detachment and transport capacity than the combined effects of rainsplash and sheetflow. Figure modified from Williams and others (2013). Rainsplash photograph (B) courtesy of United States Department of Agriculture, Natural Resources Conservation Service. All other photographs were taken by the authors.
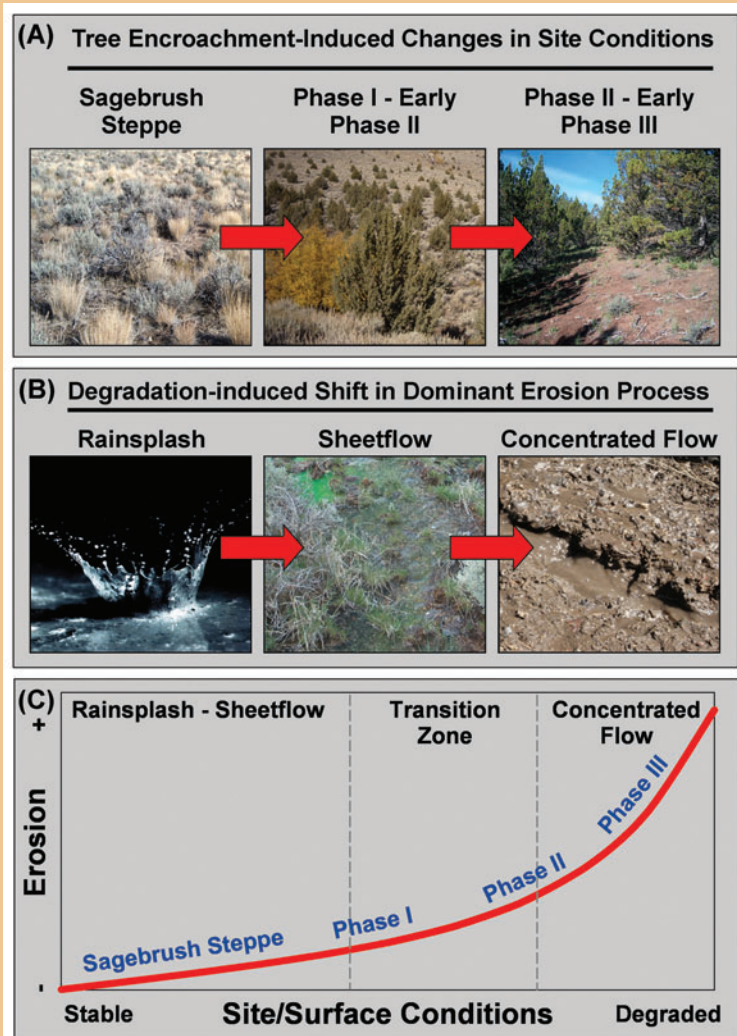
rangelands generate increasing erosion with increasing land area due to connectivity of runoff and erosion sources, while well-vegetated rangelands exhibit decreasing erosion with increasing land area (Wilcox and others 1996; Pierson and others 2009, 2010, 2013; Williams and others in press). For example, soil loss is commonly minor from undisturbed, well-vegetated sagebrush communities with >12 inches (>300 mm) annual precipitation (Pierson and others 1994, 2008a, 2009). Conversion of sagebrush plant communities to pinyon and juniper woodlands, however, can increase the spatial extent of bare ground and amplify runoff and erosion (figs. 46A, B, and C; Miller and others 2005; Pierson and others 2010, 2013). Increased runoff and erosion by water following woodland encroachment is associated with concentration of runoff from **rainsplash/sheetflow** source areas into high velocity flow paths through contiguous bare intercanopy areas (Pierson and others 2007, 2010, 2013; Williams and others 2013). Slope angle, inherent soil properties, climate regime, and degree of plant community degradation govern the magnitude of runoff and erosion (Davenport and others 1998). Steeply sloped communities with highly erodible surface soils are less resistant to soil loss and are therefore highly sensitive to minor changes in vegetation and ground cover. In contrast, gently sloping, highly erodible soil surfaces may experience low water erosion rates. Such locations may be highly susceptible to wind erosion however (Sankey and others 2009, 2012a). Finally, the **resilience** of plant communities with respect to surface water and soil retention are strongly affected by ecological site characteristics, land use activities, and disturbance history that sustain well-distributed vegetation recruitment, plant productivity, and ground cover (Petersen and Stringham 2008; Williams and others 2013) (fig. 12).

Fire Effects on Runoff and Erosion

There has been considerable work conducted in the Great Basin Region evaluating fire effects on hydrologic processes and sediment movement at the point to small plot (<20 ft^2 [<2 m^2]) and large-plot or patch (107-323 ft^2 [10-30 m^2]) scales (Pierson and others 2001, 2002, 2008a, 2009, 2011, in preparation; Sankey and others 2009, 2012b; Williams and others in press). The majority of this work, with the exception of movement by wind, was conducted using rainfall simulation methods. Studies from sagebrush shrublands indicate runoff and erosion by water may increase 3-fold and 40-fold, respectively immediately post-fire (prior to vegetation growth) over small-plot scales (Pierson and others 2011) and that large-plot scale runoff and erosion may increase 7-fold and 125-fold, respectively following burning (Pierson and others 2009). Soil loss rates by wind are less understood. Sankey and others (2009) measured 12 lbs/10 ft^2 (5.40 kg/m^2) of soil loss three months following burning of Great Basin sagebrush shrubland; wind erosion from an adjacent unburned area was negligible. Sankey and others (2012b) reported dust emission declined to near pre-burn levels in the first year following fire. Few studies have evaluated the impacts of fire on the hydrologic processes and movement of sediment at landscape or watershed scales (e.g., paired watersheds). However, numerous anecdotal reports have documented large-scale flash flooding and debris flow events following rangeland wildfires (Pierson and others 2011).

Fire primarily alters runoff and erosion processes by decreasing resistance to surface flow and increasing the hydrologic and erosion susceptibility of the soil surface (fig. 47; Pierson and others 2008b, 2009, 2011, 2013; Sankey and others 2009). Surface susceptibility is a function of ground cover, surface roughness, soil erodibility, soil water repellency, and slope. The first order effect of increased susceptibility is amplified water availability for runoff generation. Fire removal of canopy and ground cover reduces rainfall interception and surface water detention, promotes rapid runoff generation, and decreases ground surface protection against raindrop impact and soil detachment by overland flow and wind.

Figure 47. (A) Change in vegetation and ground surface conditions with post-fire recovery; and **(B)** the associated decline in erosion and shift in dominant erosion processes with decreasing surface susceptibility during post-fire recovery. Bare, water-repellent soil conditions in the immediate post-fire period facilitate runoff generation and promote formation of high-velocity concentrated flow. Erosion occurring immediately post-fire typically increases with increasing land area due to high rates of erosion from concentrated flow processes. Runoff from commonly occurring storms typically declines to near pre-fire levels within one to two years post-fire. Erosion usually remains elevated above pre-fire levels for two to three years post-fire. The decline in runoff and erosion with time post-fire is strongly related to changes in ground surface conditions that trap and store water and sediment. Although runoff and erosion rates commonly approach pre-fire levels within the first three years post-fire, burned rangelands remain susceptible to amplified runoff and soil loss from extreme events until overall site conditions (e.g., vegetation and litter biomass and well aggregated soils) return to near unburned conditions.
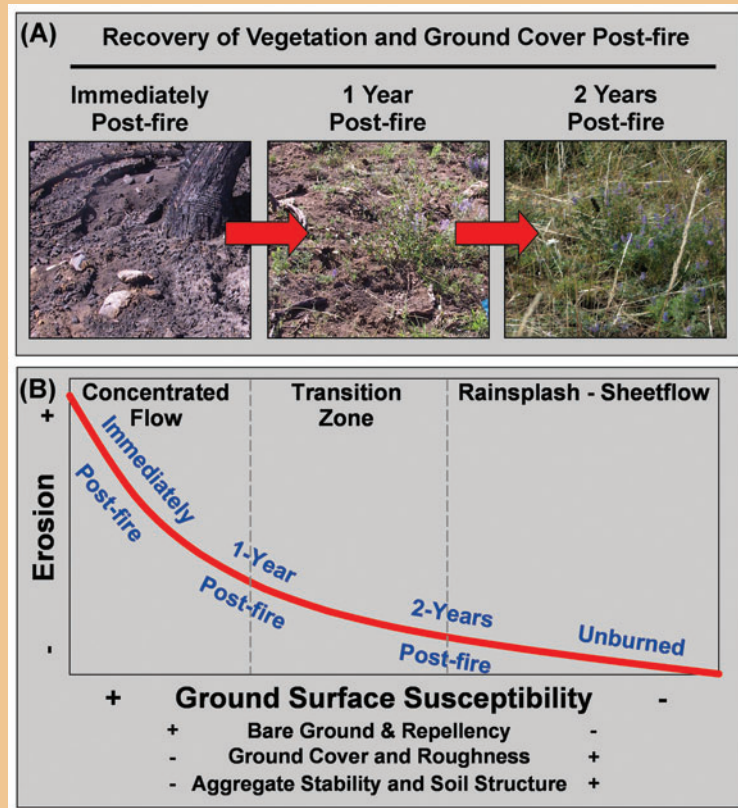
Fire effects on infiltration and runoff generation are exacerbated where soil water repellency persists post-fire or is enhanced by burning (Pierson and others 2008b, 2009, in press; Madsen and others 2011). Soil water repellency is commonly found within the first 0 to 2 inches (0 to 50 mm) of soil depth underneath unburned sagebrush and conifer litter on Great Basin rangelands (Rau and others 2005; Lebron and others 2007; Madsen and others 2008; Pierson and others 2008b, 2009, 2010, in press; Glenn and Finley 2010; Robinson and others 2010; Williams and others 2013), and its strength may increase or decrease with burning (Salih and others 1973; DeBano and others 1998; Pierson and others 2008b). Coarse-textured soils are thought to be more prone to **hydrophobicity**, but water repellent soil conditions also have been well documented for fine-textured soils (Doerr and others 2000, 2006). Fire-induced increases in runoff and soil losses are typically greater from areas underneath shrubs and trees than interspaces between woody plant canopies. Canopy locations commonly have greater post-fire sediment availability (Al-Hamdan and others 2012) and stronger soil water repellency than interspaces between canopies (Pierson and others 2008b, 2009; Madsen and others 2011; Williams and others 2013).

Amplified runoff over small-plot scales facilitates formation of highly erosive **concentrated flow** and increasing soil loss over large-plot to hillslope scales. Homogenous bare soil conditions in the immediate post-fire period allow overland flow to concentrate into high velocity flow paths with greater erosive energy and transport capacity than processes occurring at small-plot scales (Pierson and others 2009; Al-Hamdan and others 2013). Concentrated flow transports **rainsplash** and **sheetflow** detached sediment downslope while also eroding sediment from within the flow path. Concentrated flow is the dominant water-based erosion process in the first one to two years post-fire and is accentuated by steep, bare hillslopes and converging topography (Pierson and others 2009; Al-Hamdan

and others 2012). Progressive sediment-bulking of concentrated flow over hillslope scales can result in resource-, property-, and life-threatening mass erosion events. For example, a 9-minute convective rainstorm on burned rangeland hillslopes along the Boise Front Range, Idaho, generated flooding and mud-flows in the City of Boise (Pierson and others 2002). The flooding was driven by intense rainfall and formation of concentrated flow on bare, strongly water repellent soils with reduced water storage capacity and low surface roughness. A similar hydrologic response to high-intensity rainfall occurred in adjacent burned areas of the Boise Front approximately 40 years earlier, causing more than $3 million in property damage (Klade 2006). Analogous hydrologic and erosion responses to convective storms have been reported for burned cheatgrass communities and woodlands in Utah and Colorado (Craddock 1946; Cannon and others 1998, 2001). The likelihood or risk of such large-scale flooding events is related to the spatial connectivity of susceptible soil conditions and the occurrence probability of runoff generating rainfall (Pierson and others 2011, Pierson and Williams in preparation). Great Basin plant community transitions (e.g., cheatgrass and red brome) and climate trends that promote wildfire activity increase the spatial and temporal exposure of rangelands to runoff and erosion generating storms and thereby likely enhance long-term soil loss associated with frequent re-burning (Wilcox and Thurow 2006; Sankey and others in 2012b; Williams and others in press).

Post-Fire Hydrologic Recovery and Resilience

The relative hydrologic recovery and **resilience** of Great Basin plant communities is primarily influenced by the pre-fire ecological **state**, **fire severity**, and post-fire weather and land use that relate to vegetation recovery (fig. 15). The pre-fire ecological state influences spatial variability in burn severity and post-fire plant recruitment. High severity burns on productive mountain big sagebrush communities may consume nearly 100% of canopy and ground cover, but runoff and erosion can return to pre-fire levels within a few years post-fire (Pierson and others 2011). Rainfall simulations in burned mountain sagebrush communities indicate that small- to large-plot scale runoff post-fire returns to pre-fire levels within one growing season (Pierson and others 2001, 2002, 2008a, 2009). Also, post-fire soil erosion returns to near pre-fire levels once bare ground declines to near 60%, usually within two to three growing seasons depending on post-fire precipitation. Other rangeland studies in the Great Basin indicate bare ground commonly returns to pre-fire levels within two to four years (Barney and Frischknecht 1974; Bates and others 2009, 2011; Miller and others in press). Although relative hydrologic recovery appears to occur within one to three years post-fire, rangelands likely remain susceptible to runoff and erosion during extreme events until overall characteristics (e.g., live plant and litter biomass) are consistent with pre-fire conditions. Rangelands with warm/dry soil temperature/moisture regimes may require longer periods to recover hydrologically than cool/moist ecological sites and may be vulnerable to cheatgrass invasion and subsequent re-burning. Hydrologic recovery and resilience of woodland-encroached sagebrush communities have received only minor attention in the literature. A recent rainfall simulation study by Pierson and others (2013) found that burning of a **Phase II-III** woodland on a mountain big sagebrush **ecological site** resulted in large-plot scale runoff and erosion 4- and 20-fold higher than areas underneath tree canopies the first year post-fire. Overland flow experiments at the study site found erosion remained elevated underneath burned junipers two years post-fire due to delayed plant recruitment and bare ground persistence (Pierson and others 2013). Burning had no effect on large-plot runoff and erosion within the intercanopy the first year post-fire. Two years post-fire, overland flow experiments yielded less erosion from burned than unburned intercanopy areas. Pierson and others (in press) attributed the treatment effects to well-distributed intercanopy herbaceous recruitment post-fire. Williams and others (2013) found burning may represent a potential

restoration pathway for woodland encroached sagebrush steppe on cool/moist ecological sites. However, less productive woodland encroached ecological sites or ecological sites with minimal pre-fire herbaceous cover may exhibit less hydrologic resilience post-fire with respect to Phase II woodlands and intact sagebrush communities. Long-term data are scant regarding post-fire hydrologic resilience of Great Basin plant communities. Regardless of the soil temperature/moisture regime and pre-fire state, short-term post-fire hydrologic recovery is likely delayed by land use activities and/or drought conditions that inhibit vegetation and ground cover recruitment.

Fire Surrogates

Fire surrogates such as shredding trees or shrubs with mechanical equipment (e.g., brush hog), cutting and falling trees with chainsaws (fig. 42), or cutting and piling, are alternatives to using prescribed fire and can have very different effects on hydrologic processes. The impact on hydrological processes will be directly related to how mechanical treatments change the density and pattern of vegetation, alter soil surface roughness, and increase soil compaction. In Utah, application of mulch from mechanically shredded Utah juniper to bare interspaces increased infiltration by more than 70% and decreased erosion by nearly 90% at small-plot scales (Cline and others 2010). However, infiltration was nearly 60% lower in tire-compacted than non-compacted grass interspaces. Infiltration and erosion on juniper **duff** and shrub mounds were not affected by the mulch treatment or tire compaction. Soil texture ranged from gravelly-loam to loamy-skeletal. The study suggests that mechanical shredding treatments aimed at improving hydrologic stability should focus on distributing shredded tree debris into bare intercanopy areas while minimizing soil compaction. In New Mexico, slash treatments reduced sediment yields from 1.1 and 1.14 tons/acre to 0.17 and 0.4 tons/acre (2.00 and 2.07 Mg/ha to 0.3 and 0.7 Mg/ha) in two separate years across 6-paired watersheds (Hastings and others 2003). Ground cover ranged between 30 and 60% in the first year and 50 and 60% in the second year on treated watersheds compared to <10% on untreated watersheds in both years. In southeastern Oregon, sediment yields and runoff were compared between 8-paired 325 feet^2 (32.5 m^2) plots 10 years after juniper cutting (Pierson and others 2007). Total herbaceous and litter cover was 42% in the treated and 8% in the untreated plots (trees remained standing). Cumulative sediment yield was 85-fold greater in the untreated woodland after 60 minutes of rainfall. There was no runoff on the treated plots in the first 30 minutes and only a small amount in the next 30 minutes. However, on the control plots, water began running off immediately with nearly 55 inches (1400 mm) of cumulative runoff, compared to <0.8 inches (<20 mm) on the treated plots.

Chaining was a method used more extensively in the past for pinyon and juniper control. Results comparing runoff and sediment yields between chaining-with-windrowing, chaining-with-debris-left-in-place, and nearby untreated woodlands were not consistent (Gifford and Tew 1969; Gifford and others 1970; Myrick 1971; Gifford 1973, 1975). Results ranged from higher, lower, and little change in runoff and sediment yields. However, these studies did not report changes in ground surface cover or surface roughness. The effects of chaining will depend largely on slope angle and changes in the abundance and density of ground surface cover, surface roughness, compaction, soil characteristics, and soil disturbance.

Summary

The degree to which fire increases runoff and erosion from Great Basin rangelands is determined by the amount of vegetation and ground cover removed and by inherent ecological site characteristics. Fire-removal of vegetation and litter decreases interception and surface storage, reduces infiltration, and promotes overland flow. Concentrated flow erosion

becomes the dominant soil detachment and transport mechanism immediately following fire. Large, contiguous bare areas facilitate concentration of overland flow into high velocity flowpaths with substantial sediment detachment and transport capacity. Erosion immediately post-fire typically increases with increasing land area due to high rates of erosion from concentrated flow processes. Fire effects on hydrology and erosion increase on steep slopes with highly erodible soils and where water-repellent soil conditions exist. Soil water repellency is naturally occurring underneath Great Basin vegetation with substantial litter accumulation, and its persistence post-fire can significantly amplify runoff generation. Runoff and sediment erosion rates following burning frequently return to pre-fire levels within two to five years for low to moderate intensity storms, depending on plant and litter recruitment. Burned communities may remain susceptible to elevated erosion by concentrated flow during high intensity storms for greater periods of time. There is a close link between post-fire hydrologic and erosion recovery and resilience and resistance of vegetation. The key attributes that impact hydrologic processes and recovery following fire are fire severity, soil surface roughness, water repellency, soil erosion potential, slope angle, rainfall intensity, and the recruitment of spatially, well-distributed canopy and ground cover.

Section 5: Management Framework—Asking the Right Questions

Site **resilience to disturbance** and **resistance to invasives** continually changes along environmental/productivity gradients across landscapes and regions. The ability of ecological sites to recover and resist invasive species is strongly influenced by the current vegetation along this gradient (vegetation composition and structure, bare ground, etc.). Determining resilience and resistance of ecological sites or management units and predicting potential successional trajectories are essential in developing restoration plans and selecting the most appropriate locations and treatments to meet objectives and ensure a high probability of recovery. Managers should carefully assess the characteristics of the key drivers or components that influence resilience, resistance to invasion (figs. 12, 44), and successional trajectories (fig. 15). The overall components and characteristics to be addressed are presented in table 6.

Specific questions designed to determine the ecological site characteristics and treatment effects can be used to evaluate potential ecological responses on a management unit following treatment. Answers to these questions form the basis for determining if an ecological site is suitable for treatment and deciding on the most appropriate treatments. The key questions that form the framework for evaluating resilience to disturbance and resistance to invasive species and for predicting successional trajectories are reported in more detail in Miller and others in preparation a. Since treatments often are applied to relatively large areas, landscape heterogeneity will produce varying responses across the treated area. Thus, these questions should be addressed for each of the ecological sites within the management unit. The type(s) of treatments and any post-treatment management activities should be tailored to the specific ecological sites and may vary across the unit.

1. Ecological Site Characteristics

Temperature regime

(1) What is the soil temperature regime?

(2) Within the identified regime, is the ecological site on the warm, mid (typic), or cool portion of the gradient?

Moisture regime

(3) Is the ecological site dry-aridic (<10 inches [250 mm] precipitation), aridic (10-12 inches [250-300 mm] precipitation), or xeric (>12 inches [300 mm] precipitation)?

(4) Is the soil depth very shallow, shallow, moderately deep, or deep?

(5) Is the soil texture predominantly loam, clay, or sand?

Potential vegetation

(6) What sagebrush species or subspecies should occupy this ecological site?

(7) Are other shrub species present that are indicators of dry-aridic, aridic, or xeric moisture conditions?

(8) What are, or should be, the potential dominant perennial grasses on the ecological site?

(9) Are old trees or remnants of large stumps and logs present?

2. Current Vegetation as Affected by Disturbance History

(10) Does the current vegetation (phase) fall within the reference state or an alternate state?

(11) If the phase does not fall within the reference state, is it at-risk (invaded state)?

(12) What is the current distribution and abundance of invasive species?

(13) What is the potential for invasive species to increase based on ecological site characteristics, seed source (on- or off-site), and the severity of disturbance?

(14) What is the fire tolerance of native perennial species on the ecological site?

(15) If old trees (>150 years) occur on the ecological site, what is their density and distribution?

　　a. Is the prospective management unit old-growth woodland, or are old trees scattered and infilled with young trees (<150 years old)?

　　b. Is the distribution of old trees associated with topographic position or shallow soil types?

(16) If the community is primarily dominated by young trees, what is the successional woodland phase?

3. Treatment Type and Severity

(17) How will the abundance and structure of current vegetation effect fire severity?

(18) What is the most appropriate treatment (e.g., fire or fire surrogate) based on ecological site characteristics and current vegetation? (Exotics are usually favored more by fire than mechanical treatments, so resistance to invasives is a primary consideration.)

(19) What are the intensity, duration, size, complexity, and time of year of the treatment or disturbance?

4. Pre and Post-Treatment Weather

(20) How have fuel loads and structure been influenced by pre-treatment weather conditions that could influence fire spread and severity?

(21) How have potential seed banks been influenced by pre-treatment weather?

(22) Has post-treatment weather influenced successional trajectories, and are additional actions needed (e.g., invasive species control)?

5. Post-Treatment Grazing

(23) How long will grazing be deferred on the management unit?

(24) What is the post-treatment level of control of grazing in terms of duration, stocking rates, distribution, and season of use?

Table 6. Primary components (from figs. 12, 15, and 43) and their attributes that influence resilience, resistance to invasives, and successional trajectories.

Component	Attributes

Ecological site (figs. 2 & 5)	Regional location (MLRA)
	Climate
	Topography
	  Elevation, aspect, slope, landform, and landscape position
	  (consider how topography effects water movement & storage & heat loads)
	Soils
	  Soil moisture and temperature regimes
	  Depth, texture, % organic matter, structure
	  (consider factors that influence water storage and availability)
	Potential vegetation within the reference state
	  Species composition and structure (e.g., biomass, cover, and density)
	  Potential production in favorable, average, and unfavorable years

Current vegetation	Vegetation productivity (biomass)
	Species composition and structure relative to the ecological site description
	  Fire tolerant & non-tolerant (morphology)
	  Native & invasive
	Potential for invasives
	  On site and adjacent seed banks, potential seed rain
	Fuel load and structure
	  Woodland phase (fire severity increases with increased tree biomass)
	  Grass crown litter
	  (biomass, continuity, packing ratios)
	Woodland age structure (pre- & post-settlement tree densities)
	Amount and distribution of bare ground
	Amount and distribution of biological soil crusts
	At-risk-phase?

Disturbance history (pre & post)	Severity & frequency
	Time since last event
	Type
	  Fire
	  Drought
	  Herbivory (livestock, native and introduced herbivores)
	  Disease, snow-mold, fungus, etc.
	  Insects

Fire severity (fig. 8)	Fuels Topography Fire weather Season (linked with fire weather and plant phenology) Current vegetation (fuel abundance and structure) Fire type Ground, surface, crown, head fire, backfire

Weather	Timing & amount of precipitation (pre-and post-treatment)
	Temperatures (primarily extremes)
	  Consider how it has influenced:
	  Pre-treatment species composition
	  Plant vigor
	  Seed banks
	  Fuels

General Implications

The literature supports the following general implications to consider when evaluating resilience to fire and management treatments, resistance to invasion, and potential successional trajectories of ecological sites:

(1) Mountainous topography in the Great Basin creates environmental gradients in moisture and temperature regimes that are modified by aspect, slope, and soils and that are associated with predictable changes in potential vegetation, productivity, resilience, and resistance to invasive species.

(2) Decreasing temperatures and increasing moisture availability along environmental gradients results in increased resilience of sagebrush communities. However, shortened growing seasons at the cold end of the gradient result in lowered resilience.

(3) Temperature and soil moisture availability determine abiotic conditions under which native and invasive species can establish, grow, and persist. Establishment, growth, and reproduction of annual grasses are limited at the lowest elevations by low and variable precipitation and at high elevations by low soil temperatures.

(4) The size and distribution of bare soil patches within a plant community relative to those expected under reference conditions influence resistance to invasive plants. Increases in the amount of bare soil due to increases in the death of perennial plants or reductions in **biological soil crusts** decrease resistance to invasive annual grasses.

(5) Soil moisture and temperature regimes and pre-treatment species composition are major drivers of resilience to disturbance and resistance to invasive plants. Perennial native herbaceous species are especially important components.

(6) Sites with warm/dry soils are less resilient to disturbance and management treatments than those with cool/moist soils.

a. Although the USDA-NRCS has developed criteria defining the boundaries of elevation, aspect, and amounts of precipitation for these soil moisture and temperature regimes for each **MLRA**, it is important to remember that the regimes occur as continuous gradients on the landscape. These regimes also may occur outside the elevation/precipitation boundaries developed by the NRCS because of other ecological site characteristics that override or modify elevation/precipitation variables (e.g., soil depth or texture).

b. Managers can estimate where along the gradient the management unit fits within each regime. For example, on a dry ecological site (mapped aridic <12 inches [300 mm] PZ), is the mean precipitation 6, 8, 10, or closer to 12 inches (150, 200, 250, 300 mm)? Is the elevation near the elevation break for two **soil temperature regimes** (e.g., warm or cool)?

(7) **Fire severity** affects not only plant mortality but also soil physical and chemical properties. Plant mortality affects post-treatment plant composition, cover, biomass, soil surface litter, and fuel loads. Changes in soil physical properties influence soil hydrologic properties such as **hydrophobicity** and infiltration, soil erosion, and soil organic matter. Changes in soil chemical properties influence soil nutrient content and cycling.

(8) A site-at-risk (**at-risk phase**) is usually a result of low productivity, loss of perennial grasses and forbs, increases in bare ground, presence of invasive species, and conditions that can generate **high severity fire** and/or a change in **fire regime** (e.g., low- to moderate-severity **surface fires** versus less frequent, high severity **crown fires**).

(9) Changes in atmospheric CO_2 and climate warming are likely to shift current interpretations of the effects of elevation and soil temperature/moisture regimes on plants and soils and to have the greatest effects on transition zones between **ecological sites.**

(10) Although parent materials and species may differ across ecological sites and MLRAs across the Great Basin Region, managers can compare general ecological site characteristics that influence resilience to disturbance and resistance to invasives (e.g., soil temperature/moisture regimes and current vegetation composition and structure) when extrapolating results between ecological sites or MLRAs.

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Glossary

Fire History

Mean Fire Return Interval (MFRI) (also "mean fire free interval" or "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 = years). Not to be confused with "fire rotation."

Fire frequency—the number of fires per unit time in some designated area (which may be as small as a single point or represent an entire watershed); the size of the area must be specified (units = number/time period/area).

Fire interval (also "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 (also "fire incidence")—one fire event taking place within a designated area during a designated time (no units; either yes a fire occurs, or 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 Behavior

For additional fire-related definitions see:

- *The Fire Effects Information System Glossary* at: http://www.fs.fed.us/database/feis/glossary.html
- *The Forest Encyclopedia Network* at: www.forestencyclopedia.net/p/p773
- Brown and others 2000 (Glossary pages 248-249)

Backfire—a fire set along the inner edge of a fireline to consume the fuel in the path of a fire or to change the fire's convection column; backfires often move slowly since they are usually burning into the wind resulting in longer duration of the flame front.

Backing fire—usually moving into the wind or downslope; moves slowly through the fuelbed consuming most of the available fuel so that the fire quickly dies out with very little smoldering. In a heading fire, the flaming front passes quickly and the ignited fuels continue to smolder until consumed.

Burn barrel—a portable combustion chamber typically used in the field for applying fire to small plots. Chamber diameters commonly range from 24 inches (610 mm) such as a 55 gallon barrel used by Wright and Klemmedson (1965) to 13 ft (4 m) by using four pieces of sheet metal to fence in the fire.

Crown fire—a 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 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 intensity—a general term relating to the heat energy released in a fire; the amount and rate of surface fuel consumption.

Fire severity—the effects of fire on ecological processes, soil, flora, and fauna; degree to which a community 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.

Head fire—front of a spreading fire usually moving with the wind or upslope.

High severity fire—typically infrequent fires (>100 year MFRI), high intensity, stand-replacing fires. Dominant vegetation structure, composition, and function are all markedly charged by the incident. Consumption of light surface fuels >80% with unburned patches <15% (see table 2).

Hydrophobicity—a condition that can be created by fire that temporarily increases water repellency of the soil, resulting in increases in runoff and soil erosion.

Low severity fire—typically frequent fires (1-25 year MFRI) that produce only slight or no modification of vegetation structure; most mature plants survive. Usually <50% consumption of light surface fuels and little to no ash is formed, with ground fuels blackened but recognizable (see table 2).

Moderate severity fire—typically infrequent fires (25-100 year MFRI); partial stand-replacing fires, including burned areas ranging from low to high severity. Usually 50 to 80% consumption of light surface fuels, unburned patches 15-50%, thin layer of ash (see table 2).

Packing ratio—a measure of fuelbed compactness that is defined as the fraction of the fuel array volume occupied by fuel. Packing ratio is an mportant factor in predicting fire behavior. At very low packing ratios (or fuel densities), fire spread is limited and fire intensities are therefore low. At very high packing ratios, lack of oxygen limits fuel combustion.

Residence time—the amount of time a flaming front takes to pass over a point (duration).

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).

Community Ecology

Alliance—a physiognomically uniform group of plant associations sharing one or more dominant or diagnostic species that, as a rule, are found in the uppermost stratum of the vegetation; typically designated by a diagnostic overstory species.

Aridic—see "soil moisture regime."

At-risk phase—a community phase that is highly vulnerable to transition to an alternative state (i.e., least resilient). See also "phase."

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

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

Biome—A large, relatively distinct terrestrial region, encompassing many interacting ecosystems and characterized by similar climate, soil, plants, and animals. A biome is commonly named for its plant cover.

Concentrated flow—relatively continuous, narrow flow paths of high-velocity overland flow capable of surface incision and transportation of substantial amounts of sediment.

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

Cover type—a classification based on the dominant or co-dominant overstory species that predominate in a particular area. The type classification is named for that species.

Cryic—see "soil temperature regime."

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: 7).

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 in its response to management. Synonymous with "ecological type" used by the USDA Forest Service, and with "ecological site" used by the USDA Agricultural Research Service and Natural Resources Conservation Service.

Ecoregion—areas with generally similar ecosystems and 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.

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."

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

Great Basin Region—a floristically similar area that includes the northern two thirds of the hydrologic Great Basin, the Columbia River Basin, and the Snake River Plain. This is the Region covered by the synthesis.

Habitat type—see "plant association."

Herb—(also "herbaceous species")—refers to the non-woody vascular plant component that includes both perennial and annual grasses and forbs; aboveground stems usually die back at the end of the growing season.

Historic 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).

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 arrangements giving each landscape a unique pattern" (Urban and others 1987).

A mosaic of land forms, vegetation, and land uses; varying in extent with the organism(s) of interest, objectives, or purpose.

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

Major Land Resource Areas/MLRAs—geographically associated land resource units, usually encompassing several thousand acres. MLRAs 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 temperature regime."

Obligate—essential or necessary.

Paralithic—weathered layer of bedrock that becomes soft or crumbly.

Patch—a homogeneous unit of area; frequently, patches are defined as distinct plant communities that reoccur across the landscape.

Persistent pinyon-juniper woodlands—found where 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. Notably, these woodlands do not represent Twentieth Century conversion of formerly non-woodland vegetation types to woodland but are locations where trees have been an important stand component for at least the past several hundred years.

Phase (community)—a plant community within a state that is hypothesized to replace other communities along traditional succession-retrogression pathways; transition from one community to the next is readily reversible over short time periods (years to decades) without management intervention because phases 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; *phase II*: trees are co-dominant 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 (from Miller and others 2005). Phases can be calculated using % cover (from Tausch and others, in preparation).

- Phase I = total tree / total tree + shrub + perennial grass = <0.33 (tree biomass <1/3)
- Phase II = total tree / total tree + shrub + perennial grass = 0.34-0.65 (tree biomass 1/3 to 2/3)
- Phase III = total tree / total tree + shrub + perennial grass = >0.66 (tree biomass >2/3)

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 diagnostic overstory and understory species (e.g., ponderosa pine/Idaho fescue, mountain big sagebrush/bluebunch wheatgrass). "Relatively stable, fully developed (mature) communities" (Tansley 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 1939).

Precipitation zone/PZ—used by the Natural Resources Conservation Service when mapping soils and ecological sites.

Rainsplash—the transport of water and soil particles resulting from raindrop impact with the ground surface.

Reference site—a location (stand or watershed) that demonstrates a reference state.

Reference state—historic or potential plant community, including seral (successional) stages; based on conditions believed to be present before widespread alterations by Euro-Americans.

Resilience—the capacity of an ecosystem to regain its fundamental structure, processes, and functioning when subjected to stressors or disturbances such as 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.

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.

Semi-arid climate—regions that receive precipitation below potential evapotranspiration but not extremely; intermediate between arid and humid, often supporting shrubs and grasses compared to arid that supports scrubby vegetation.

Seral (sere)—a plant community that occurs during ecological succession. Seral stages for a ecological 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.

Sheetflow—a shallow film of water flowing over the ground surface with minor to moderate sediment detachment and transport capacity.

Soil moisture regime—considered an important soil property in combination with season 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 (<12 inches [300 mm] or >12 inches), soil moisture regimes occur along continuous gradients and change with location and elevation. It is important to consider where the ecological site fits along the gradient (e.g., a ecological site with an aridic moisture regime that receives 11.5 inches [290 mm] of precipitation will often be more resilient to disturbance than an aridic ecological site receiving 9 inches [230] of precipitation. For a detailed definition and description of each soil regime, see USDA-NRCS (1999).

a. **Aridic** (dry; mapped at <12 inches [300 mm])—dry in all parts of the profile more than half the time when soils are >41 °F at 20 inches [510 mm]. In the Great Basin and Columbia Basin, aridic soils are usually mapped in precipitation zones receiving <12 inches [300 mm]. However, soils (e.g., lithic or shallow) with limited storage capacity may be mapped as aridic even though the ecological site may receive >12 inches [300 mm] of precipitation. Indicator plants are desert shrub, black sagebrush, and Wyoming big sagebrush.

b. **Xeric** (moist; mapped at >12 inches [300 mm])—soils are moist in some part of the profile more than 50% of the time that soils are >41 °F, or 6 or more years out of 10 are 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 >12 [300 mm] 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 [300 mm] and/or near the boundary of frigid and cryic soils.

Soil temperature regime—an important property of a soil that influences plant growth and biological soil processes along with soil moisture. Usually measured at the 20-inch (50-cm) depth (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).

a. **Mesic** (warm)—mean annual soil temperature is >47 °F but <59 °F and the difference between mean summer and winter soil temperature is >9 °F at 20 inches [510 mm] soil depth. The elevation break between mesic and frigid varies with MLRA; (e.g., Malheur High Plateau: 4000 ft on the flat, + 500-ft adjustment for north or south aspect; Central Nevada Basin and Range and the Great Salt Lake Basin MLRAs: mesic soils are mapped up to 5500 and 6500 ft). Wyoming big sagebrush, black sagebrush, and desert shrub most frequently occupy mesic soils. Mesic soils have a low relative resistance to invasives compared to frigid and cryic soils. They are also considered to have lower resilience.

b. **Frigid** (cool)—mean annual soil temperature is <47 °F and the difference between mean summer and winter soil temperature is >9 °F at 20 inches [510 mm] soil depth. Usually mapped above 4000 ft (+ 500 ft depending on south or north aspect) in the Malheur High Plateau and between 6000 and 8500 ft 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.

c. **Cryic** (cold)—mean annual soil temperature is >32 °F but <47 °F and is cooler in the summer than frigid soils. Usually mapped above 6000 ft (+ 500 ft depending on south or north aspect) in the Malheur High Plateau and above 8200 ft in the Central Nevada Basin and Range and the Great Salt Lake Basin MLRAs. Indicator species are curleaf mountain mahogany, white and grand fir, limber pine, and lodgepole and white bark pine, which typically intermingles with mountain big and low sagebrush. Resistance to invasive species is higher in cryic soils than in mesic and likely frigid soils (although limited data to support). Resilience is also high on the warm end of this regime but declines as colder temperatures limit plant growth.

Special habitats—native biological communities or ecosystems that are rare, unique, or highly productive elements of regional landscapes (Salwasser 1990).

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.

Stochastic event—events whose behavior is intrinsically non-deterministic; random and hence unpredictable.

Subclimax—where natural disturbance (e.g., fire) maintains vegetation different from the climatic climax conditions.

Succession—a continuous process of change in vegetation that can be separated into series or phases (Tansley 1935). A predictable, directional, and stepwise progression of plant assemblages that culminates with climax (Clements 1936).

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 (Beisner and others 2003; Stringham and others 2003).

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.

Xeric—see "soil moisture regime."

Appendix 1. Common and scientific plant names used in the text are from the Natural Resources Conservation Service Plants Database: http://plants.usda.gov/java/.

Common name	Scientific name
alkali cordgrass	*Spartina gracilis*
alkali sacaton	*Sporobolus airoides*
arrowleaf balsamroot	*Balsamorhiza sagittatata*
aspen	*Populus tremuloides*
Bailey's greasewood	*Sarcobatus baileyi*
barbwire Russian thistle	*Salsola paulsenii*
basin big sagebrush	*Artemisia tridentata spp. tridentata*
bitterbrush	*Purshi tridentata*
black greasewood	*Sarcobatus vermiculatus*
black sagebrush	*Artemisia nova*
bluebunch wheatgrass	*Pseudoroegnaria spicata*
maiden blue eyed Mary	*Collinsia parviflora*
bottlebrush squirreltail	*Elymus elymoides*
bristle cone pine	*Pinus longaeva*
broom snakeweed	*Gutierrezia sarothrae*
buckwheat	*Eriogonum*
budsage	*Picrothamnus desertorum (Artemsia spinescens)*
cheatgrass	*Bromus tectorum*
columbia needlegrass	*Achnatherum nelsonii*
curleaf mountain mahogany	*Cercocarpus ledifolius*
curveseed butterwort	*Ceratocephala testiculata*
desert alyssum	*Alyssum desertorm*
desert bitterbrush	*Purshia glandulosa*
desert gooseberry	*Ribes velutinum*
desert wheatgrass	*Agropyron desertorum*
Douglas-fir	*Pseudotsuga menziesii*
flatspine stickseed	*Lappula occidentalis*
fourwing saltbush	*Atriplex confertifolia*
Gardner's saltbush	*Atriplex gardneri*
grand fir	*Abies grandis*
green rabbitbrush	*Chrysothamnus viscidiflorus*
halogeton	*Halogeton glomerata*
Idaho fescue	*Festuca idahoensis*
Indian ricegrass	*Acnatherum hymenoides*
June grass	*Koeleria macrantha*
James' galleta	*Pleuraphis jamesii*

largeflower hawksbeard	*Crepis occidentalis*
Lemmon's needlegrass	*Achnatherum lemmonii*
Lettermen's needlegrass	*Achnatherum lettermanii*
limber pine	*Pinus flexilis*
little (low) sagebrush	*Artemisia arbuscula*
lodgepole	*Pinus contorta*
maiden blue eyed Mary	*Collinsia parviflora*
Mexican cliffrose	*Purshia mexicana*
mountain big sagebrush	*Artemisia tridentata spp. vaseyana*
mountain brome	*Bromus marginatus*
mule-ears	*Wyethia mollis*
mutton grass	*Poa fendleriana*
needle and thread	*Hesperostipa comata*
Nevada Mormon tea	*Ephedera nevadensis*
Nuttall's saltbush	*Atriplex nuttallii*
pale alyssum (madwort)	*Alyssum alyssoides*
phlox	*Phlox*
pine needlegrass	*Achnatherum pinetorum*
pussytoes	*Antennaria*
red brome	*Bromus rubens*
red fescue	*Festuca rubra*
ripgut brome	*Bromus diandrus*
rubber rabbitbrush	*Ericameria nauseosa*
Russian thistle	*Salsola kali*
saltgrass	*Distichlis spicata*
sand dropseed	*Sporobolus cryptandrus*
Sandberg bluegrass	*Poa sandbergii*
sandwort	*Arenaria*
serviceberry	*Amelanchier alnifolia*
shadscale	*Atriplex confertifolia*
Sierra juniper	*Juniperus occidentalis spp. australis*
silver sagebrush	*Artemisia cana*
silvery lupine	*Lupinus argenteus*
singleleaf pinyon	*Pinus monophylla*
slender phlox	*Microsteris gracilis*
snowberry	*Symphoricarpos sp.*
snowfield sagebrush	*Artemisia spiciformis*
spike fescue	*Leucopoa kingii*
spiny hopsage	*Grayia spinosa*
Stansbury cliffrose	*Purshia stansburiana*
sulphur-flowered buckwheat	*Eriogonum umbellatum*

tailcup lupine	*Lupinus caudatus*
tansy-mustard	*Descurainia pinnata*
tapertip hawksbeard	*Crepis acuminata*
thorn skeleton weed	*Pleiacanthus spinosus*
three-tip sagebrush	*Artemisia tripartite*
Thurber's needlegrass	*Achnatherum thurberianum*
Torrey's saltbush	*Atriplex torreyi*
tortual	*Tortula ruralis*
tumble mustard	*Sisymbrium altissimum*
Utah juniper	*Juniperus osteosperma*
wax current	*Ribes cereum*
western juniper	*Juniperus occidentalis spp. occidentalis*
western needlegrass	*Achnatherum occidentale*
white bark pine	*Pinus albicaulis*
white fir	*Abicies concolor*
winterfat	*Krascheninnikovia ceratoides*
woolypod milkvetch	*Astragalus purshii*
Woods' rose	*Rosa woodsii*
Wyoming big sagebrush	*Artemisia tridentata spp. wyomingensis*

Appendix 2. Raunkiaer life form classification system based on location of the perennating tissue (vegetative growing point that persists during unfavorable growing conditions; e.g., summer drought or winter).

Life form	Location of perennating tissue	Plant types
Phanerophyte	>0.5 m	Trees and tall shrubs
Chamaephyte	0 to 0.5 m	Small shrubs and herbs
Hemicryptophyte	Soil surface	Prostrate shrubs and herbaceous plants that dieback each year
Cryptophyte	In the soil	Rhizomatous grasses or bulb forming herbs
Therophyte	Seed	Annuals

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