Chapter 7: Effects of Climate Change on Nonforest Vegetation

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Introduction

Nonforest ecosystems, as they are addressed in this chapter, contain woodland, shrubland, herbaceous, wetland, or riparian vegetation types. They are estimated to occupy over 30 million acres and 50 percent of the U.S. Department of Agriculture Forest Service (USFS) Intermountain Region (table 7.1). These diverse ecosystems range in elevation from desert floors to mountain peaks above 11,000 feet and occupy a wide variety of sites, from deep and highly productive soils to very shallow nonproductive soils. Other than riparian and wetland ecosystems, nonforest vegetation types tend to occur in more arid environments or are otherwise controlled by edaphic features such as soil depth, drainage, or chemical (saline) characteristics.

The diversity and varied conditions of nonforest ecosystems in the Intermountain Adaptation Partnership (IAP) region present challenges for studying the effects of climate change. These ecosystems have been exposed to a wide variety of uses and impacts, resulting in varied ecological conditions across landscapes. Some ecosystems will be less resilient to environmental changes such as increasing carbon dioxide and other greenhouse gases, warming temperatures, and altered amount and timing of seasonal and annual precipitation. The objective of this chapter is to provide insight into the climate change vulnerability of nonforest ecosystems in the IAP region. Climate change vulnerability can be defined as the degree to which a system is susceptible to and unable to cope with adverse effects of climate change (IPCC 2007). This information is intended to provide a basis for developing adaptation actions to increase resilience of nonforest ecosystems in the IAP region (Chapter 14).

Vulnerability Assessment Methods

Climate change vulnerability is a function of the exposure of a system, its sensitivity, and its adaptive capacity (IPCC 2007). In a climate change context, exposure can be thought of as the degree, duration, or extent of deviation in climate to which a system is exposed. Sensitivity is the degree to which a system is affected, either positively or negatively and directly or indirectly, by climate-related stimuli (IPCC 2007). Adaptive capacity is the ability of a system to adjust to climate change (including climatic variability and extremes) by moderating potential damages, taking advantage of opportunities, or coping with the consequences (IPCC 2007).

In considering the potential vulnerability of nonforest ecosystems to the effects of climate change in the IAP region, we modeled our assessment on work done by NatureServe (Comer et al. 2012) for the Mojave and Sonoran Deserts. NatureServe evaluated sensitivity of ecosystems to the direct effects of climate change, as well as their resilience to climate change (based on landscape condition, invasive species, and adaptive capacity). The combined relative ratings for sensitivity and resilience were used to determine climate change vulnerability by the year 2060 for each vegetation cover type.

For our nonforest vegetation vulnerability assessment, a team of experts evaluated sensitivity and adaptive capacity (as already defined). Adaptive capacity incorporates factors such as landscape condition, characteristic species and genetic diversity, and occurrence of invasive species. For example, degraded landscape condition, loss of native species and genetic diversity, and high abundance of invasive species would lower the adaptive capacity of an ecosystem. We relied on published literature and expert evaluations to establish a broad rating system that included five categories for evaluating the sensitivity and adaptive capacity of the vegetation cover types discussed in this report (table 7.2). We also developed numerical vulnerability scores, which combined sensitivity and adaptive capacity (e.g., a value of 5 was used for high sensitivity and low adaptive capacity and a value of 1 was given to low sensitivity and high adaptive capacity) (table 7.2). This system creates some transparency in the assessment process and provides a means to update the assessment with new information as it becomes available.

Vulnerability of Nonforest Ecosystems to Climate Change

Many of the rangelands in the IAP region have sustained, at one time or another, unmanaged livestock grazing. In 1902, Albert Potter, a staff member under Gifford Pinchot,

Table 7.1—Amount of non-forest vegetation cover types in the IAP region, developed from LANDFIRE data.

	IAP Region	Middle Rockies	Southern Greater Yellowstone	Uintas and Wasatch Front	Plateaus	Great Basin and Semi Desert
	Percent					
Forest	49.3	62.0	65.5	55.4	45.6	15.0
Non-forest	50.7	38.0	34.5	44.6	54.4	85.0
Pinyon-juniper shrublands and woodlands	12.6	0.0	0.0	4.9	29.2	37.0
Oak-maple woodlands	2.2	0.0	0.1	9.7	4.4	0.5
Mountain-mahogany woodlands	2.1	0.1	0.0	2.3	3.2	6.1
Mountain big sagebrush shrublands	13.0	17.9	12.0	13.4	3.5	11.4
Dry big sagebrush shrublands	6.5	2.3	0.5	5.4	3.3	20.2
Mountain shrublands	2.2	3.7	2.8	1.4	1.4	0.3
Dwarf sagebrush shrublands	1.2	0.7	0.0	0.3	4.3	1.6
Blackbrush shrublands	0.5	0.0	0.0	0.0	0.1	2.2
Salt desert shrublands	0.3	0.0	0.1	0.4	0.3	1.1
Grasslands	4.3	9.9	2.1	1.0	0.7	1.9
Subalpine forb	2.5	1.6	7.9	2.4	1.5	0.6
Alpine	1.6	0.5	5.7	1.6	1.0	0.3
Riparian	1.3	0.5	1.6	1.9	1.5	1.8
Wetland	0.6	0.7	1.6	0.0	0.0	0.2

the first USFS chief, evaluated the conditions of forests and rangelands in Utah (Prevedel and Johnson 2005). Potter's diary provides detailed descriptions of the effects of unmanaged sheep and cattle grazing on the vegetation and soils of the forest reserves throughout Utah at that time (Potter 1902). As he traveled from northern to southern Utah, Potter often referred to lands that were "heavily grazed" and "heavily stocked" and described lands that were "badly tramped out" and "bare of vegetation." These historical uses often led to a change in site potential and ecological states. Degraded ecological condition from unmanaged grazing, combined with landscape fragmentation, will render many sites less resilient to changing climate. These sites have lost their diversity in species, structure, and genetic composition, and many plants on these sites have decreased vigor, lowering their ability to respond to and cope with the direct (e.g., increased temperatures) and indirect effects (e.g., increased fire) of climate change.

Other primary management concerns in the IAP region include invasive species and uncharacteristic fire regimes, or fire regimes (intensity, severity, extent, and timing of fire) that differ from those before Euro-American settlement. Many low-elevation sagebrush habitats now have significantly shortened fire return intervals (Balch et al. 2013). Increasing dominance by invasive cool-season, annual grasses has created a positive feedback cycle, characterized by frequent fire followed by increased dominance of annual grasses creating fuel conditions that facilitate combustion (Balch et al. 2013). The invasive species of greatest concern is cheatgrass (*Bromus tectorum*), although other invasive annuals such as medusahead (*Taeniatherum caput-medusae*) are growing concerns throughout the region. The expansion of these and other species may be supported by elevated atmospheric carbon dioxide concentrations, increased area burned, and increased soil disturbance (Chambers et al. 2014; Nowak et al. 2004). In addition, the frequent-fire cycle may be exacerbated by wetter and warmer winters, which promote cool-season grass growth (fuel production), increased fuel levels and continuity, and increased area burned (if ignitions occur) (Bradley et al. 2016). Where improper grazing occurs, it can also accelerate annual grass invasion, resulting in changes in the fire cycle, especially in the drier sagebrush types.

Land use legacies, coupled with changing climate, pose unique challenges for managers in the region. Potential interactions between land use change, management, and climate change are not well understood, but the extent to which ecosystem resilience has been affected by human

Table 7.2—Vulnerability ratings for sensitivity and adaptive capacity of non-forest cover types in the IAP region, based on published literature and expert evaluations by a team of scientists.

	Sensitivity rating	Sensitivity score	Adaptive capacity rating	Adaptive capacity score	Combined score	Vulnerability
Alpine	H	5	L	5	10	Very High
Dry big sagebrush shrublands	H	5	L	5	10	Very High
Low-elevation riparian	H	5	L-M	4	9	High-Very High
Subalpine forb communities	H	5	M	3	8	High
Persistent pinyon-juniper woodlands	H	5	M	3	8	High
High-elevation riparian	M-H	4	L-M	4	8	High
Mountain-mahogany woodlands	M	3	L-M	4	7	Moderate-High
Mountain big sagebrush shrublands	M	3	L-M	4	7	Moderate-High
Mountain grasslands	M	3	L-M	4	7	Moderate-High
Salt desert shrublands	M	3	L-M	4	7	Moderate-High
Mid-elevation riparian	M-H	4	M	3	7	Moderate-High
Blackbrush	L-M	2	L	5	7	Moderate-High
Dwarf sagebrush shrublands	M-H	4	M-H	2	6	Moderate
Sprouting sagebrush	M	3	M	3	6	Moderate
Oak-maple woodlands	L-M	2	M	3	5	Low-Moderate
Mountain shrublands	L-M	2	M-H	2	4	Low-Moderate

uses will ultimately affect the ability of those ecosystems to respond to changing climate.

Climate change projections for the IAP region (Chapter 3) indicate that average annual minimum and maximum temperatures are likely to increase by 5 to 12 °F, mean annual precipitation will remain the same or increase slightly, extreme events (e.g., drought and extreme precipitation events) will occur more frequently and be more severe, and concentrations of carbon dioxide and other greenhouse gases in the atmosphere will continue to increase through the end of the 21st century. Minimum daily temperatures in the Great Basin in the 20th century increased more than maximum temperatures (Chambers 2008). In addition, these increased minimum daily temperatures have resulted in longer frost-free periods. Projections vary somewhat by subregion, but even where precipitation is projected to increase slightly, higher temperatures are likely to lead to greater effective drought and soil water deficit.

Despite increased moisture stress, net primary productivity (NPP) of vegetation in the IAP region may increase with warming temperatures due to greater water-use efficiency associated with carbon dioxide fertilization effects (Reeves et al. 2014). Projections suggest that there will be a greater increase in NPP in the northern cooler and wetter portions of the IAP region (Southern Greater Yellowstone and Middle Rockies subregions). A short decline in NPP will precede a smaller increase in NPP in the southern warmer and drier portions of the region (Plateaus and Great Basin and Semi Desert subregions) (fig. 7.1). However, the capacity to respond to carbon dioxide fertilization varies greatly among and within plant functional groups, suggesting that changes in NPP will not be expressed uniformly by species within plant communities. Ecosystem response to climate change throughout the IAP region will vary with local site characteristics (e.g., water holding capacity, soil characteristics) and ecological condition.

Paleoecological studies have shown that species respond individualistically and at different rates with changing climates, resulting in reshuffling species associations and novel community combinations (Delcourt and Delcourt 1981; Williams and Jackson 2007). Thus, each species is likely to respond differently to future climatic changes and carbon dioxide fertilization (Anderson and Inouye 2001), depending on physiological tolerances and the competitive ability of the species. Consequently, we are likely to see new vegetation communities in the IAP region under changing climate. However, because vegetation types, or groups of associated species, are widely known and provide a convenient unit of assessment, we discuss climate change effects by vegetation type, highlighting likely species-level responses.

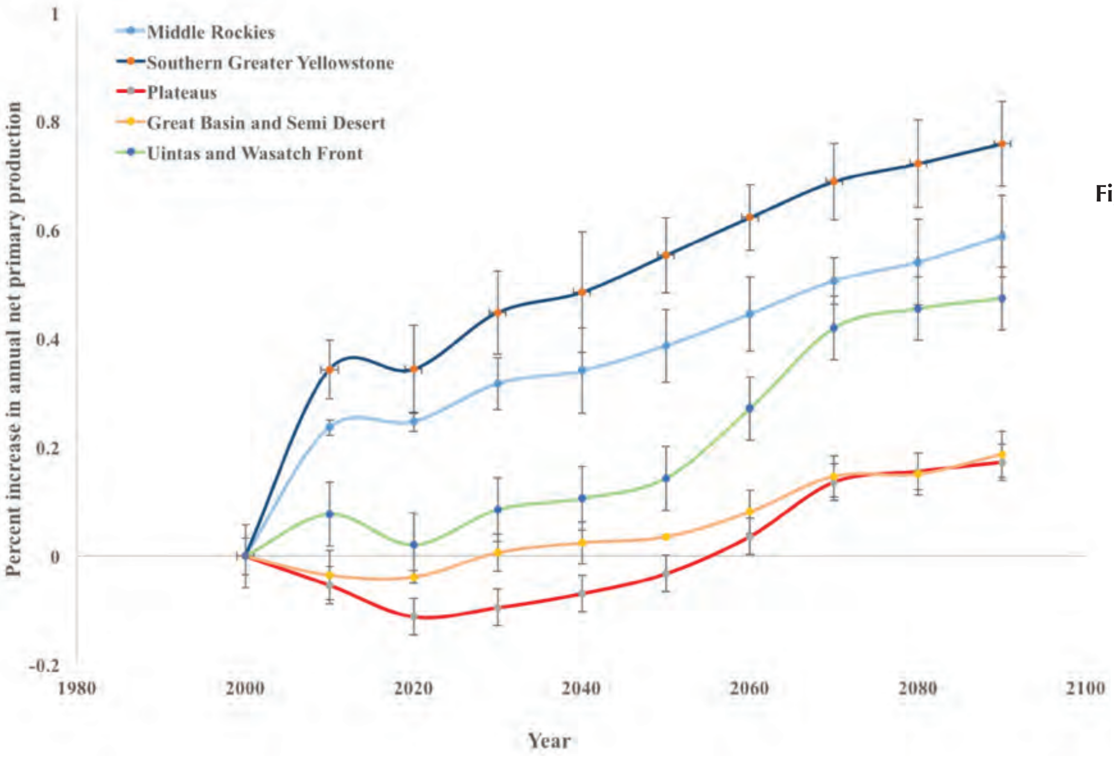

Figure 7.1—Average and standard deviation of net primary production under the A1B, A2, and B2 climate change scenarios for five subregions in the Intermountain Adaptation Partnership region (data from Reeves et al. [2014]).

Woodland Ecosystems

Woodland ecosystems include vegetation stands with at least 10 percent cover of tree species that are typically less than 40 feet tall at maturity, and often less than 16 feet tall on relatively harsh sites. Woodlands, in general, are more abundant in Utah and Nevada than in Idaho or Wyoming (table 7.1). Three woodland types are included in this assessment: persistent pinyon-juniper, oak-maple, and mountain mahogany. Persistent pinyon-juniper woodlands are those dominated by pinyon pine (*Pinus edulis, P. monophylla*) or juniper, either in combination or as individual species. Oak-maple woodlands are dominated by Gambel oak (*Quercus gambelii*) or bigtooth maple (*Acer grandidentatum*), or both; mountain mahogany woodlands are dominated by curl-leaf mountain mahogany (*Cercocarpus ledifolius*). These woodland types generally occur on mid-elevation sites (but can be found on south-facing slopes at higher elevations) and are found on a wide variety of soils.

Persistent Pinyon-Juniper Woodlands

Three general pinyon-juniper vegetation types have been defined based on canopy structure, characteristics of the understory, and historical disturbance regimes (Romme et al. 2009): persistent pinyon-juniper woodlands, wooded shrublands, and pinyon-juniper savannas. Pinyon-juniper savannas are uncommon in the IAP region and are not described here. Persistent pinyon-juniper woodlands are those that were historically dominated by pinyon (singleleaf pinyon [*Pinus monophylla*] or twoneedle pinyon [*P. edulis*]) or juniper, or both, and where fire was rare, usually because of poor soil conditions and low surface fuel levels and continuity. Wooded shrublands are characterized by a dominant shrub component (most notably mountain big sagebrush [*Artemisia tridentata* ssp. *vaseyana*] and Wyoming big sagebrush [*A. t.* ssp. *wyomingensis*], but wooded shrublands also occur in dwarf sagebrush ecosystems). The density of pinyon and juniper in various combinations increases and decreases over time in response to climate and disturbance, including fire and insect outbreaks.

Only those plant communities that historically occurred as tree-dominated sites for a majority of time under pre-Euro-American natural disturbance regimes are included in our persistent pinyon-juniper woodlands discussion. Many sites now dominated by pinyon pines or junipers, or both, were historically dominated by sagebrush or other shrubby species because of more frequent fire and lack of grazing, and these are not included in the woodlands discussion. They are, however, included in the discussions of the shrubland landscapes they now occupy.

In many areas where Wyoming big sagebrush, mountain big sagebrush, black sagebrush (*Artemisia nova*), and low sagebrush (*Artemisia arbuscula*) species historically dominated the landscape, expansion by pinyon pine and juniper, and to a lesser extent, other conifers, is occurring (Miller and Tausch 2001). In advanced stages of expansion, dense woodlands completely replace shrubland communities, and these changes are commonly attributed to a lengthening of fire-free intervals associated with 20th-century fire suppression. However, livestock grazing and climatic conditions favorable for tree establishment in the early 20th century also affected vegetation (Miller and Tausch 2001). Burkhardt and Tisdale (1969) found that western juniper (*Juniperus occidentalis*) had more than doubled its distribution between the

1860s, when Euro-American settlement of the West began, and the time of their study about 100 years later. Miller et al. (2008) found that 50 to 75 percent of the sagebrush-steppe communities in portions of Idaho, Oregon, Nevada, and Utah supported expansion of western juniper, Utah juniper (*Juniperus osteosperma*), or singleleaf pinyon by 1920. This rate of expansion has decreased, possibly because of a reduction in the rate of establishment (Miller et al. 2008).

To determine the degree to which pinyon-juniper woodlands have expanded in the Intermountain Region, we compared LANDFIRE biophysical setting (BpS) (LANDFIRE 2008) and existing vegetation type (EVT) (LANDFIRE 2012) data for the acreage dominated by sagebrush shrublands and pinyon-juniper woodlands and shrublands. The BpS layer represents vegetation cover types that may have been present before Euro-American settlement. This layer is based on both the current physical environment (NatureServe's ecological systems classification [Comer et al. 2003]) and an approximation of historical disturbance regimes. The EVT layer is an approximation of existing land cover types that relies on decision tree models, field data, Landsat imagery, elevation, and biophysical gradient data as predictors of vegetation.

Table 7.3 shows the difference in acres of pinyon-juniper woodlands and shrublands and sagebrush shrublands between the BpS and EVT layers for each IAP subregion, estimating change in dominance of sagebrush and pinyon-juniper dominated landscapes in the Intermountain Region since Euro-American settlement. It indicates that pinyon-juniper has increased the most in the Great Basin and Semi Desert subregion. Pinyon-juniper has also increased, but to a lesser degree, in the Plateaus and the Uintas and Wasatch Front subregions.

Vegetation Type Description and Distribution

Persistent pinyon-juniper woodlands are dominated by singleleaf pinyon or twoneedle pinyon, and by western juniper or Utah juniper in various combinations. Persistent juniper woodlands occur throughout the Great Basin and Semi Desert, Plateaus, and Uintas and Wasatch Front subregions,

Table 7.3—LANDFIRE-derived estimates (percent of the landscape) of change in dominance of sagebrush and pinyon-juniper dominated landscapes in the Intermountain Region since Euro-American settlement. The biophysical settings (BpS) layer (LANDFIRE 2008) represents the vegetation cover type that may have dominated the landscape prior to Euro-American settlement. The existing vegetation type (EVT) layer (LANDFIRE 2012) is an approximation of existing land cover types that relies on decision tree models, field data, Landsat imagery, elevation, and biophysical gradient data as predictors of vegetation.

Subregion and cover type	BpS	EVT	Difference (EVT – BpS)
		Percent	
Middle Rockies			
Pinyon-juniper woodlands and shrublands	0.0	0.0	0.0
Sagebrush shrublands	100.0	100.0	
Southern Greater Yellowstone			
Pinyon-juniper woodlands and shrublands	0.0	0.2	+0.2
Sagebrush shrublands	100.0	99.8	
Uintas and Wasatch Front			
Pinyon-juniper woodlands and shrublands	4.5	20.4	+15.9
Sagebrush shrublands	95.5	79.6	
Plateaus			
Pinyon-juniper woodlands and shrublands	48.1	72.2	+24.1
Sagebrush shrublands	51.9	27.8	
Great Basin and Semi Desert			
Pinyon-juniper woodlands and shrublands	0.3	52.7	+52.4
Sagebrush shrublands	99.7	47.3	

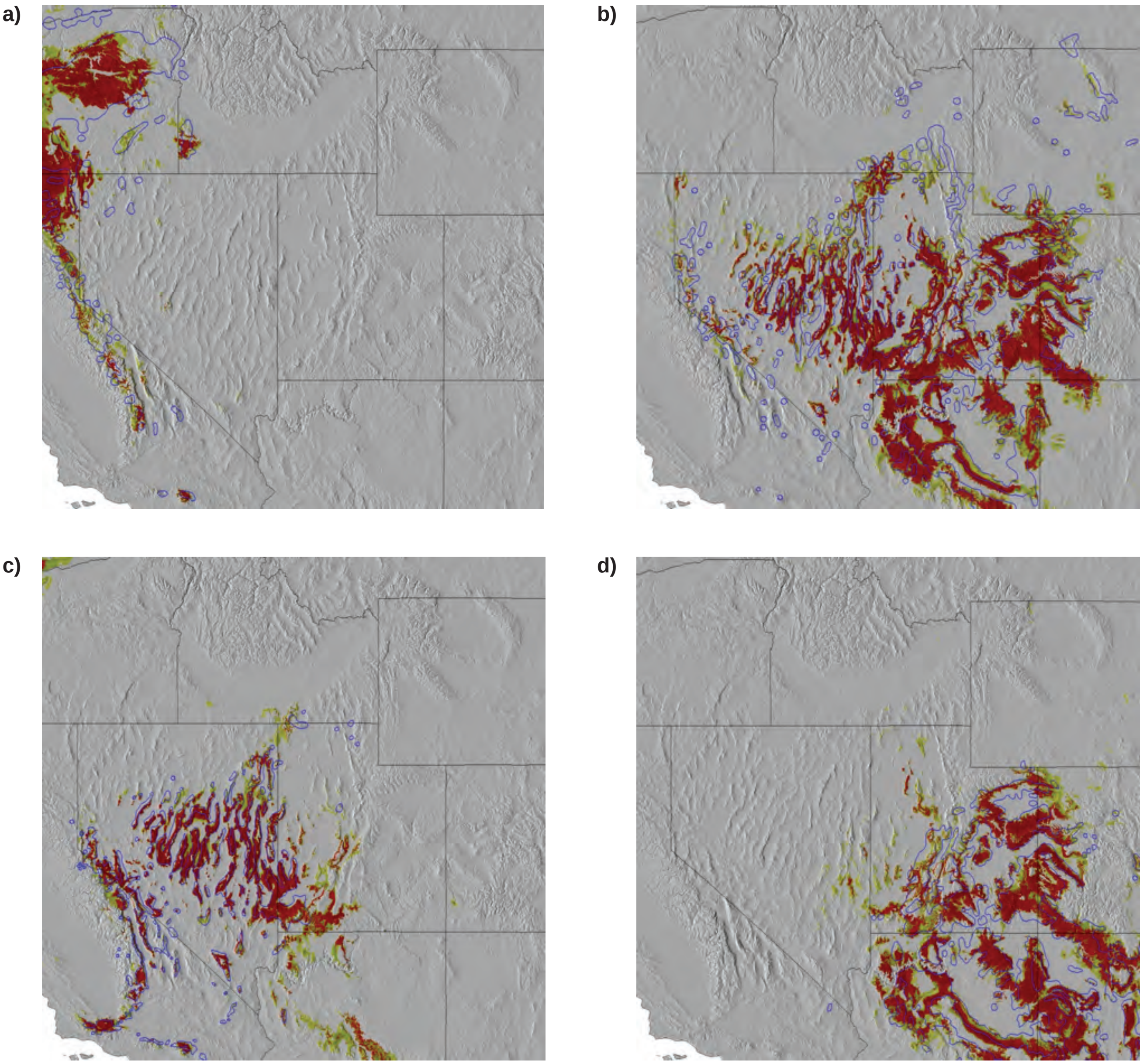

Figure 7.2—Modeled distribution of juniper and pinyon pine species in the Intermountain Adaptation Partnership region: (a) western juniper, (b) Utah juniper, (c) singleleaf pinyon, and (d) twoneedle pinyon. Data are from USDA FS (2017). Model projections to 2090 are based on two global circulation models (HadCM3GGa1, CGCM2_ghga), assuming an increase in greenhouse gas emissions of 1 percent per year since 1990 (see Rehfeldt et al. [2006]).

but are relatively insignificant in the Middle Rockies and Southern Greater Yellowstone subregions (fig. 7.2). In the IAP region, persistent pinyon-juniper woodlands generally make up between 2 and 10 percent of the total woodland areas in any given geographic area (Miller et al. 1999), but they are particularly abundant on portions of the Colorado Plateau at lower elevations (Romme et al. 2009).

Species composition of pinyon-juniper woodland varies across the IAP region. Western juniper occurs along the western edge of the Great Basin and Semi Desert subregion in southwestern Idaho and northwestern Nevada (fig. 7.2a). Utah juniper is the most common tree in the Great Basin and Semi Desert subregion, and is widely distributed throughout the Plateaus, and Uintas and Wasatch Front subregions; it is much less abundant in the Middle Rockies and Southern Greater Yellowstone subregions (Lanner 1983) (fig. 7.2b). Singleleaf pinyon is mostly limited to woodlands in the California, Nevada, and extreme southwestern Utah portions of the Great Basin and Semi Desert subregion (fig. 7.2c). A few disjunct populations occur in other National Forest lands in Utah, notably in the Mollens Hollow Research Natural Area on the Logan District of the Uinta-Wasatch-Cache National Forest. Twoneedle pinyon occurs at lower elevations of National Forests in the Plateaus subregion and in adjacent landscapes of the Uintas and Wasatch Front subregion (fig. 7.2d).

Figure 7.3—Utah juniper. The presence of very old Utah juniper suggests that this rocky site would only rarely develop a grassy understory capable of carrying a surface fire (from Tausch and Hood [2007]).

Pinyon-juniper woodlands often occur on rocky upland sites with shallow and coarse-textured soils that support sparse herbaceous vegetation cover (fig. 7.3). Curl-leaf mountain mahogany may be a codominant or subdominant woodland species. Immediately after disturbance, these sites are dominated by shrubs, grasses, or forbs, or a combination, which may persist for long periods of time.

The age of many pinyon and junipers in persistent stands throughout the West suggests that natural stand-replacement fires of mixed to high severity may be infrequent to rare, averaging 100 to 500 years (Bauer and Weisberg 2009; Miller et al. 1999; Romme et al. 2009). Low-intensity surface fires had a very limited role in affecting stand structure and dynamics in most persistent woodlands historically; most fires were high-severity, stand-replacement fires (Romme et al. 2009). However, fire history is often difficult to measure in these ecosystems because of the lack of fire scar evidence in many pinyon-juniper ecosystems (Baker and Shinneman 2004).

Sensitivity to Climate Change: High

Occupying the transition zone between mesic forests at higher elevations and xeric environments too dry for trees at lower elevations, pinyon-juniper woodlands may be highly sensitive to changes in temperature and precipitation (Romme et al. 2009). Climate envelope model projections for species dominant in pinyon-juniper woodlands indicate a significant contraction of this type in Nevada and Utah (classified as Great Basin conifer forest by Rehfeldt et al. 2012). The likely causes of these projected contractions include higher temperatures and increasing drought stress.

In addition to the direct effects of climate change, pinyon-juniper woodlands may be sensitive to the indirect effects of climate change, including increased area burned (McKenzie et al. 2004) and insect outbreaks (Romme et al. 2009). For example, a major pinyon pine mortality event in 2002–2004 in Colorado, Arizona, New Mexico, and Utah resulted from high temperatures, drought, and bark beetle outbreaks (Romme et al. 2009). Fire directly causes tree mortality, and warm and dry conditions after fires may also inhibit tree regeneration, affecting species composition and long-term vegetation trajectories (Floyd et al. 2015).

As a result of these sensitivities to the direct and indirect effects of warming, persistent pinyon-juniper woodlands are rated as highly sensitive to climate change (table 7.2). However, Utah juniper, which is the most common juniper in the region, has been observed moving downslope into communities currently dominated by sagebrush. At the same time, pinyon pines have been observed moving upslope. It is unclear which novel communities will form in a changing climate, but some new communities that include juniper and possibly pinyon are likely to remain, at least on portions of the landscape.

Adaptive Capacity: Moderate

Persistent pinyon-juniper woodlands are a complex set of ecosystems with a high degree of variation among sites (Romme et al. 2009). Many sites on which this vegetation type occurs have relatively low abundance of invasive species, and because many sites with persistent pinyon-juniper woodlands occur in well-protected, rocky areas with relatively little pressure from livestock grazing and other human uses, their ecological condition has not been degraded. However, with climate change, they may be affected by invasive species from adjacent plant communities, specifically invasive annual grasses such as cheatgrass, particularly at the lower-elevation ecotones (Chambers et al. 2014). Cheatgrass invasion significantly shortens fire return intervals (Chambers et al. 2014) and could cause major ecological change in these woodlands. For this reason, the adaptive capacity of pinyon-juniper woodlands is rated as moderate (table 7.2).

Vulnerability to Climate Change: High

The vulnerability of persistent pinyon-juniper is high because of high sensitivity to the direct and indirect effects of climate change and moderate adaptive capacity (table 7.2). Comer et al. (2012) found that the vulnerability of Great Basin pinyon-juniper woodlands in the Mojave Desert, south of the Great Basin, was moderate, because although these ecosystems were highly sensitive to climate change (they are projected to contract with warming), their adaptive capacity was also high. In our assessment, we gave greater importance to the effects of climate change on adjacent landscapes that would indirectly affect the resilience of the persistent pinyon-juniper woodlands, leading to a vulnerability rating of high.

Oak-Maple Woodlands

Vegetation Type Description and Distribution

Oak-maple woodlands are dominated by mature stands of bigtooth maple or Gambel oak, or both, under natural disturbance regimes. These woodlands are most abundant in the Uintas and Wasatch Front subregion and are also found in the Plateaus subregion (table 7.1, fig. 7.4). Gambel oak is more widespread in the Plateaus subregion, occurring over a greater range of elevations, but generally does not extend north of Brigham City in northern Utah. Bigtooth maple, on the other hand, extends through central Utah into the Southern Greater Yellowstone subregion. Although characteristic of these woodlands, both species (especially Gambel oak) occur as subdominant components of mountain shrubland communities, and boundaries between these vegetation types are sometimes arbitrary.

Historical fire regimes in oak-maple woodlands are not well understood because of a lack of physical evidence such as fire scars (Kaufmann et al. 2016). However, these woodlands are well adapted to fire. Immediately after disturbance, these sites are dominated by shrubs, grasses, or forbs. Both dominant tree species, as well as many of the associated shrubs and herbaceous species, sprout from the root crown following top kill, so postdisturbance grass-forb dominance is short lived.

Sensitivity to Climate Change: Low to Moderate

Climate envelope model projections show a slight restriction of current habitats of Gambel oak and an expansion of its climate envelope into Idaho and Montana by 2060; a few of these models show expansion into eastern Nevada as well (Rehfeldt et al. 2006). During that same time period, some models indicate an expansion of the bigtooth maple climate envelope into eastern Idaho and Montana, although the distribution throughout much of its current range decreases. For these reasons, we have determined that this vegetation type has a low to moderate sensitivity to the effects of climate change (table 7.2).

Adaptive Capacity: Moderate

Gambel oak and bigtooth maple sprout after fire, and can easily reestablish following disturbance (Engel 1983). In addition, there are many species associated with Gambel oak or bigtooth maple communities (Simonin 2000; Tollefson 2006), many of which sprout following fire. This diversity of fire-adapted species provides these communities with significant adaptive capacity. However, adaptive capacity

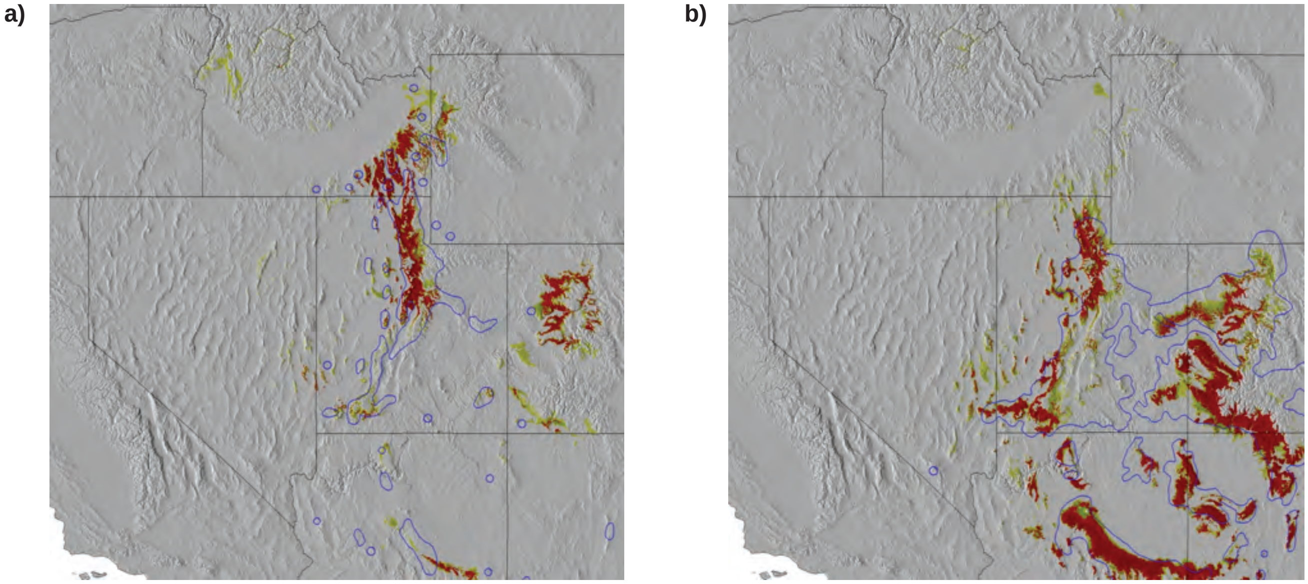

Figure 7.4—Modeled distribution of (a) bigtooth maple and (b) Gambel oak in the Intermountain Adaptation Partnership region. Data are from USDA FS (2017). Model projections to 2090 are based on two global circulation models (HadCM3GGa1, CGCM2_ghga), assuming an increase of greenhouse gas emissions of 1 percent per year since 1990 (see Rehfeldt et al. [2006]).

of these ecosystems is affected to some degree by the number of invasive species capable of invading following disturbance, which can lower species diversity and alter fire regimes. This type is given a moderate adaptive capacity rating because of the potential for invasive species establishment (table 7.2).

Vulnerability to Climate Change: Low to Moderate

The vulnerability of oak-maple woodlands is rated as low to moderate because of the low to moderate sensitivity and moderate adaptive capacity to climate change (table 7.2). Although the current locations of these woodlands may change over time, the amount of land covered by this vegetation type may increase to some degree outside its current distribution.

Mountain Mahogany Woodlands

Vegetation Type Description and Distribution

Curl-leaf mountain mahogany woodlands typically occur throughout the IAP region in isolated patches on warm, rocky ridges and on other sites with dry, coarse-textured soils, primarily on western or southern exposures (Davis and Brotherson 1991). These woodlands are most common in the Great Basin and Semi Desert, Plateaus, and Uintas and Wasatch Front subregions (table 7.1). Historically, this type was restricted to rocky areas that were probably protected from fire, but curl-leaf mountain mahogany, like pinyon-juniper (although to a lesser extent), has expanded because of fire exclusion in habitats where frequent fires historically kept it in check (Davis and Brotherson 1991). Most often, curl-leaf mountain mahogany is killed by fire, and regeneration is only by seed (Gruell et al. 1985). However, early postfire recolonization by mountain mahogany is facilitated by seeds that are well adapted for wind dispersal. Consequently, this species is often among the first nonsprouting shrubs and trees to reoccupy a burn when unburned plants provide a seed source nearby. Curl-leaf mountain mahogany can survive for long periods (Riegel et al. 2006), and on sites that have sustained long fire-free periods, mahogany trees have been found to be over 400 years old (Dealy 1975).

Sensitivity to Climate Change: Moderate

Climate envelope models project a significant restriction in the climate envelope of curl-leaf mountain mahogany (Rehfeldt et al. 2006). The species occurs across a broad elevational gradient (7,000–11,000 feet), which suggests some resilience to climate change. Curl-leaf mountain mahogany is often found on soils that have low fertility (Gucker 2006) or contain calcium carbonates (although this trait does not seem to be as evident in the Great Basin as elsewhere in its distribution) (Gonella and Neel 1995). These traits, combined with relatively high predation of its seeds (Dealy 1979), suggest the species may not be a good competitor. However, the affinity of the species for poor soils suggests it is tolerant of poor conditions, and it could potentially expand into areas where other species become less competitive in warmer and drier conditions. For these reasons, and because of the similarity of some mountain mahogany woodland sites to those of persistent pinyon-juniper woodland sites, the sensitivity of these woodlands is rated as moderate (table 7.2).

Adaptive Capacity: Low to Moderate

The lack of fire since the early 1900s has allowed curl-leaf mountain mahogany to expand to some degree and occupy new habitats. Livestock grazing has also been largely absent from these woodland communities because of the difficult terrain and sparse forage (USDA FS 2013). However, abundance of invasive species has increased in some of these communities, potentially affecting fire return intervals and resilience. In addition, this vegetation type generally does not have a high level of site diversity. For these reasons, and because the species grows slowly and does not sprout following fire, the adaptive capacity of mountain mahogany woodlands is rated as low to moderate (table 7.2).

Vulnerability to Climate Change: Moderate to High

The vulnerability of mountain mahogany woodlands to climate change is rated moderate to high because of the moderate sensitivity rating and the low to moderate adaptive capacity rating (table 7.2). These communities are limited to specific sites and have few places where they can expand.

Shrubland Ecosystems

Shrubland ecosystems are vegetation communities with at least 10 percent cover of shrub species that are generally less than 6.5 feet tall at maturity, and often less than 1.5 feet tall on relatively harsh sites. Shrubland ecosystems include those dominated by dwarf and big sagebrush or a variety of upland shrub species, as well as all salt desert communities. The term shrub-steppe is often applied to shrubland ecosystems when herbaceous understory vegetation (generally perennial grasses and forbs) is sufficiently abundant to codominant. Mountain big sagebrush shrubland (shrub-steppe) is the most common shrubland type in the IAP region. Mountain big sagebrush and dry big sagebrush shrublands make up nearly 40 percent of the nonforest vegetation in the IAP region (table 7.1). As already noted (table 7.3), much of the area historically dominated by these shrublands, as well as other sagebrush-dominated shrublands, has been invaded by pinyon pine or juniper. Mountain shrublands, though present throughout the region, are more extensive in the northern subregions (Middle Rockies and Southern Greater Yellowstone). Dry big sagebrush shrublands are most abundant on National Forest lands in the Great Basin and Semi Desert subregion. Dwarf sagebrush shrublands also occur throughout the region, but are most abundant in the Plateaus subregion. Blackbrush and salt desert shrublands occupy only minor portions of National Forest lands in the region, but are more widespread on adjacent landscapes at lower elevations in the Plateaus and Great Basin and Semi Desert subregions.

Various species and varieties of sagebrush have been combined into four unique sagebrush types because of similarities in environments they inhabit, plant structure, or response to disturbances. These sagebrush types are:

- **Mountain big sagebrush shrublands**—Mountain big sagebrush and Bonneville big sagebrush (*Artemisia tridentata* ssp. *vaseyana* × *wyomingensis*).
- **Dry big sagebrush shrublands**—Wyoming big sagebrush and basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*).
- **Sprouting sagebrush shrublands**—Mountain silver sagebrush (*A. cana* ssp. *viscidula*), snowfield sagebrush (*A. spiciformis*), and threetip sagebrush (*A. tripartita*). Timberline sagebrush (*A. rothrockii*) is endemic to the Sierra Nevada in California and has very limited distribution within the region. Thus, it is not addressed in any detail in this report.
- **Dwarf sagebrush shrublands**—Low sagebrush, black sagebrush, scabland sagebrush (*A. rigida*), Bigelow sagebrush (*A. bigelovii*), Owyhee sagebrush (*A. papposa*), budsage (*A. spinescens*), and pygmy sagebrush (*A. pygmaea*).

Although Wyoming and various low-growing sagebrush species are common throughout the region, mountain big sagebrush communities occupy the greatest area of USFS lands (table 7.1). Wyoming and basin big sagebrush types were aggregated because of similarities in life history traits, and because they represent critical habitats for many species of birds and wild and domestic ungulates. However, basin big sagebrush occupies sites with deeper soils (often on alluvial fans). These conditions tend to increase available moisture, with higher coverage by perennial bunchgrasses, suggesting these sites may be more resilient and resistant to various threats (Chambers et al. 2013). Similarly, the low or dwarf sagebrush species were combined for their common physical structure, unique habitats they represent, and similar life histories. Finally, all sprouting sagebrush species were combined because of their similar response to fire.

Overall, about 10 percent of the sagebrush-steppe of the Southwest and Intermountain West has been converted to dryland or irrigated agriculture (Noss et al. 1995). Over 99 percent of the remaining sagebrush-steppe has been affected by livestock, and about 30 percent of that area has been heavily grazed. In addition, much of the sagebrush in the IAP region contains annual invasive species, with impacts concentrated in lower-elevation, more xeric sagebrush landscapes (Miller et al. 2011). Various nonnative perennial species, such as crested wheatgrass (*Agropyron cristatum* and *A. desertorum*), intermediate wheatgrass (*Thinopyrum intermedium*), smooth brome (*Bromus inermis*), and forage kochia (*Bassia prostrata*), have been seeded for forage, fire breaks, or erosion control throughout the region. These species now dominate large areas, and continue to be seeded during postfire rehabilitation across much of the region despite some concerns for impacts on native species diversity (Davies et al. 2013; Lesica and Deluca 1996).

Effects of Climate Change

Several recent studies modeled the effects of climate change on sagebrush shrublands in the western United States (Balzotti et al. 2016; Bradley 2010; Schlaepfer et al. 2012; Still and Richardson 2015). Each of these studies concluded that climate change is likely to have significant effects on sagebrush ecosystems. Climate change risk to sagebrush is most pronounced in southern Nevada because decreased summer precipitation and increased temperatures there could make current sagebrush habitat climatically unsuitable in the future (Bradley 2010; Schlaepfer et al. 2012). Still and Richardson (2015) projected a 39 percent loss of the climate suitable for Wyoming big sagebrush across its range in the West. Neilson (2005) projected a loss of 12 percent of all current sagebrush habitat with each 1.8-°F increase in temperature, and the southern limit of suitable climate for many sagebrush species may shift to the northern Great Basin.

Sagebrush species, however, commonly hybridize, which has been shown to provide greater ability to adapt to changing environments (Hoffman and Sgrò 2011; McArthur 2000). In addition, all major species of sagebrush included in this discussion have both diploid and polyploid populations (McArthur 2000). Polyploids are smaller with slower growth rates that make them better adapted to stressful environments than their diploid relatives (Sanderson et al. 1989). Sagebrush, as a complex, may have greater ability to adapt to climate change than other associated taxa. It is impossible, however, to understand what effect new genotypes could have on future generations of sagebrush, how quickly they may develop, and how emergence of new genotypes may affect their ability to occupy sites that are becoming increasingly warmer and drier.

Mountain Big Sagebrush Shrublands

Vegetation Type Description and Distribution

Mountain big sagebrush plant communities occur throughout the IAP region and are generally found at elevations between 4,500 and 10,200 feet on moderately deep to deep, well-drained soils and on sites that are more cool and mesic than those associated with Wyoming big sagebrush plant communities (Blaisdell et al. 1982; Tueller and Eckert 1987; West et al. 1978; Winward 1980; Winward and Tisdale 1977). Mountain big sagebrush communities are dominated by mountain big sagebrush or xeric big sagebrush for long periods of time under historical disturbance regimes. Basin big sagebrush, snowfield sagebrush, mountain silver sagebrush, or threetip sagebrush may also occur as minor components in this vegetation type. Other shrub taxa, such as yellow rabbitbrush (*Chrysothamnus viscidiflorus*), rubber rabbitbrush (*Ericameria nauseosa*), mountain snowberry (*Symphoricapos oreophilus*), Woods' rose (*Rosa woodsii*), Saskatoon serviceberry (*Amalenchier alnifolia*), and chokecherry (*Prunus virginiana*) may be present at subdominant levels. Except where degraded by chronic overgrazing, the herbaceous understory of mountain big sagebrush communities supports a rich variety of perennial grasses and forbs that are critical for system resilience and wildlife habitat, and codominates even in late-seral communities. Natural fire regimes maintain spatial and temporal mosaics of herb-dominated to shrub-dominated patches in various stages of succession, and prevent conversion to conifer forests or woodlands.

Historically the presettlement fire return intervals for mountain big sagebrush were relatively short, (35–80 years) (Heyerdahl et al. 2006; Kitchen and McArthur 2007; Miller et al. 2001) compared to Wyoming big sagebrush (more than 100 years) (Baker 2006; Lesica et al. 2007). On cooler, more mesic sites, fire-free intervals have increased to between 50 and 150 years. The lack of fire, combined with the effects of livestock herbivory, has caused much of the increase in pinyon pines and juniper, with a consequent loss of herbaceous and shrub cover throughout much of the range of mountain big sagebrush (Crawford et al. 2004; Miller and Rose 1999).

Sensitivity to Climate Change: Moderate

Mountain big sagebrush shrublands have moderate sensitivity to climate change. Sensitivity to climate change varies with elevation, with lower-elevation stands more sensitive than those at higher elevations (Balzotti et al. 2016). Mountain big sagebrush growth is dependent on temperature, precipitation, and maximum snow depth (Poore et al. 2009). Winter precipitation has the strongest relationship with growth (Poore et al. 2009). At higher elevations growth is positively correlated with temperature, but lower elevations may experience decreased growth with warming. Likewise, production in higher-elevation vegetation communities may increase in the future (Reeves et al. 2014). Given that increased temperatures and a reduction in snowpack are likely with climate change, mountain big sagebrush growth rates are likely to decrease at lower elevations but could be improved at temperature-limited sites toward the higher end of the species distribution, thus potentially having significant and asymmetric effects on sagebrush cover.

Adaptive Capacity: Low to Moderate

Adaptive capacity of mountain big sagebrush shrublands is rated as low to moderate (table 7.2), depending on elevation and site conditions, land use history, fire suppression, and abundance of invasive species. A few areas of mountain big sagebrush shrublands have been converted to agricultural lands, and most of those that remain are used for domestic livestock grazing because of the palatable herbaceous undergrowth. Those that have had chronic improper grazing typically have high sagebrush canopy cover and low vigor of native herbaceous species, and thus may have invasive plant species present in varying amounts. In intact mountain big sagebrush shrublands, species and functional type diversity may confer resilience to climate change.

Mountain big sagebrush is easily killed by fire and does not resprout; postfire recovery is from seed that survives fire or disperses from unburned areas. Although recovery for mountain big sagebrush is often rapid (15–35 years) (Kitchen and McArthur 2007; Nelson et al. 2014), longer recovery times (50–150 years) are expected if residual seed are absent or if seedlings fail to establish from the short-lived seed bank (Baker 2006, 2011; Nelson et al. 2014). Postfire recovery is currently problematic on warmer and drier sites and may become a problem on cooler and moister sites in the future if the frequency and intensity of fires increase as projected (Abatzoglou and Kolden 2013). Regeneration of big sagebrush postfire is strongly linked to winter and spring precipitation (Nelson et al. 2014), which is not projected to change significantly in the IAP region (Chapter 3).

With increased fire severity and frequency, there is likely to be a shift in community composition to dominance by fire-adapted shrub and herbaceous species and possibly nonnative species. Fire-adapted shrub species (e.g., rubber rabbitbrush, yellow rabbitbrush, mountain snowberry, Wood's rose, Saskatoon serviceberry, chokecherry) may increase in abundance following fire (Fischer and Clayton 1983; Smith and Fischer 1997). In addition, more spring and winter precipitation and increased minimum temperatures may facilitate the establishment of nonnative annual grasses (particularly cheatgrass, which germinates in winter to early spring) or other invasive species, although this pattern is seldom observed in the cooler, moister mountain big sagebrush communities with healthy herbaceous understories.

Overall, mountain big sagebrush shows higher adaptive capacity than Wyoming big sagebrush, but is likely to be stressed somewhat by drought as climate patterns change (Balzotti et al. 2016). Compositional shifts in herbaceous species are likely. Mountain big sagebrush may be able to persist on mesic sites (Chambers et al. 2013), but mountain big sagebrush communities may be subject to upslope pressure from woodland tree species (unless disturbance or disturbance surrogates are used to reset successional processes). Conifer expansion, especially by juniper and pine species, into sagebrush communities is especially pronounced in the Great Basin (Miller et al. 2008). Interruptions to wildfire cycles and favorable climatic periods, combined with other factors, have led to the proliferation of trees, often occurring in sagebrush sites that previously did not support trees. Consequently, land cover type has gradually shifted from shrubland to woodland across numerous sites (Miller et al. 2011). These transitions significantly reduce resilience to changing climates, as the increased abundance of trees negatively affects soil moisture available for perennial herbaceous species. Conversely, mountain big sagebrush could expand into drier persistent aspen stands, as these areas are likely to be negatively affected by climate change (Chapter 6). This species is well adapted to the soils on which these aspen stands occur, and this replacement is already occurring in some areas.

Subspecies of big sagebrush can hybridize or undergo polyploidization, offering greater genetic diversity and potentially providing the species with the capacity to undergo selection and adapt to shifting climatic regimes (Poore et al. 2009). Garrison et al. (2013) found that what has been called Bonneville big sagebrush (Garrison 2006; McArthur and Sanderson 1999; Rivera et al. 2011; Winward 2004) has been shown to be a hybrid between mountain big sagebrush and Wyoming big sagebrush. This hybrid is found in southeastern Idaho and extreme northern Utah. Hybridization creates a greater level of uncertainty regarding the future distribution of this subspecies, as well as all other species and varieties of sagebrush. The ability of the expected new hybrids to survive on changing habitats under future climates is poorly understood at this time. Bonneville big sagebrush has also been observed in southern Utah, where it occupies sites ecologically similar to Wyoming big sagebrush. Hybridization contributes to the evolution of sagebrush (McArthur and Sanderson 1999), which may at least maintain morphologically similar sagebrush communities in the future.

Vulnerability to Climate Change: Moderate to High

Vulnerability of mountain big sagebrush shrublands varies from moderate to high because of the broad elevational range at which mountain big sagebrush occurs, and because of the wide range in current conditions of these shrubland communities (table 7.2). Factors contributing to the vulnerability of these communities include livestock grazing, expansion of pinyon-juniper shrublands, altered wildfire regimes, and nonnative invasive species, including cheatgrass and seeded forage species.

Dry Big Sagebrush Shrublands

Vegetation Type Description and Distribution

Dry big sagebrush shrublands are those dominated by Wyoming big sagebrush, basin big sagebrush, Parish big sagebrush (*A. t.* ssp. *parishii*), or sand sagebrush (*A. filifolia*) for long periods of time under historical disturbance regimes. Small amounts of threetip sagebrush may also occur in this vegetation type. The perennial herbaceous understory is less productive and less diverse in this vegetation type than in mountain big sagebrush-steppe and may be codominant or subdominant in intact communities.

Wyoming big sagebrush occurs throughout the IAP region in locations where winter or spring precipitation is sufficiently reliable to support spring growth; it is often found in areas receiving 8 to 16 inches of precipitation annually (Welsh et al. 2008). It typically grows in the warm, dry conditions of valleys and foothills, generally below 6,500 feet elevation (Welsh et al. 2008; Winward and Tisdale 1977), and often below National Forest boundaries. Soils on which Wyoming big sagebrush occurs are often underlain by an argillic, caliche, or silica layer (Miller et al. 2011). Basin big sagebrush also occurs throughout the region, but most of its habitat has been converted to agricultural use and other development because it typically occurs in valley bottoms on highly productive soils. Surviving stands are common in the deep soils of canyon bottoms and other areas of soil aggradation. Sand sagebrush is limited in the IAP region to southern Utah and possibly the Spring Mountains of southern Nevada. Parish big sagebrush is found on the Bridgeport District of the Humboldt-Toiyabe National Forest (central Nevada). Small populations also occur in southern Utah (Shultz 2006).

Sensitivity to Climate Change: High

Climate change is projected to have significant effects on dry big sagebrush shrublands, and these ecosystems are highly sensitive to a changing climate (table 7.2). Projections suggest potential loss of more than one-third of the climatically suitable area of Wyoming sagebrush by 2050 (Still and Richardson 2015). Amount and timing of precipitation control seeding establishment at low elevations because soil water content primarily controls seedling survival (Nelson et al. 2014; Poore et al. 2009; Schlaepfer et al. 2014). Conditions suitable for seedling establishment are infrequent under contemporary climatic conditions and are likely to become less frequent with climate change. Thus, these ecosystems remain vulnerable to drought, and sagebrush establishment is likely to be more difficult as years with adequate snowfall become less frequent (Meyer and Warren 2015). Even after seedling establishment, drought and increased summer temperature can affect survival and growth of adult plants because growth is positively correlated with winter precipitation and winter snow depth (Poore et al. 2009). Hence, if drought events increase in frequency and severity in the future, big sagebrush biomass and the abundance and diversity of perennial grasses and forbs may decrease.

Adaptive Capacity: Low to Moderate

Adaptive capacity of dry big sagebrush ecosystems to climate change is low (table 7.2) because of the effects of historical grazing on the composition and structure of these warmer and drier sites, fragmentation with conversion to agricultural uses (Noss et al. 1995), and oil and gas development, which is prominent in the IAP in the Uinta Basin of eastern Utah. Prior to Euro-American settlement in the West, much of the land occupied by Wyoming big sagebrush shrubland had understories dominated by spatially discontinuous perennial grasses. These communities carried fires only when humidity was low and winds were high, or after several wet years when fine fuels could accumulate (Hull and Hull 1974; Mensing et al. 2006; Vale 1975). These fire-free intervals were relatively long in comparison to other more mesic sagebrush-dominated sites, often 100 to 200 years or more. Where perennial bunchgrasses and shrubs have been lost to improper livestock grazing and invasion by annual grasses, fire frequency has increased dramatically, to more than double that of sagebrush shrublands with intact, native understory in the Great Basin (Balch et al. 2013). Observations of increased fire frequencies were reported as early as the early to mid–1900s after these annual grasses had invaded much of the Intermountain West (Pickford 1932; Piemeisel 1951; Robertson and Kennedy1954).

Strong negative relationships exist between cover of cheatgrass and perennial native grasses and forbs in Wyoming big sagebrush shrublands (Anderson and Inouye 2001; Chambers et al. 2014; West and Yorks 2002). Chambers et al. (2007) found that on relatively intact sites, native perennial herbaceous vegetation resprouted after fire, which then limited the growth and reproduction of cheatgrass. About 15 percent cover of perennial native herbaceous species is required to prevent an increase of medusahead or cheatgrass following fire or management treatments in these shrublands (Chambers et al. 2014; Davies et al. 2008).

The genetic variability within these species of sagebrush, how that variability is spread across the distribution of the species, and the relationship of this variability to climate change effects on the species are of critical importance (Chaney et al. 2017). Cytotypic variation, or individuals within a species that have different chromosomal factors (e.g., diploid versus tetraploid) than others within the same species, may be as important as subspecific variation in explaining adaptation and functional diversity within the big sagebrush complex (Brabec et al. 2016).

Research has also shown that minimum temperatures play a bigger role in the probability of sagebrush survival than water-related responses (Brabec et al. 2016; Chaney et al. 2017). In common garden studies, Chaney et al. (2017) found greater survival from cytotypes collected from regions with greater seasonal differences in temperature and higher summer precipitation (interior regions of the continent) than those collected from regions with moderate winter temperatures and drier summers. They also found that Wyoming big sagebrush had a greater physiological avoidance and resistance to freezing than mountain big sagebrush. These differences may have been the result of a greater insulating effect of snow cover at higher elevations where mountain big sagebrush occurs, and the resulting differences in the need to adapt to cold temperatures by the more exposed Wyoming big sagebrush. The importance of integrating genetic diversity into our understanding of the adaptive capacity of all sagebrush species is becoming more evident as the research in this area begins to evaluate how these cover types will respond to climate change.

Vulnerability to Climate Change: Very High

Dry big sagebrush shrublands have a very high vulnerability to climate change because of high sensitivity and low adaptive capacity (table 7.2). Evidence of this is found in the loss of this type across large areas of southern Utah in response to the 2002–2003 drought. Dry big sagebrush shrublands occupying lower elevations of the Great Basin are expected to be some of the most vulnerable to climate change. Western Wyoming, eastern Idaho, and higher elevations in the Great Basin are predicted to retain or gain climatically suitable areas for the most abundant component of dry big sagebrush shrublands, Wyoming big sagebrush (Still and Richardson 2015). Although suitable Wyoming big sagebrush habitat is projected to expand in some areas within and beyond the IAP region, its overall distribution is projected to decrease by at least 39 percent (Still and Richardson 2015). The distances between current and projected future habitats capable of supporting Wyoming big sagebrush often exceed the estimated migration rate of 6 to 19 miles per century (McLachlan et al. 2005; Yansa 2006). Thus, this species may lose significantly more habitat to climate change than it can gain (Still and Richardson 2015) without active assistance.

Sprouting Sagebrush Shrublands

Vegetation Type Description and Distribution

Sprouting sagebrush shrublands include communities dominated by mountain silver sagebrush, snowfield sagebrush, threetip sagebrush, or timberline sagebrush. These species are all capable of sprouting from the root crown following fire or other form of top kill, and because of their ability to sprout, a postdisturbance stage dominated by grasses and forbs is short lived.

Mountain silver sagebrush occurs through most of the IAP region (fig. 7.5), commonly on heavy soils in riparian terraces and in areas with high snowpack in mountainous areas (McArthur 2000). In the Sierra Nevada, similar sites are occupied by Bolander silver sagebrush (*A. cana* ssp. *bolanderi*) (Shultz 2006). In some areas around the Greater Yellowstone Area, silver sagebrush has replaced lodgepole pine (Jakubos and Romme 1993).

There is some disagreement on the distribution of threetip sagebrush in the literature. Shultz (2006) describes this variety as occurring in portions of Idaho, Nevada, and Wyoming, whereas Winward (2004) includes northern Utah in the distribution of this variety. Much of its habitat has been converted to agriculture because of the productive soils on which it occurs. Remaining populations are isolated throughout its presettlement distribution (Shultz 2006).

Timberline sagebrush is a California endemic and is uncommon in the IAP region (fig. 7.5). It occurs in deep soils along forest margins of the Sierra Nevada in California and Nevada (McArthur 2000), and collections on or near the Bridgeport District in Humboldt-Toiyabe National Forest appear to generally be above 10,000 feet elevation (Jepson Flora Project 2016). Because it is rare in the region, we did not include it in this assessment.

Snowfield sagebrush occurs at high elevations in the IAP region throughout northern and central Utah, western Wyoming, central and southeastern Idaho, and the eastern Sierras. It typically occurs at higher elevations than, or as inclusions within, mountain big sagebrush shrublands in areas where snow depth and subsequent soil moisture are higher. However, it is included here because of its ability to sprout in response to fire.

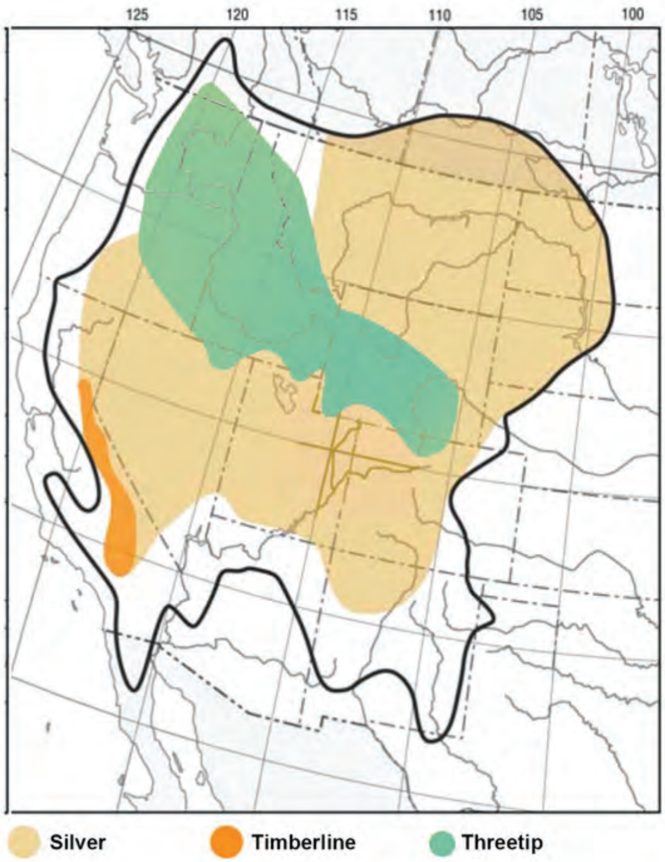

Figure 7.5—Distribution of silver sagebrush, timberline sagebrush, and threetip sagebrush in the western United States (from Mahalovich and McArthur [2004]).

Sensitivity to Climate Change: Moderate

Warmer and drier climates will negatively affect the vigor and abundance of sprouting sagebrush species, which are adapted to more mesic conditions. Although these species can sprout following disturbance, they also reproduce by seed. Like other sagebrush species, however, seed viability is short for many species, including silver sagebrush (Romo and Young 2002). Seed viability is likely to be negatively affected by increased temperatures, prolonged droughts, and irregular precipitation patterns. For these reasons, the sensitivity of sprouting sagebrush shrublands is rated as moderate (table 7.2).

Adaptive Capacity: Moderate

All three subspecies of sagebrush in these communities sprout after fire. In addition, silver sagebrush spreads by underground rhizomes (Schultz and McAdoo 2002) and therefore can recover more quickly than other species of sagebrush following disturbance. These factors, when combined with the more mesic habitat conditions, led to a rating of moderate adaptive capacity (Balch et al. 2013) (table 7.2).

Vulnerability to Climate Change: Moderate

Sprouting sagebrush shrublands have been given a moderate climate change vulnerability rating because of their moderate sensitivity and adaptive capacity to climate change (Balch et al. 2013) (table 7.2). Although the sagebrush species in this type can sprout, their higher dependence on soil moisture than other sagebrush shrublands makes them vulnerable to increasing temperatures and drought. In addition, increased fire frequency and severity (particularly in threetip sagebrush communities) may cause a shift in community composition to dominance by fire-adapted herbaceous or nonnative species. Other fire-adapted shrub species (e.g., rubber and yellow rabbitbrush) may increase, particularly following fire. Nonnative invasive species respond favorably after fire, and if present, may increase in cover and density. Understory composition in both silver and threetip sagebrush communities may possibly shift to more xeric grassland species (e.g., bluebunch wheatgrass [*Pseudoroegneria spicata*], needle-and-thread [*Hesperostipa comata*]), which are better adapted to warmer and drier conditions. Sprouting sagebrush species may shift landscape position to sites with more moisture and cooler temperatures (e.g., higher-elevation, lower landscape position, and northeast aspects).

Dwarf Sagebrush Shrublands

Vegetation Type Description and Distribution

Dwarf sagebrush shrublands are those communities dominated by low sagebrush (including the subspecies low sagebrush [*A. a.* ssp. *arbuscula*], alkali sagebrush [*A. a.* ssp. *longiloba*], cleftleaf low sagebrush [*A. a.* ssp. *thermopola*], and Lahontan sagebrush [*A. a.* ssp. *longicaulis*]), black sagebrush, Bigelow sagebrush, Owyhee sagebrush, scabland sagebrush, or pygmy sagebrush. These dwarf sagebrush shrublands occur across a broad elevational range, often on sites with shallow or rocky soils, or on soils with high clay content. The abundance and diversity of perennial grasses and forbs vary but are generally similar to or less than those associated with dry big sagebrush shrubland communities. Fires were rare historically because fine fuels are typically low, but when fires occur, the grass-forb stage can persist for long periods of time on harsh sites or where erosion occurs after fire (Young 1983). Pinyon and juniper may invade on the more mesic sites in the absence of disturbance. Some sites are susceptible to invasion by introduced annual grasses, and where this occurs, fire frequency often increases.

Dwarf sagebrush species occur throughout the IAP region (fig. 7.6). Black sagebrush and one or more varieties of low sagebrush are found throughout most of the region. Bigelow sagebrush occurs in the southern portions of Utah and Nevada. Scabland and Owyhee sagebrush are limited to the western and southern portions of Idaho, and northeastern Nevada. Lahontan sagebrush is generally restricted to northwestern Nevada and adjacent areas in California and Oregon. Pygmy sagebrush is uncommon but is locally abundant in east-central and eastern Nevada, western Utah, and the Uinta Basin of northeastern Utah (Ulev 2005).

Low, Bigelow, and black sagebrush occur across a broad geographic and elevational range. Black sagebrush generally occurs between 4,600 and 8,500 feet elevation in the

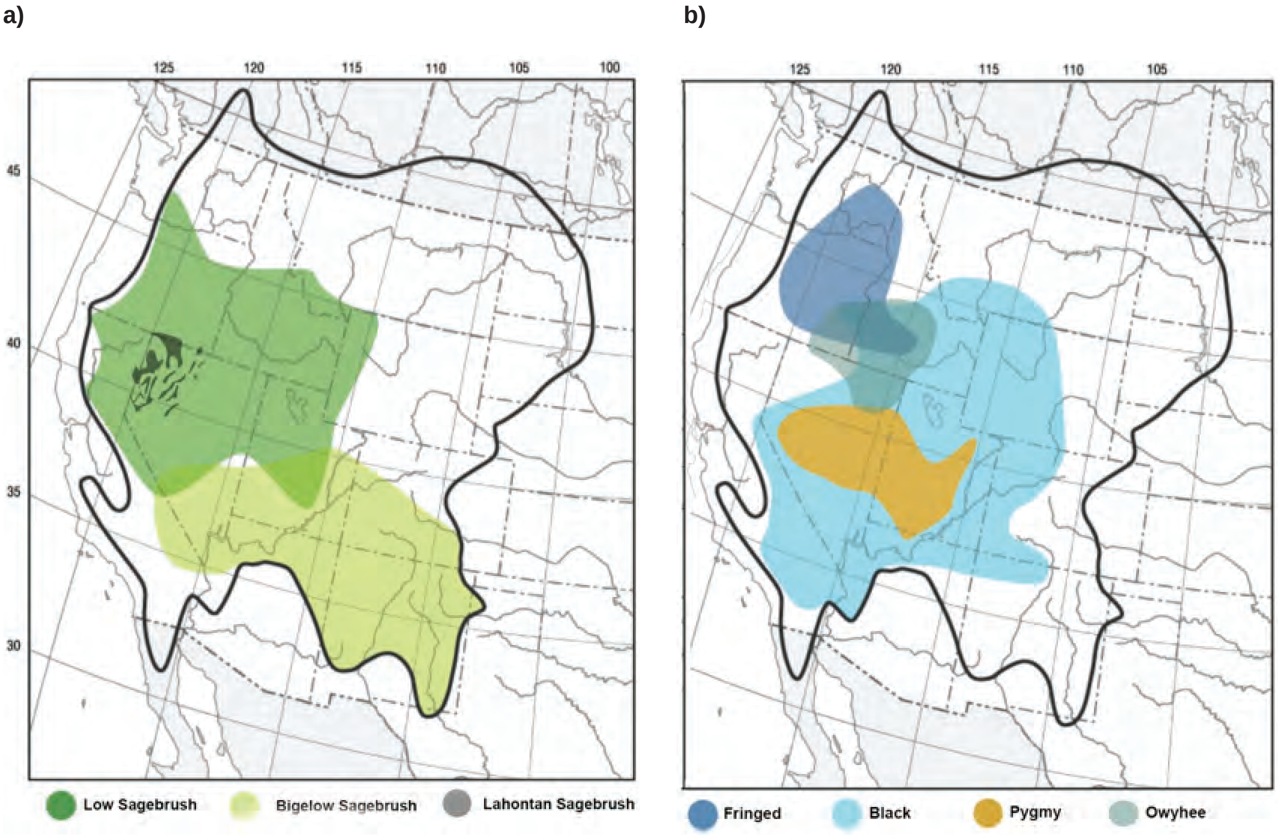

Figure 7.6—Distribution of low sagebrush species in the western United States: (a) low, Bigelow, and Lahontan sagebrush; and (b) fringed, black, pygmy, and Owyhee sagebrush (from Mahalovich and McArthur [2004]).

Intermountain West, and up to 11,000 feet in Nevada (Fryer 2009). Low sagebrush ranges from 2,300 feet to over 11,000 feet in Nevada (Steinberg 2002), but tends to occur primarily above 8,000 feet in a band across central Nevada, from Ely to Bridgeport. Bigelow sagebrush ranges in elevation between 3,000 and 7,000 feet in Nevada and Utah (Howard 2003). Scabland and pygmy sagebrush have a narrower distribution, but elevational range for pygmy sagebrush is 5,000 to 11,000 feet in Nevada (Ulev 2005). Pygmy, Bigelow, scabland, and alkali sagebrush grow in edaphically limited habitats, and all other dwarf sagebrush species generally occur on shallow or rocky soils, making them more resistant to cheatgrass dominance, and therefore more resistant to the large or severe fires to which other sagebrush shrublands have been subjected.

Sensitivity to Climate Change: Moderate to High

All low-growing sagebrush species are likely to be negatively affected by higher temperatures and increased periods of drought. As with all sagebrush species, seed viability of dwarf sagebrush species is short and their dependence on spring soil moisture will make them susceptible to prolonged droughts and to changes in climate that may otherwise affect the timing and amount of spring moisture. Increases in fire, coupled with drought, could inhibit regeneration of the dwarf sagebrush species, particularly on harsh sites (Young 1983).

Adaptive Capacity: Moderate to High

Dwarf sagebrush shrublands are likely to have a moderate to relatively high adaptive capacity to climate change (table 7.2). Species in these shrublands have broad distributions and occur over a wide elevational range in the IAP region. Adaptive capacity may be moderated, however, because of the relatively low productivity characterizing these species, especially where other risk factors (e.g., nonnative annual grasses) are present. Sites dominated by scabland sagebrush occur over a narrower range in elevation (McWilliams 2003), and thus it may be more susceptible to the effects of climate change because alternative suitable sites may not be available.

None of the dwarf sagebrush species can sprout following fire, with the possible exception of hybrids between black and silver sagebrush; sprouting is thought to be a heritable trait in crosses between nonsprouting and sprouting sagebrush species (McArthur 1994). Hybridization may play a role in increasing adaptive capacity of other dwarf sagebrush species to the effects of climate change. In central Nevada, black sagebrush commonly forms hybrids at all elevations. Lahontan sagebrush is a putative stable hybrid between low sagebrush and Wyoming big sagebrush (McArthur and Sanderson 1999).

Vulnerability to Climate Change: Moderate

Dwarf sagebrush shrublands are rated as moderately vulnerable because sensitivity and adaptive capacity are rated as moderate to high (table 7.2). Climate change is likely to result in shifts in the distribution of conditions suitable to support the dwarf sagebrush species in the region. All dwarf sagebrush species are intolerant of fire, and most do not sprout following fire. Because of the low productivity of these sites, however, cheatgrass may not be able to establish on harsh sites (Chambers et al. 2013). Thus, these ecosystems may be more likely to resist a significant change in fire regimes. However, these sites will be exposed to higher temperatures and more erratic precipitation patterns, reducing the ability of seedlings to establish during unfavorable years.

Mountain Shrublands

Vegetation Type Description and Distribution

Mountain shrublands are typically associated with mountain big sagebrush shrublands, oak-maple woodlands, and pinyon-juniper woodlands, as well as montane and subalpine forests. They can occur as large patches within wooded and forested landscapes. Combinations of species such as chokecherry, serviceberry (*Amelanchier alnifoli*a), snowberry, currant (*Ribes* spp.), rose (*Rosa* spp.), maple (*Acer* spp.), sumac (*Rhus* spp.), ceanothus (*Ceanothus* spp.), Scouler willow (*Salix scouleriana*), elderberry (*Sambucus* spp.), thimbleberry (*Rubus parviflorus*), alderleaf mountain mahogany (*Cercocarpus montanus*), bitterbrush (*Purshia tridentata*), wild crab apple (*Peraphyllum ramosissimum*), and mountain ash (*Sorbus scopulina*) are common. Mountain big sagebrush is also common as a subdominant element. Species dominating the overstory of these shrublands are typically adapted to a wide range of elevations (table 7.4). In addition to the wide variety of shrub species, there is an even greater diversity of associated perennial herbaceous species that occur in the understory.

Sensitivity to Climate Change: Low to Moderate

High species diversity, coupled with the broad elevational range over which these communities occur, is likely to result in relatively low sensitivity of mountain shrublands to climate change (table 7.2). Though not directly related to the IAP region, studies in Alberta, Canada, found little change in the spring flowering response of either serviceberry or chokecherry between 1936 and 2006 (Beaubien and Hamann 2011). However, declining snowpacks, more frequent and severe droughts, and warmer temperatures may cause hotter fires and, at the same time, sites may become drier, causing variable amounts of mortality, depending on site conditions.

Adaptive Capacity: Moderate to High

Montane shrublands were historically maintained by relatively frequent fire (approximately every 30 years or less) (Smith and Fischer 1997), and most montane shrubs sprout following fires. Stressors to these shrublands include fire exclusion and resulting conifer encroachment, browsing by both native wildlife and domestic livestock, and insects and disease. As noted earlier, the diversity of species in these communities is often very high. However, there is the potential that more frequent and severe fires will decrease resilience. Loss of topsoil and creation of hydrophobic (water-repellent) soils after frequent, hot fires, can lead to loss of species over time (DeBano 1981; Wellner 1970). As sites become drier, there may be a shift away from mesic species to more xeric and fire-adapted shrubs, such as rubber rabbitbrush, yellow rabbitbrush, bitterbrush, and mountain big sagebrush.

Vulnerability to Climate Change: Low to Moderate

Of all the ecosystems in the IAP region, montane shrublands appear to have the lowest vulnerability to climate change (table 7.2) because of high species diversity, high sprouter diversity, wide range in elevation, and broad distribution of dominant overstory species. Even with increasing temperatures and uncertain precipitation, species of the

Table 7.4—Elevation ranges of species that dominate or codominate the overstory of mountain shrublands in the IAP region.

Species	Elevation	Source
	Feet	
Saskatoon serviceberry (*Amelanchier alnifolia*)	4,000-9,500	Welsh et al. (2008)
Utah serviceberry (*Amelanchier utahensis*)	3,000-9,000	Welsh et al. (2008)
Birchleaf mountain-mahogany (*Cercocarpus montanus*)	3,900-9,800	Cronquist et al. (1997)
Chokecherry (*Prunus virginiana*)	3,100-10,170	Johnson (2000)
Skunkbush sumac (*Rhus trilobata*)	2,900-7,700	Pendleton et al. (1989)
Thimbleberry (*Rubus parviflorus*)	4,700-9,000	Gucker (2012)
Scouler willow (*Salix scouleriana*)	Sea level-10,000	Anderson (2001)
Mountain snowberry (*Symphoricarpos oreophilus*)	4,000-10,000	Aleksoff (1999)

montane shrublands are probably the most capable of expanding into niches at higher elevations and onto adjacent, more mesic portions of the landscapes in which they occur.

Blackbrush Shrublands

Vegetation Type Description and Distribution

Blackbrush shrublands are very limited on National Forest lands in the IAP region, occurring at the lowest elevations on the southern edge of the region in the Spring Mountains National Recreation Area in Humboldt-Toiyabe National Forest and on the Moab District in Manti-La Sal National Forest. Distinct ecotypes of blackbrush (*Coleogyne ramosissima*) occur in the region: one entering the Great Basin and Semi Desert subregion from the adjacent Mojave Desert to the south, and the other in the Plateaus subregion (Richardson and Meyer 2012; Richardson et al. 2014). Communities are dominated by blackbrush with jointfir (*Ephedra* spp.), burrobush (*Ambrosia dumosa*), hedgehog cactus (*Echinocereus* spp.), spiny menodora (*Menodora spinosa*), various goldenbush species (*Ericameria* spp.), prickly pear (*Opuntia* spp.), Apache plume (*Fallugia paradoxa*), and others sometimes present as subdominants. Historically, interspaces in these communities were probably mostly bare, even during years of higher precipitation, because of competition from blackbrush (Brooks et al. 2007). Perennial grasses and seral shrubs probably occurred sporadically in areas where blackbrush cover was low (Brooks et al. 2007).

Sensitivity to Climate Change: Low to Moderate

As a long-lived, stress-tolerant shrub, blackbrush has relatively low sensitivity to the direct effects of climate variability in the absence of disturbance (Kitchen et al. 2015). It sheds its microphyllous leaves in response to drought stress, is well adapted to high temperatures (Munson et al. 2011; Summers et al. 2009), and occurs on shallow soils with a rooting system that allows it to capture soil water opportunistically (Schwinning et al. 2005, 2008).

Adaptive Capacity: Low

There is a high level of genetic differentiation between populations of blackbrush that occur in the Mojave Desert (those of the Spring Mountains in southern Nevada) and those of the Colorado Plateau (Dixie and Manti-La Sal National Forests), which has implications for population persistence and migration in response to climate change (Richardson and Meyer 2012; Richardson et al. 2014). Pendleton et al. (2015) found records that indicate blackbrush has the ability to migrate in response to changes in climate, but that the rate at which climate change is expected to occur may preclude natural migration because of its episodic recruitment. In addition, blackbrush communities have little resistance to invasive plant species and very low resilience to the fires accompanying the increase in invasive annual grasses. Large areas of blackbrush in the Mojave Desert, where red brome (*Bromus rubens*) has increased significantly, have burned in the past decade (Pendleton et al. 2015). Blackbrush does not sprout after fire, and the species is not regenerating in these burned areas (Pendleton et al. 2015).

Vulnerability to Climate Change: Moderate to High

Despite the low sensitivity of blackbrush to the direct effects of climate change, vulnerability is rated as moderate to high (table 7.2) because of its lack of resistance to invasion by exotic species and its inability to resprout following fire. With increased area burned under changing climate (Abatzoglou and Kolden 2013), a loss of dominance by blackbrush is likely to occur, dominance by invasive annual grasses will increase, and a subsequent increase in fire frequency and size may occur with increased horizontal fuel continuity.

With climate change, there may be some expansion of blackbrush communities onto adjacent sites that are currently higher in elevation or on sites that have somewhat higher available soil moisture. This expansion is more probable in the Plateaus subregion, where invasive species have had less impact on fire and existing blackbrush communities, and where some evidence exists for contemporary blackbrush migration (Kay 2015). Expansion is much less likely on National Forest lands close to the Mojave Desert, where replacement of blackbrush by invasive species is already resulting in net loss of the blackbrush vegetation type.

Salt Desert Shrublands

Vegetation Type Description and Distribution

North American salt desert shrublands are dominated by a mixture of drought- and salt-tolerant (halophytic) shrub, sub-shrub, and herbaceous species and occupy landscapes too dry or too salty to support sagebrush. Salt desert shrublands are a minor component on National Forest lands in the IAP region, occurring primarily in the Utah and Nevada portions of the region (fig. 7.7), where their distribution on National Forest lands is limited to lower elevations. However, this type is extensive on adjacent lands managed by the Bureau of Land Management in Nevada and Utah as well as outside the region in southeastern Wyoming.

Salt desert shrublands are dominated primarily by species belonging to the Chenopod plant family, such as greasewood (*Sarcobatus* spp.), fourwing saltbush (*Atriplex canescens*), shadscale (*A. confertifolia*), Gardner saltbush (*A. gardneri*) and close relatives, mat saltbush (*A. corrugata*), winterfat (*Krascheninnikovia lanata*), gray molly (*Bassia americana*), spiny hopsage (*Grayia spinosa*), iodine bush (*Allenrolfea occidentalis*), and seepweed (*Sueda* spp.), along with a variety of other shrub species (Blaisdell and Holmgren 1984). Perennial grasses are often codominant, with the relative importance of warm and cool-season species dependent on the reliability of seasonal moisture for the sites. Common warm season grasses can occur in areas with warm, wet summers, which occur where salt desert shrublands are found in the extreme southern portions of the region. Cool-season

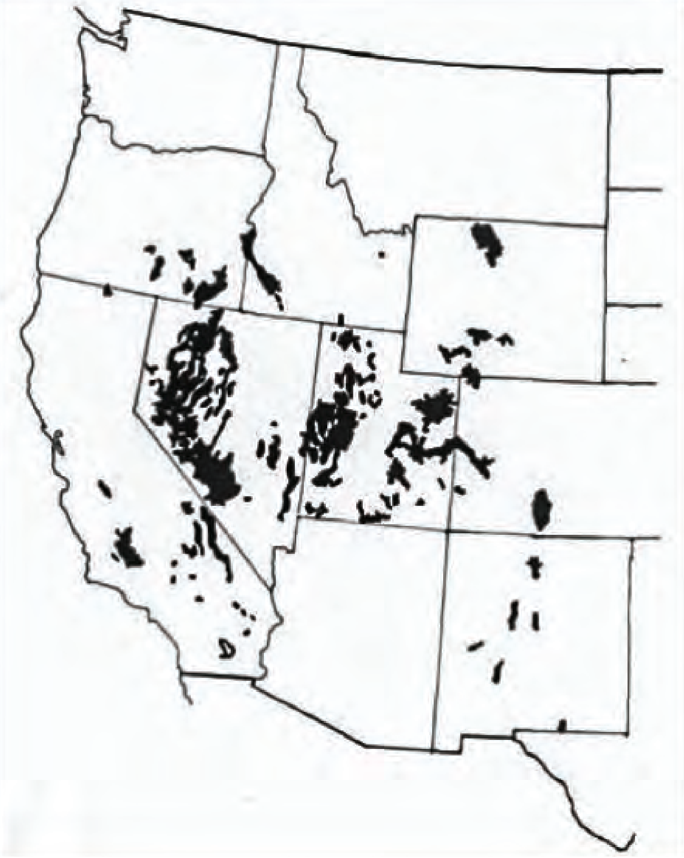

Figure 7.7—Distribution of salt desert shrublands in the western United States (derived from West [1983]).

grasses can occur elsewhere in the region, where spring is typically cooler and wetter (Paruelo and Lauenroth 1996). In the southern salt desert shrublands, warm season grasses include galleta grass (*Pleuraphis jamesii*), blue grama (*Bouteloua gracilis*), alkali sacaton (*Sporobolus airoides*), sand dropseed (*Sporobolus cryptandrus*), and saltgrass (*Distichlis spicata*). Common cool-season grasses include Indian ricegrass (*Achnatherum hymenoides*), squirreltail (*Elymus elymoides*), Salina wildrye (*Leymus salinus*), needle-and-thread grass (*Hesperostipa comata*), and purple three-awn (*Aristida purpurea*). A diverse mixture of native forbs responds opportunistically to variability in the timing and amount of precipitation and support a rich diversity of desert pollinators. Total plant cover in these communities is typically 20 percent or less, and height of shrubs is usually less than 1.5 feet (West 1983).

Cheatgrass establishment in dry salt desert communities is limited by low and sporadic precipitation (Meyer et al. 2001), but has been observed to be increasing. With increasing cheatgrass comes the potential for impacts from fire, which was not historically a significant disturbance factor (West 1994).

Sensitivity to Climate Change: Moderate

Many of the species associated with salt desert shrublands have wide ecological distributions and are tolerant of a wide range of climatic conditions. Species typically combine various morphological and physiological attributes (such as small, heavily protected leaves, high root-to-shoot ratios) that enable them to tolerate stress with others (such as seed and shoot dormancy) that facilitate stress avoidance. The effects of climate change on these plant communities will include both positive and negative shifts for individual species. However, the plant communities are likely to be relatively insensitive to the direct effects of climate change.

Adaptive Capacity: Low to Moderate

Many of the species that characterize salt desert ecosystems are fire intolerant (Chambers et al. 2013; Meyer et al. 2001). With the introduction of nonnative annual grasses, an increase in fine fuels may allow for increased area burned (West 1994), which would likely decrease the abundance of many characteristic species in this type. Disturbed salt desert shrublands are particularly susceptible to invasion by nonnative halophytic species such as halogeton (*Halogeton glomeratus*) and Russian thistle (*Salsola* spp.). Halogeton is a succulent summer annual that quickly spreads and establishes in disturbed areas (such as roadways and livestock watering areas) within intact perennial communities. It is a prolific seed producer, and seeds may remain in the soil seed bank for 10 years or more (Cronin and Williams 1966). Once established, halogeton prevents natural regeneration of native shrubs, such as winterfat (Eckert 1954; Harper et al. 1996; Kitchen and Jorgensen 1999) and Gardner saltbush (Goodrich and Zobell 2011). Today, halogeton stands are frequently found adjacent to remnant winterfat communities throughout the Great Basin (Kitchen and Jorgensen 2001).

Vulnerability to Climate Change: Moderate to High

With moderate sensitivity and low to moderate adaptive capacity, salt desert shrublands have moderate to high vulnerability to climate change (table 7.2). Risks of direct and indirect (i.e., fire) effects of introduced species render the vulnerability of this vegetation type relatively high to the combination of future impacts. Climate change is expected to result in more extreme precipitation events (West 1994). The combination of wet years and the fertilization effects of increased atmospheric carbon dioxide may result in an increase in annual grasses, which will in turn be more likely to fuel wildfire (Bradley et al. 2016; Salo 2005; Smith et al. 1987). Most of the dominant woody species in salt desert shrublands are poorly adapted to fire, and they will be vulnerable to increases in fire frequency.

Alpine Forblands and Grasslands

Alpine forblands and grasslands include communities dominated by either a variety of broadleaf forb species or by grasses, as well as the wide variety of species that occur in alpine ecosystems. Edaphic and climatic factors in these communities inhibit the establishment or significant growth of woody species.

Alpine Communities

Vegetation Type Description and Distribution

Alpine ecosystems occur at the highest elevations above treeline in the IAP region, at elevations generally above 8,000 feet in the northern portions to over 11,000 feet in the south. Although relatively small in area, they have high aesthetic value and high visitor and recreational use. They are possibly the ecosystems in the IAP region that are most at risk from the effects of climate change because of their shrinking habitat. These high-elevation locations are characterized by a very short growing season. Alpine plant communities are diverse and complex across the IAP region (Hayward 1952) and can include a variety of growth forms, including upland krummholz, shrubland, grassland, and herbaceous communities, herbaceous wetlands, and sparsely vegetated bedrock and scree communities (NatureServe 2013). These diverse types have been combined here because of the relatively small area they cover within the region, and because they are expected to have similar (negative) responses to climate change.

Sensitivity to Climate Change: High

The length and depth of snow cover, which are strongly correlated with mean temperature and precipitation, are key factors controlling alpine ecosystems (Beniston 2003). Snow cover provides frost protection for alpine plants in the winter, as well as the water supply in spring. Reduced snowpack with warming is likely to cause major changes in alpine plant communities (Gottfried et al. 2012). Alpine plants may be at greater risk from competition from plant species that are adapted to warmer temperatures and longer growing seasons. Research from the European Alps showed a significant increase in species richness in alpine ecosystems with the invasion of plants from lower altitudes (Pauli et al. 2003). It will be essential to account for new competitive interactions among species to better predict the responses of individual alpine species and entire communities to climate change (Alexander et al. 2015).

Adaptive Capacity: Low

The adaptive capacity of plant species in alpine ecosystems to climate change is likely to be low (table 7.2) because they have limited geographic space into which they can expand, and they are isolated communities (Alo and Wang 2008). In addition, the physiological traits that allow their persistence in alpine climates also reduce their ability to adapt to changing climates. The fate of individual species in a changing climate is likely to depend on their individual ecophysiological responses to the direct effects of increased temperatures and carbon dioxide levels, as well as the indirect effects of rising temperatures, such as the length of the snow-free period (Pauli et al. 2003).

The introduction of mountain goats (*Oreamnos americanus*), a nonnative species, to nearly every mountain range in Utah with alpine vegetation and the Ruby Mountains in Nevada has the potential to impact existing native vegetation, introduce noxious and invasive weeds, and result in a significant reduction of ground cover. As a result, there is potential for loss of native plants from trampling and subsequent soil erosion, further decreasing the capacity of alpine plant communities to adapt to climate change.

Vulnerability to Climate Change: Very High

The composition and distribution of alpine ecosystems will be affected by decreasing snowpack. For high-elevation vegetation, climate change may affect seed germination and survival by modifying moisture availability and therefore result in reduced plant success. Specific effects will depend on vulnerability thresholds of the characteristic species and the rate and magnitude of changes over time (Beniston 2003). In addition, climate change could lead to a mismatch between plant flowering and pollinator emergence (Parmesan 2006), which could adversely affect both plants and pollinators.

Alpine communities often have a relatively high number of endemic species because they are isolated (Beniston 2003), meaning that highly endemic alpine biota will have a disproportionately high risk of extinction (Parry et al. 2007). Local extinctions of otherwise widespread alpine species such as arctic gentian (*Gentianodes algida*) and alpine chaenactis (*Chaenactis douglasii* var. *alpina*) have already occurred in portions of Idaho because of habitat loss and fragmentation (USEPA 1998). Warming temperatures and longer growing seasons are likely to allow more competitive shrubs, trees, and herbs to expand upslope from adjacent ecosystems and potentially outcompete existing alpine vegetation (Alexander et al. 2015).

Mountain Grasslands

Vegetation Type Description and Distribution

Grasslands are areas where grasses and grass-like species dominate and trees and shrubs have no more than a minor presence. Forbs are typically present, although forb abundance and diversity vary, and forbs are subdominant to grasses. Grasslands that occur on the mid- to high-elevation landscapes are composed primarily of perennial cool-season bunchgrasses. Typical species for higher elevations include slender wheatgrass (*Elymus trachycaulus*), mountain brome (*Bromus carinatus*), needlegrasses (*Achnatherum* spp.), and blue grasses (*Poa*). Dominant species at middle elevations may include bluebunch wheatgrass, Idaho fescue (*Festuca idahoensis*), and Hood sedge (*Carex hoodii*). On many sites that have transitional winter- and summer-dominant weather patterns, a mixture of cool and warm season grasses can coexist.

Boulder Top Mountain on the Aquarius Plateau in Dixie National Forest has broad landscapes dominated by a low-growing fescue that was historically included in the sheep fescue (*Festuca ovina*) complex. Welsh et al. (2008) note that there are native forms of *Festuca ovina*, whereas another database (NRCS 2017) indicates that this species is entirely introduced. The ecosystems on the Aquarius Plateau appear to be native grasslands and are treated as such here.

These apparent native grassland communities occur on one of the largest contiguous flat-top landscapes above 11,000 feet elevation in the IAP region. Youngblood (1980) also recognized grassland communities on the Bridger-Teton National Forest dominated by spike fescue (*Leucopoa kingii*), bluebunch wheatgrass, and Idaho fescue. This important cover type is very limited in distribution in the IAP region.

Sensitivity to Climate Change: Moderate

Determining the sensitivity of grasslands to climate change is complex. As noted earlier, cool-season grasses occur in areas with cool, wet springs, which occur throughout most of the IAP region. Warm season grasses occur in areas with warm, wet summers, which occur at lower elevations in the southern portion of the IAP region (Paruelo and Lauenroth 1996). Some studies, based solely on projected increases in temperature, suggest that grasslands dominated by cool-season grasses may decline and that grasslands dominated by warm season grasses could, at the same time, expand into those environments. To further complicate this assessment, the increased atmospheric carbon dioxide favors cool-season grasses and enhances biomass production. However, warming favors warm season grasses because of increased water-use efficiency (Morgan et al. 2004, 2007). For these reasons, we cautiously rank sensitivity of these ecosystems as moderate (table 7.2).

Adaptive Capacity: Low to Moderate

The adaptive capacity of these grassland communities is rated as low to moderate because of historical impacts, and inherent adaptive capacity of species dominating these sites (table 7.2). Many low-elevation grasslands have been converted to agricultural use. Those grasslands that remain, particularly at lower elevations, are often highly disturbed, fragmented, and frequently occupied by many nonnative invasive plant species (Finch 2012). More frequent or severe fire associated with climate change may encourage further expansion of invasive species in grasslands, especially at lower elevations where adjacent landscapes are dominated by annual grasses (Bradley et al. 2016).

Vulnerability to Climate Change: Moderate to High

With moderate climate sensitivity and low to moderate resilience, these cool-season grass-dominated communities are rated as having a moderate to high vulnerability to climate change (table 7.2). Although some studies suggest that cool-season grasses will respond positively to increased carbon dioxide levels, other models show that these same species will decline because of increasing temperatures. Warm season grasses have been shown to be favored by increased temperatures alone because of increased water-use efficiency (Morgan et al. 2004, 2007); thus, they may have a competitive advantage over cool-season grasses and could expand into the region from warmer and drier climates to the south. Increasing fire would also encourage more invasive species in grasslands (Bradley 2009; D'Antonio and Vitousek 1992), converting many warmer and drier systems to invasive annual grasslands.

Subalpine Forb Communities

Vegetation Type Description and Distribution

Subalpine forb communities are upland communities dominated by non-grass herbaceous species, commonly called forbs (Ellison 1954; Shiflet 1994). Grasses are typically present but are subdominant (Shiflet 1994). If present, trees and shrubs constitute only a minor element of these communities (Ellison 1954). Subalpine forb communities occur at moderate to high elevations (7,000–11,000 feet) where forb growth and reproduction are favored by topographic, edaphic, and climatic conditions (Shiflet 1994). Mean annual precipitation is 25 to 40 inches. These communities can be found in various patch sizes, from small subalpine meadows to a dominant vegetation type covering miles of ridgetops and gentle slopes. They are most extensive in areas where midsummer thunderstorms of late July to mid-August coincide with the prime flowering season in the subalpine zone. Subalpine forb communities merge into mountain sagebrush-steppe, subalpine conifer forest, and aspen forest ecosystems and share numerous species with each (Ellison 1954).

Subalpine forb communities are limited in the IAP region (fig. 7.8). Subalpine forb communities are especially prominent on the Wasatch Plateau in central Utah (Ellison 1954), in the Teton Range of the Idaho-Wyoming border, and in the Wind

Figure 7.8—Distribution of tall subalpine forb communities (in orange) in the Intermountain Adaptation Partnership region. Forested area is shown in green (from U.S. Forest Service, http://www.fs.fed.us/wildflowers/beauty/Tall_Forb/what.shtml).

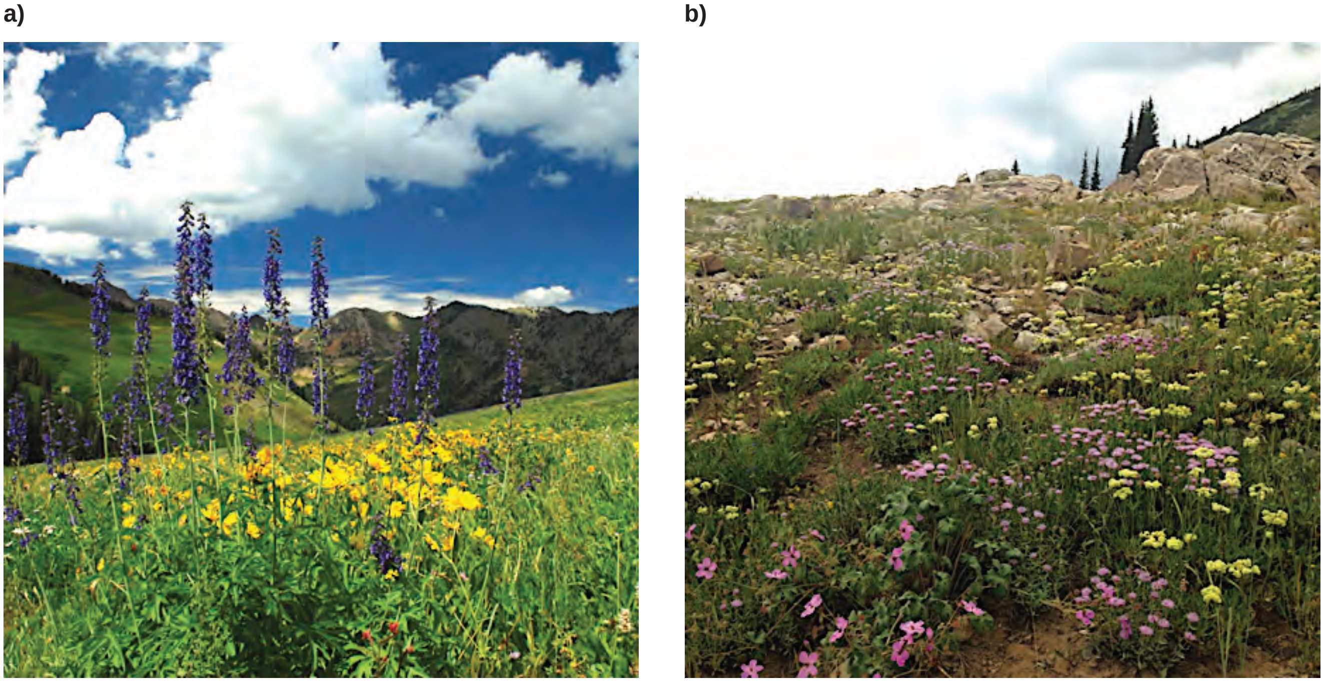

Figure 7.9—Subalpine forb communities in the Intermountain Adaptation Partnership region: (a) a subalpine tall forb community occurring on deep, productive soils, and (b) a subalpine low forb community occurring on shallow, well-drained soils (photos: W. Padgett, U.S. Forest Service).

River Range of western Wyoming (Gregory 1982). These communities have also been found in the Jarbidge Mountains and Ruby Mountains of northern Nevada (Lewis 1971, 1975; Loope 1970) and in small amounts elsewhere in the region.

Subalpine forb communities are characterized by high vascular plant species diversity. For example, 54 forb genera (65 native, mostly perennial species) representing 22 families are found on 30 acres of the Elk Knoll Research Natural Area administered by the Manti-La Sal National Forest (unpublished records on file at the USFS, Shrub Sciences Laboratory, Provo, Utah). In addition to forbs, 11 grass and 11 shrub species are found at subdominant to incidental levels.

Subalpine forb communities include species assemblages classified as tall forb, which are typically associated with deep soils (fig. 7.9a) (Shiflet 1994), as well as assemblages of short forbs that occur on well-drained, typically shallow and rocky soils (fig. 7.9b). In addition, there are a variety of mixed and intermediate phases that combine elements of each. Common tall forb species include false hellebore (*Veratrum californicum*), false springparsley (*Pseudocymopterus montanus*), western sweetroot (*Osmorhiza occidentalis*), licorice root (*Ligusticum filicinum*), biscuit root (*Lomatium* spp.), valerian (*Valeriana* spp.), one-flower helianthella (*Helianthella uniflora*), showy goldeneye (*Viguiera multiflora*), geraniums (*Geranium* spp.), peavine (*Lathryus* spp.), lupines (*Lupinus* spp.), American vetch (*Vicia americana*), elk weed (*Fraseria speciosa*), larkspur (*Delphinium* x*occidentale*), columbine (*Aquilegia* spp.), jacobsladder (*Polemonium foliosissimum*), bluebells (*Mertensia* spp.), asters (*Symphiotrichum* spp.), and paintbrushes (*Castilleja* spp.), among many others. Common shorter forbs include various buckwheats (*Eriogonum* spp.), yarrow (*Achillea millifolium*), agoseris (*Agoseris* spp.), scarlet gilia (*Ipomopsis aggregata*), beebalm (*Mondardella* spp.), cinquefoil (*Potentilla* spp.), penstemons (*Penstemon* spp.), groundsels (*Packera* spp.), and paintbrushes (*Castilleja* spp.). Common grasses include slender wheatgrass, mountain brome, Porter brome (*Bromus porteri*), bluegrass (*Poa* spp.) and needlegrasses.

Sensitivity to Climate Change: High

Species that occur in subalpine forb communities occur across a broad elevational range and occupy a wide variety of habitats, but little literature is available regarding the specific requirements for the establishment and maintenance of these ecosystems. Soil characteristics are critical for preservation of the tall forb assemblages (Lewis 1993). Where those deep soils have eroded, the type has been compromised and in some cases, sites are no longer capable of maintaining species that once dominated (Shiflet 1994). These communities respond to summer rainfall, and it is unclear whether these precipitation events will increase or decrease in frequency and amounts with changing climate. However, higher temperatures will lead to reduced soil moisture and are likely to alter the conditions necessary to support these unique ecosystems. Although species in these communities may be able to move to higher elevations with warming, lack of soil development at higher elevations may prevent their establishment.

Adaptive Capacity: Moderate

Many acres of this cover type have been degraded or lost because of historical livestock grazing at unsustainable levels (Ellison 1954; Lewis 1993; McArthur et al. 2013). Heavy grazing has resulted in a loss of productive topsoil in many places, which limits the establishment and growth of many dominant native species (Shiflet 1994). On the Wasatch Plateau of central Utah, Lewis (1993) found significant improvement in conditions once livestock were removed from sites that had lost tall forb species through excessive grazing in the late 1800s. This is not always the case; intensive grazing by livestock and subsequent loss of topsoil can result in establishment of species such as tarweed (*Madia glomerata*) that can remain in place for years (Shiflet 1994). Because much of the area in subalpine forb communities is in a degraded condition, adaptive capacity is rated as moderate (table 7.2).

Vulnerability to Climate Change: High

Although some subalpine forb communities may be able to move higher in elevation where current alpine environments occur, the lack of soil development at higher elevations may support only the lower-growing species found in this vegetation cover type. In some areas, such as the Wasatch Plateau in central Utah, the tall forb communities occur at the highest elevations of the plateaus, and therefore the vulnerability to the communities is high to very high. In addition, increased drought stress with higher temperatures is likely to stress species in these communities. The overall vulnerability of this type to climate change is therefore high (table 7.2).

Riparian and Wetland Communities

Riparian and wetland communities occupy about 1 percent of the land surface in the Great Basin (Sada 2008), and they very likely occupy about that same percentage of the landscape throughout the IAP region. Though the area in these types is relatively small, they have very high species diversity and support a variety of ecosystem functions (Naiman and Dècamps 1997). From high to low elevation, riparian and wetland communities throughout the region have been subjected to relatively high impacts from human uses, including road construction, land development, conversion to agricultural uses, and changes in stream discharge because of dam construction and water diversions. In addition, these areas have been affected by intensive use by domestic livestock, beaver removal, and nonnative species (Sada 2008).

Riparian and wetland communities are described by elevation in this report. This organization was chosen because of differences in stream size, localized climates, species composition and associated structure, and processes such as erosion, transport, and deposition that dominate these communities at different elevations. Historical and current impacts and threats and predicted responses to climate change also tend to vary by elevation.

High-elevation areas often have smaller and steeper stream channels, with some large snowmelt- and spring-fed wetlands. Where stream systems are characterized by steep gradients, they tend to be dominated by erosional processes. Riparian and wetland vegetation composition and sometimes structure vary with elevation (Engelhardt et al. 2015).

Middle elevations often have larger stream channels with lower gradients. They are dominated by transport processes, moving sediments from higher elevation to lower-elevation stream channels. Riparian and wetland vegetation composition and structure are highly variable, with trees, low and tall shrubs, and herbaceous species.

Low elevations have the largest channels and are often dominated by depositional processes. Most streams are alluvial and armored by riparian vegetation. Historically, the largest cottonwood gallery forests and natural wetlands occurred at lower elevations. Low-elevation riparian areas have a highly variable vegetation structure and contain trees, low and tall shrubs, and herbaceous species.

Across elevations, wetlands can vary in size and are dependent on water availability and site characteristics (e.g., valley bottom and associated stream type). Species composition varies with elevation. Upper-elevation wetlands are typically dependent on snowpack and snowmelt to sustain their water supply. They are often characterized by herbaceous species (sedges and rushes) but may also have low-growing willows as a community dominant. Drainage and development have eliminated many lower-elevation wetlands.

All riparian areas can be influenced by beaver activity, which results in ponding and flooding because of dam building. Historically, beaver occurred throughout the IAP region, except in the Great Basin. Much has been written on the hydrological and ecological roles that beaver populations play in riparian ecosystems (Jenkins and Busher 1979). Beaver dams can reduce peak discharge and stream velocity, and they can reduce sediment flows by increasing deposition in the ponded areas (Collen and Gibson 2001). Beaver dams also spread water over broad areas, expanding habitat for riparian and wetland species (Pollock et al. 2003). The widespread removal of beaver has resulted in significant changes to stream hydrology, geomorphology, and ultimately the ability of valley bottoms to support healthy and diverse riparian and wetland ecosystems (Pollock et al. 2003). The introduction or reintroduction of beaver, however, does not always have a significant positive effect (Rosell et al. 2005). Locations for reintroductions must be carefully considered.

Sensitivity

Watershed geomorphic and hydrological characteristics, as well as climatic factors such as temperature, precipitation type, and precipitation amount, influence the volume and timing of streamflows (Patten 1998). Whereas base flow conditions result from the gradual release of groundwater and snowmelt, periodic flooding can result from either rapid spring snowmelt or high-intensity summer thunderstorms. The distribution, health, composition, and maintenance of riparian communities depend on volume and timing of streamflows (Auble et al. 1994; Poff et al. 1997; Scott et al. 1996, 1997; Stromberg 1993; Stromberg and Patten 1995).

The Great Basin and Semi Desert and Plateaus subregions are among the driest areas in the western United States. Climate change is likely to have the greatest effects in these relatively hot and dry portions of the region (Perry et al. 2012). Water availability is projected to decrease because of increased drought, earlier runoff, and lower late-spring and summer streamflows. High flows required for channel maintenance will be reduced. Plant community composition and structure will be affected by increased water stress, and drought-tolerant species are likely to replace riparian and wetland species. In addition, geomorphic and hydrological processes and dynamics that have been responsible for riparian and wetland ecosystem development at lower elevations have already been affected by construction of dams and water diversions in most places.

Adaptive Capacity

From high to low elevations, most riparian and wetland systems have been altered from historical conditions, resulting in changes in stream geomorphic and hydrological processes, including stream downcutting and channel straightening. Stream discharge has been reduced because of dam construction and water diversions. These changes have decreased water availability to riparian ecosystems because of greatly reduced floodplain access and recharge. Riparian areas and wetlands have also been affected by domestic livestock grazing, road construction, and nonnative species (Sada 2008).

Riparian systems are inherently driven by frequent disturbances, in particular seasonal floods or high water flows (Kauffman 2001). These flows affect the movement and deposition of sediment and large woody debris (Nakamura et al. 2000). The flow regime of riparian systems is of primary importance in maintaining their ecological integrity (Poff et al. 1997). The magnitude, frequency, duration, timing, and rate of change of streamflows directly and indirectly affect water quality, energy sources, physical habitat, and the biotic interactions within the stream systems. The modification of any one of these can have a cascading effect on ecological integrity.

Changes in flow regimes, whether through climate change or through human-caused alterations such as those from water diversions and dams, impact the amount, season, and timing of flows. This can substantially alter associated riparian and wetland species because of their dependence on fluvial geomorphic process, surface water, and groundwater (Merritt et al. 2010; Nilsson and Berggren 2000; Poff et al. 1997). Floods are responsible for erosion, transport, and deposition of sediments, as well as the amounts and location of vegetation and debris. Many dominant riparian species, such as cottonwoods and willows, are pioneer species that depend on these events to provide bare, moist substrates necessary for seed germination and plant establishment (Cooper et al. 2003; Scott et al. 1996; Stella et al. 2011).

Vulnerability

Factors considered in characterizing the vulnerability of each riparian and wetland vegetation type to climate change include regeneration success, response to disturbance (changes in amount, timing, and location of runoff), and plant life history traits.

High-Elevation Riparian and Wetland Communities

Vegetation Type Description and Distribution

High-elevation riparian and wetland communities include forests, shrublands, and herbaceous communities occurring in meadows, adjacent to streams and water bodies, or around seeps and springs. High-elevation wetland sites are often associated with bogs, fens, springs, and streams at low-gradient sites, such as glacial cirque floors and slumps, or around small lakes and ponds proximal to high ridgelines. These communities generally occur above 8,500 feet elevation throughout the IAP region. As noted previously, upper-elevation streams are erosional in nature, providing sediments to their connected systems.

Forest communities occur near the boundary between high- and mid-elevation riparian and wetland communities and can include species such as aspen and conifers, including subalpine fir (*Abies lasiocarpa*) and spruce (*Picea* spp.); cottonwoods generally do not occur at these elevations. Low-growing willows such as Wolf's (*Salix wolfii*) and plainleaf (*S. planifolia*) can dominate broad meadows, along with other shrubs such as resin birch (*Betula glandulosa*) and bog blueberry (*Vacinnium occidentale*). Some tall willows, such as Drummond's (*Salix drummondiana*), may also occur. High-elevation sedges (e.g., *Carex aquatilis*, *C. illota*, *C. limosa*, *C. scopulorum*, *C. luzulina*) can dominate these wetland and riparian systems, along with tufted hairgrass (*Deschampsia cespitosa*) and alpine bentgrass (*Agrostis humilis*).

Existing stressors in high-elevation riparian and wetland communities include drought, livestock grazing (particularly domestic sheep), and grazing by both introduced ungulates (e.g., mountain goats), and large populations of native ungulates. In addition, recreational uses can be significant, especially in areas adjacent to high populations and relatively easy access. Roads in the valley bottoms are a major factor affecting erosional processes. Improper all-terrain vehicle use can also cause severe soil and vegetation damage, particularly in seasonally wet riparian areas, meadows, and peatlands.

Sensitivity to Climate Change: Moderate to High

Warming temperatures and reduced snowpack may result in the loss of high-elevation riparian and wetland habitats, resulting in drier, less productive systems. With rising temperatures, frigid snow- and water-dependent ecosystems in the upper portions of watersheds will have very little room to move upslope. Elevating temperatures will increase competition from riparian species now occurring at lower elevations, and smaller snowpacks will increase competition from upland species that occupy drier sites.

Adaptive Capacity: Low to Moderate

Although these ecosystems have been less impacted by humans than mid- and low-elevation riparian and wetland communities, existing stressors still include drought, livestock grazing, introduced ungulates (e.g., mountain goats), and large populations of native ungulates, as well as some recreational uses. There tend to be few invasive species in these high-elevation ecosystems, and because of historically late seasonal snow cover and associated later plant growth, these ecosystems have had shorter grazing seasons by domestic livestock. Like riparian and wetland species of mid- to lower elevations, nearly all tree species occurring in these areas sprout following fire. These combined factors result in a low to moderate adaptive capacity for these communities (table 7.2).

Vulnerability to Climate Change: High

High-elevation riparian and wetland communities have a high vulnerability to climate change because of moderate to high sensitivity and low to moderate adaptive capacity (table 7.2). Mid-elevation riparian and wetlands communities are likely to move higher in elevation with warming climate. Systems currently in place are in danger of losing their water source, and soil moisture is likely to be reduced as snowpack amount and duration decrease.

Mid-Elevation Riparian and Wetland Communities

Vegetation Type Description and Distribution

Mid-elevation riparian and wetland communities include forests, shrublands, and herbaceous communities occurring adjacent to streams, in wet meadows, and surrounding water bodies, or proximal to seeps and springs. These communities generally occur between 5,500 and 8,500 feet throughout the IAP region. As noted earlier, mid-elevation streams transport sediments from these and higher-elevation riparian areas to the lower-elevation systems.

Mid-elevation riparian communities may be dominated by a variety of tree, shrub, and herbaceous species. Tree species, such as narrowleaf cottonwood (*Populus angustifolia*), quaking aspen (*P. tremuloides*), western river birch (*Betula occidentalis*), and thinleaf alder (*Alnus incana*) occur in these areas. Conifer species dominating adjacent landscapes, such as Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), Engelmann spruce (*Picea engelmannii*), and blue spruce (*P. pungens*) may also occur at stream edges. Shrubs include mid-elevation willows, such as Booth's willow (*Salix boothii*), Drummond's willow, shining willow (*S. lucida* subsp. *caudata*), and dusky willow (*S. melanopsis*), and a variety of herbaceous meadow and wetland species. At the lower range of these communities, Nebraska sedge (*Carex nebrascensis*) can dominate meadows, along with tufted hairgrass.

Sensitivity to Climate Change: Moderate to High

Riparian areas, because of their high water tables, have some of the highest capacity to adapt to changing climates. However, as snowpacks are reduced and seasonality of runoff changes, the amount of water available for subsurface storage is likely to be reduced. Increasing temperatures will increase competition from invasive and riparian species from lower elevations, and reduced water tables will increase competition from adjacent upland species. Thus, the species composition of these riparian areas could change considerably in a changing climate.

Adaptive Capacity: Moderate

Adaptive capacity of these mid-elevation riparian and wetland ecosystems is moderate (table 7.2) and may be less in areas subjected to a wide variety of human influences. Historically, these ecosystems were affected by heavy livestock grazing. In addition, these areas have been used as locations for road construction, concentrated recreational uses, and several other developments. Many nonnative invasive species, such as Canada thistle (*Cirsium arvense*), nodding plumeless thistle (*Carduus nutans*), scotch thistle (*Onopordum acanthium*), and bull thistle (*Cirsium vulgare*), occur in these habitats. Waterways provide a means for dispersing these species widely. Because of the high level and variety of human impacts on these riparian ecosystems, many of these mid-elevation communities have lost resilience. These systems typically have high fuel moisture and are not very susceptible to wildland fire. When fires occur, however, they often move from adjacent upland communities into these environments (Dwire and Kauffman 2003).

Vulnerability to Climate Change: Moderate to High

Climate change vulnerability of mid-elevation riparian and wetland communities is rated as moderate to high because these communities have moderate to high sensitivity and moderate adaptive capacity to the effects of climate change (table 7.2). Mid-elevation riparian plant species may have the ability to move upward in elevation, but where resilience has been compromised by human uses, these systems may not be able to easily adjust to changes in their environment. Invasive species that already dominate many mid-elevation sites are likely to expand their dominance. As riparian areas become drier, upland species will continue to expand into these sites.

Low-Elevation Riparian and Wetland Communities

Vegetation Type Description and Distribution

Low-elevation riparian and wetland communities include forests, shrublands, and herbaceous communities occurring adjacent to streams and water bodies, in meadows, or around seeps and springs. These communities generally occur below about 5,500 feet throughout the IAP region. Lower-elevation streams are generally where sediments from mid- and upper-elevation sources are deposited.

These riparian communities may be dominated by a variety of tree, shrub, and herbaceous species. Tree species include narrowleaf cottonwood, lanceleaf cottonwood (*Populus* ×*acuminata*), Fremont cottonwood (*P. fremontii*), black cottonwood (*P. balsamifera* ssp. *trichocarpa*), and box elder (*Acer negundo*), as well as a wide variety of nonnative tree species. Shrubs include a wide variety of willows, such as yellow willow (*Salix lutea*), Geyer willow (*S. geyeriana*), Booth willow, Pacific willow (*S. caudata*) and narrowleaf willow (*S. exigua*). Beaked sedge (*Carex utriculata*), Nebraska sedge, and Baltic rush (*Juncus arcticus*) grow at many elevations, but are typically common at lower elevations. Low-elevation wetland and riparian communities are limited in their occurrence on National Forest System lands throughout the region because most of these habitats occur near or below the forest boundaries.

Sensitivity to Climate Change: High

Although riparian and wetland species at lower elevations in the IAP region may not be adapted to increasing temperatures, species from adjacent geographic areas could replace species that currently dominate these ecosystems. However, the low-elevation riparian and wetland communities are more likely to be affected by decreased flows and water availability through continued diversions. In addition, the timing of water availability (because of lower snowpacks) is likely to affect species with high water demands throughout the summer. Changes in the amount and timing of runoff events could greatly impact water tables and soil moisture relationships and eliminate much of the riparian and wetland habitats that remain at these lower elevations. Much has been written on the hydrological requirements for the germination of various cottonwoods and willows (Auble and Scott 1998; Mahoney and Rood 1998; Siegel and Brock 1990; Young and Clements 2003). The connections among changes in climate, hydrology (timing and amount of flows), and the ability of these species to continue to germinate and establish are only now being investigated (Gori et al. 2014; Smith and Finch 2016; Stromberg et al. 2010). However, climate change has the potential to greatly affect the ability of these woody riparian dominant species to germinate and establish in the future; accordingly, low-elevation riparian and wetland ecosystems are rated as highly sensitive to climate change (table 7.2).

Adaptive Capacity: Low to Moderate

Many low-elevation riparian and wetland communities have been degraded from a wide variety of human influences (e.g., fig. 7.10), such as road construction, concentrated recreational uses, and other development. These areas have also been subjected to excessive, unmanaged livestock grazing, especially in the past. Management efforts by Federal agencies since the early 1980s have focused on reducing impacts and improving conditions of these systems.

As a result of historical land uses, many nonnative invasive species occur in these habitats. For example, these areas have had some of the greatest increases in nonnative invasive woody species, such as tamarisk (*Tamarix chilensis*, *T. ramosissima*) and Russian olive (*Elaeagnus angustifolia*), as well as nonnative invasive herbaceous species. Many of these herbaceous species are listed as noxious. Many low-elevation wetlands in the region have become dominated by the nonnative common reedgrass (*Phragmites australis*) (fig. 7.11). Purple loosestrife (*Lythrum salicaria*) can also invade wetlands and replace existing native wetland species.

Vulnerability to Climate Change: High to Very High

The direct effects of reduced flows and changes in timing and duration of spring runoff because of climate change will reduce resilience in low-elevation riparian and wetland communities, and thus their vulnerability to climate change is rated as high to very high (table 7.2). These systems have also been affected by upstream diversions of water and wetland drainage, and by livestock grazing, development, road construction, and concentrated recreational uses. Additional pressures on these already vulnerable ecosystems could have significant effects in the future.

Figure 7.10—Heavily grazed riparian area. Heavy livestock grazing in riparian areas inhibits regeneration and growth of woody riparian species such as cottonwoods and willows (photo: W. Padgett, U.S. Forest Service).

Figure 7.11—Common reedgrass that has invaded and dominated low-elevation wetlands along the Great Salt Lake in northern Utah (photo: W. Padgett, U.S. Forest Service).

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